Hard questions, concrete solutions: mitigating the ecological impacts of the global infrastructure boom

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Til Mor og Far. Jeg ved, i ville være stolt.

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Abstract

By 2060, an estimated >230 billion m^2 of additional built floor area will be added to the global building stock, equivalent to the built area of Japan each year. Effective tools are urgently required to mitigate the ecological impacts of this global infrastructure boom. In this thesis I explore the effectiveness of one of the most high-profile tools, biodiversity offsetting. I review the implementation and outcomes of biodiversity offsetting around the world, and identify large evidence gaps around its effectiveness. To address these I then evaluate the outcomes of one of the world's oldest biodiversity offsetting systems (Victoria, Australia), and pre-emptively evaluate one of the world's newest compensation systems ("Biodiversity Net Gain", England). Both evaluations indicate that these compensation systems are unlikely to fully mitigate the ecological impacts of development. In Victoria, we find preliminary evidence of self-selection bias undermining the additionality of offsets, and in England, we identify serious governance gaps that leave the majority of the policy's biodiversity benefits unenforceable. Both of these systems implement regulatory offset markets, so I then explore the economics of offsetting regulatory markets, and identify one barrier to their successful implementation may be contradictions between the way that biodiversity needs to be treated to create effective market-like mechanisms, and effective ecological outcomes. Recognising that there may be systemic barriers to biodiversity offsetting fully mitigating the impacts of development, I finally explore whether it is possible to create economies in which the rate of infrastructure expansion is fundamentally slowed. I use the case study of the housing crisis in England to explore whether the projected rates of infrastructure expansion are compatible with national carbon and biodiversity targets, and identify policies that might dampen the drivers behind rapid infrastructure expansion and its ecological impacts. Slowing the rate of infrastructure expansion without sacrificing human wellbeing appears possible, but it faces a daunting political economy.

Keywords: biodiversity offsetting; impact evaluation; mitigation hierarchy; biodiversity net gain; infrastructure sustainability; ecological economics; postgrowth economics; net zero

Table of contents

List of tables

List of figures

Chapter 1 Introduction

1.1 The state of biodiversity

Since the industrial revolution, the world has experienced an exponential increase in the physical scale of the human economy (Steffen et al. 2015a). This economic expansion has historically helped increase the living standards of the majority of humanity. However, it has come with costs. Today, the rate of resource extraction and waste generation of the global economy now exceeds the planet's capacity to replenish those resources or process its wastes, risking destabilising the Earth system (Daly & Farley 2011). One of the major losers from rapid industrialisation has been biodiversity, or the diversity of living things within the natural world. On average, monitored wildlife populations have fallen by approximately 69% over the last 40 years (WWF 2022), with wide variations between taxa and geographies in the winners and losers of these planetary changes (Blowes et al. 2019; Leung et al. 2020). Biodiversity plays a multitude of essential roles for society, including through the direct use of natural products, contributing to the resilience of ecosystems that we rely on for biophysical stability, and through contributions to human culture (Díaz et al. 2019). Therefore these ongoing declines generate major risks to human society: as stated in the 2021 Dasgupta review on the Economics of Biodiversity (Dasgupta 2021): "If, as is nearly certain, our global demand continues to increase for several decades, the biosphere is likely to be damaged sufficiently to make future economic prospects a lot dimmer than we like to imagine today. What intellectuals have interpreted as economic success over the past 70 years may thus have been a down payment for future failure."

The drivers of biodiversity loss are complex and diverse, with the three predominant drivers (measured in terms of the number of species on the IUCN Red List threatened by these mechanisms) being species overexploitation, agriculture, and infrastructure (Maxwell et al. 2016). This thesis focuses on reducing the biodiversity loss caused by land use change, predominantly through infrastructure expansion, but some of the policy tools explored in this thesis have also been used to regulate biodiversity losses from agriculturally-induced land use change.

1.2 The global infrastructure boom

The world is in the midst of the most rapid expansion in built infrastructure in human history (Steffen et al. 2015a; Krausmann et al. 2018), with economies around the world putting infrastructure investment at the heart of their post-coronavirus economic recovery strategies (OECD 2021a). As of 2020, the world's anthropogenic mass exceeded that of the biosphere, with most of that mass associated with concrete-based infrastructure (Elhacham et al. 2020). By 2040, it is projected that countries around the world will spend an additional $60 trillion on new built infrastructure (Global Infrastructure Hub 2018). Such investment is expected to yield an additional four million km of new roads by 2050 (Meijer et al. 2018), rapid expansions in energy and mining infrastructure (Zarfl et al. 2015; Sonter et al. 2020a), and the addition of 230 billion m^2 to the global building stock – equivalent to adding the built area of Japan each year (UNEP & IEA 2017). We have also entered the era of the infrastructure megaproject (defined as infrastructure projects costing >$1 billion; Flyvbjerg 2014); vast infrastructure projects implemented to achieve geopolitical or economic objectives that fundamentally alter the societies in which they are embedded (Flyvbjerg 2014). Such megaprojects include development corridors (Juffe-Bignoli et al. 2021), with the most ambitious coordinated series of development corridors being the Chinese Belt and Road initiative, aiming to connect two-thirds of the global population within a coherent transport, trade and industrial network (Ascensão et al. 2018).

The ecological impacts of this infrastructure boom are truly profound. Müller et al. (2013) project that, if the entire world were to develop levels of infrastructure stocks equivalent to those in today's wealthy countries, the embodied emissions of merely constructing all this infrastructure under 2013 production technologies would consume 350Gt CO_2 (the global carbon budget for a 50% chance of remaining within 1.5°C from 2021 onwards is 420Gt CO_2). Ultimately 79% of all global annual carbon emissions are tied to the construction and operation of built infrastructure (including energy infrastructure; UNOPS 2021).

The biodiversity impacts of this new infrastructure are similarly projected to be immense. Simkin et al. (2022) estimate that urban expansion over the next 30 years has the potential to threaten approximately 10,000 known species globally. Numerous studies have assessed the potential impacts of new development corridors on biodiversity, and concluded that development corridors are one of the major threats in several of the world's biodiversity and carbon storage hotspots, including Borneo, Sumatra, the Congo basin, Papua New Guinea, and the Amazon basin (Alamgir et al. 2019a, 2019b, 2020; Kleinschroth et al. 2019; Sloan et al. 2019; Vilela et al. 2020). New infrastructure networks impact on biodiversity through a range of mechanisms, including direct land use change, fragmentation effects, indirect effects beyond the immediate spatial boundaries of the project footprint and inducing further infrastructure proliferation (Laurance et al. 2015; Torres et al. 2016; Johnson et al. 2019; Tulloch et al. 2019).

However the biodiversity impacts of new infrastructure extend beyond the mere land cover impacts. New research is also beginning to uncover the biodiversity impacts of the raw materials used in infrastructure production associated with construction mineral and concrete supply chains (Torres et al. 2021; Torres et al. 2022; see Appendix 1). Torres et al. (2022) find that 612 species on the IUCN Red List threatened with extinction are impacted by the extraction of construction minerals, with over 24,000 expected to be threatened if the estimates are extrapolated using the methods used by IPBES to include unassessed species (Díaz et al. 2019; Purvis et al. 2019).

Accepting the need to remain within the planet's safe operating space (Steffen et al. 2015b), there are two facets to addressing the enormous ecological and climate threat posed by this ongoing infrastructure boom. One is greening the supply of new and existing infrastructure, by implementing policies to minimise and compensate for the ecological harms caused by the construction of new infrastructure. The other is addressing the ultimate driver of biodiversity loss, by reducing the need and demand for additional built infrastructure. Understanding the policies that compensate for the ecological harms caused by new infrastructure is the core aim of this thesis. In my final empirical chapter I then begin to explore the policy options for, and political economy of, reducing the rate of additional infrastructure expansion to reduce this fundamental driver of biodiversity loss.

1.3 Supply-side approaches: biodiversity offsetting

A family of policies have emerged to attempt to govern potential trade-offs between new infrastructure and biodiversity by implementing the mitigation hierarchy to the impacts of new developments, most commonly as part of the infrastructure planning and implementation process (see Chapter 2). The mitigation hierarchy is a framework for mitigating the biodiversity impacts of new developments or land use change (but it is being increasingly utilised in other contexts such as mitigating organisational biodiversity impacts; Bull et al. 2020; Milner-Gulland et al. 2021). The mitigation hierarchy framework is comprised of the avoidance, remediation, minimisation, and offsetting steps, with these steps theoretically implemented sequentially (i.e. with preference given to avoidance) with the aim of achieving an overall outcome of No Net Loss or Net Gain in biodiversity. Broadly speaking, avoidance aims to avoid unnecessary biodiversity impacts where possible (e.g. by relocating developments to areas of lower biodiversity value, or rejecting the need for the proposed development entirely; Phalan et al. 2018); minimisation focuses on reducing the impacts of developments as far as possible through high-quality project planning and the implementation of ecologically-sensitive construction methods; remediation aims to recover biodiversity through restoration when projects are temporary (e.g. post-mining); and offsetting aims to fully compensate for the residual negative biodiversity impact to deliver an overall outcome consistent with the policy goal.

Biodiversity compensation first emerged in the 1960s, and was first operationalised as contemporary offsetting in the US in the 1970s (Damiens et al. 2020). From the 1970s, the concept of offsetting pollution emissions gained traction as a method for making achieving regulatory compliance more flexible, marking the beginning of a series of policies that treated ecological harms as disconnected from their source in space and time, and advocating for compensating for those harms rather than tackling the source of pollutants directly (ibid). Wetland banking emerged in the 1980s, and then offsets began to spread globally as cultural attitudes towards regulation shifted from direct command-and-control approaches towards more flexible market-based mechanisms. To date, offsets are now embedded in some form in legislation (mostly national environmental impact assessment frameworks) in 37 countries across every continent except Antarctica (GIBOP 2019).

Biodiversity offsets are receiving increasing attention in policy and finance discussions around the world as governments and organisations seek to reconcile trade-offs between their development and biodiversity goals (see Chapter 2). The global database of implemented biodiversity offsets assembled by Bull & Strange (2018) shows that, as of 2018, the area of land under biodiversity offset management was approximately 150,000km^2 globally, demonstrating that offsets already span a spatial footprint that has the potential to make considerable contributions to nature conservation. Governments around the world have adopted, or are increasingly adopting, biodiversity compensation policies that institutionalise mechanisms for delivering biodiversity conservation actions that are coupled with development activities (GIBOP 2019, Chapter 2). In addition, offsets are receiving attention from the financial and corporate sectors for their perceived potential as an innovative financial mechanism for addressing global biodiversity funding shortfalls, with one high-profile industry analysis suggesting that offsets have the potential to contribute US$165 billion/year to addressing global conservation funding shortfalls by 2030 with favourable public policy (Deutz et al. 2020). Given an implementation history that spans over 50 years (Damiens et al. 2020), offsets are perceived by many as a relatively well-tested and scalable mechanism for directing private sector funding towards conservation, especially in contrast with other more recently-emerging and less well-tested financial instruments such as environmental impact bonds.

Despite this increasing attention, biodiversity offsets have been criticised on numerous theoretical grounds (Table 1). Given the extensive fundamental criticisms of offsetting, it would be reasonable to expect a high degree of confidence that offsets have been conclusively demonstrated to work. A core aim of this thesis is to assess the global state of knowledge regarding the effectiveness of biodiversity offsets. I review the peer-reviewed literature for evidence for the effectiveness of offsets around the world in Chapter 3. Ultimately identifying that there are in reality serious shortfalls in both the quantity and quality of evidence exploring the effectiveness of biodiversity offsetting, I then aim to address these evidence gaps by conducting two detailed studies to evaluate the outcomes or likely outcomes of jurisdictional offsetting policies (Chapters 4 and 6).

Critique	**Reasoning**
Economic	Biodiversity offsets are a defensive expenditure, and as such should not count as a mechanism for drawing private sector finance into restoration (Spash 2015).
Ethical	Offsets operationalise an instrumental and anthropocentric view of nature which is ultimately a driver of unsustainable behaviours and disconnection between people and nature, thereby further undermining transitions towards sustainability rather than assisting them (Ives & Bekessy 2015; Spash 2015; Apostolopoulou & Adams 2017)
Governance	Offsets trade often permanent biodiversity losses (in the case of hard infrastructure) for gains, but the governance mechanisms for overseeing the protection of these gains are unsuited to the task and are unlikely to be robust to future political or legal change (Damiens et al. 2021).
Biological	Biodiversity has co-evolved with that of its surroundings and as such is location-specific, and cannot therefore simply be re-located elsewhere. A focus on recreating specific biotopes or species elsewhere will not recreate the complex ecological relationships and evolutionary history present at the initial site (Moreno-Mateos et al. 2015)
Ecological	Trading biodiversity losses today for biodiversity gains implemented today but which will develop over time and pay off in future presumes a deterministic view of ecosystem restoration which is unrealistic because of the complexity of potential restoration trajectories and risks of project failure (Maron et al. 2012)
Political	The technocratic metrics and calculation methods underpinning offsets are used to depoliticise environmentally-damaging developments and exclude public engagement by re-framing the debate about whether or not the associated development should proceed as a technical question (Bormpoudakis et al. 2019)
	Offsets create a forum for debate in which motivated vested interests are able to express their preferences more strongly than the more diffuse counter-interests of the public. Implementing safeguards to prevent negative biodiversity outcomes but to the detriment of the function of the regulatory market is not in the interest of either the developers of the public officials, so biodiversity safeguards will tend to be eroded over time (Walker et al. 2009).

	Vested interests have historically captured offsetting initiatives to direct them away from a radical interpretation of offsets in which the true costs of development are internalised into the development process (Damiens et al. 2020).

Table 1. Summary of the major critiques of biodiversity offsetting as a conservation strategy

The key results of my evaluations are that neither of the biodiversity offsetting systems that I have researched in detail in this thesis are unambiguously on track to achieve no net loss of biodiversity, and so a key question is why: what explains why some biodiversity offsets succeed (e.g. Devenish et al. 2022) whilst others fail? In Chapter 5 I explore this question in more detail addressing a specific tendency that can be observed in many offsetting regulatory markets around the world – the tendency towards increasing the flexibility of biodiversity offsetting trading rules over time. I argue this tendency represents the deeper idea that, when offsets become restrictive on development (which means they are effectively internalising the impacts of biodiversity into the development planning process and avoiding damaging new development), they are susceptible to political alterations to weaken their transformative potential. This could be a core mechanism behind why regulatory offsetting markets often fall short on their stated policy goals.

1.4 Demand-side approaches: addressing fundamental drivers of infrastructure proliferation

The second major facet to mitigating the impacts of infrastructure and land use change as they occur is to develop a systematic understanding of the ultimate drivers of threats, and to attempt to address them by reducing the demand for additional infrastructure or land use change. This turns out to be an enormously complex, necessarily interdisciplinary and understudied research area. I argue in the thesis conclusion (Chapter 8) that this may be a vital frontier of conservation science research, and that there may be opportunities for better integration between conservation science and postgrowth economics to address the systemic drivers of biodiversity loss.

One of the key ambitions of the Sustainable Development Goals is the rapid expansion of infrastructure networks (embedded in SDG 9). There is clear evidence that providing additional infrastructure in regions where a lack of infrastructure is a fundamental barrier to the population having access to a key service can yield improvements in livelihoods and wellbeing (Thacker et al. 2019). For example, in a set of classic papers, Donaldson & Hornbeck (2016) and Donaldson (2018) demonstrate substantial welfare gains from the economic integration facilitated by the expansion of early transport infrastructure (rail networks) in the USA and India. Expansion of infrastructure networks is generally perceived as a sensible policy option for generating rapid employment in the construction sector and enabling future economic productivity gains (Thacker et al. 2019), and as such, there is widespread agreement in the policy community that additional infrastructure is desirable.

However, as discussed above, infrastructure also comes with a bundle of associated costs, often in the form of negative environmental externalities which are commonly underestimated or sometimes ignored. Research has also revealed biases towards justifying the construction of new infrastructure in commonly used infrastructure appraisal methods such as cost-benefit analysis (Næss 2016). Examples of infrastructure projects in which the economic costs outweigh the benefits include Vilela et al. (2020) demonstrating for a sample of planned new roads constructed in the Amazon basin that 45% of them deliver net negative economic benefits even without accounting for the costs of environmental externalities such as the associated deforestation and carbon emissions. Additionally, Chapman & Postle (2021) show that the failure to account properly for the carbon emissions associated with airport expansions in the UK amounts to a hidden subsidy from the UK taxpayer of £2.4-13.4 billion.

1.4.1 Has infrastructure expansion in wealthy countries become 'uneconomic'?

As mentioned, in areas where infrastructure relieves a constraint on people's access to fundamental goods and services, infrastructure is clearly desirable and essential. However, there has been little research or political discussion regarding whether these assumptions hold in wealthy nations with already-abundant infrastructure stocks. Haberl et al. (2019) demonstrate that, for countries with material stocks below 50t/capita, there is a clear linear correlation between increases in society's stock of built infrastructure and improvements in social welfare (Figure 1). However, past 50t/capita, this relationships dissolves. This data suggests that the welfare benefits of infrastructure might be subject to diminishing marginal returns which yield the classic satiation curve. If this is true, then it follows that there might be some threshold beyond which additional infrastructure is 'uneconomic' (i.e. the economic costs exceed the economic benefits; Daly 2013). If this threshold is exceeded – what are the political economic drivers of continued infrastructure expansion?

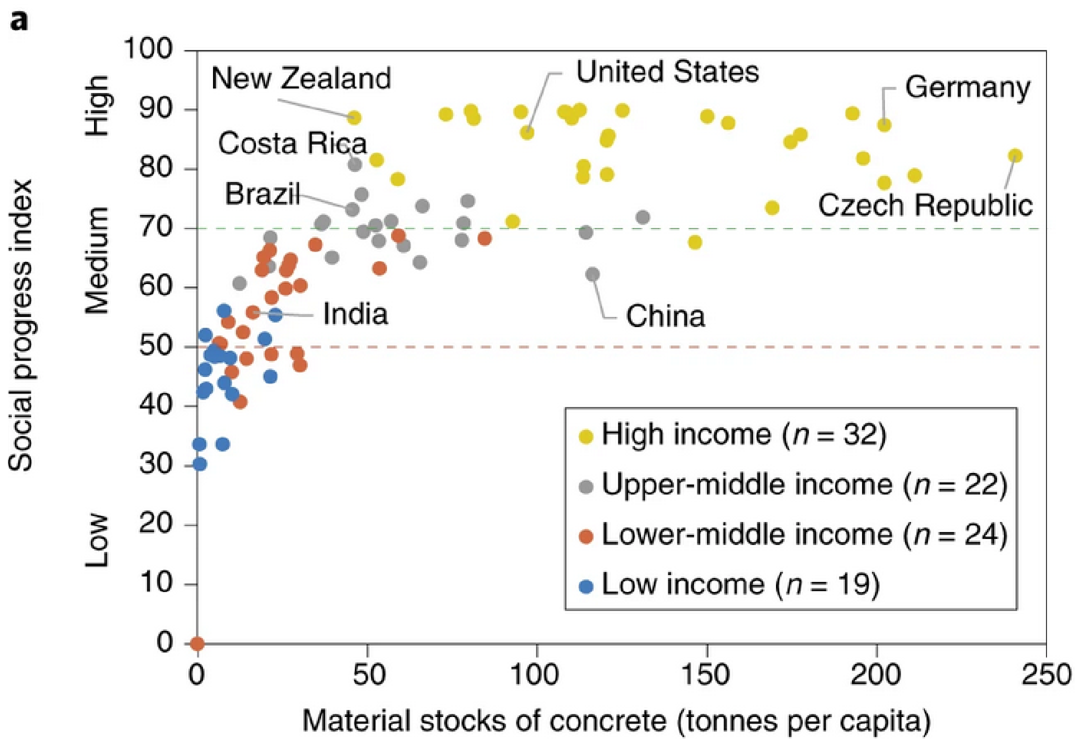

Figure 1. Material stocks and social progress. Concrete stocks versus SPI in 97 countries. Reproduced from Haberl et al. (2019) with permission from Helmut Haberl and Springer Nature.

In Chapter 7 I present novel, interdisciplinary work that is amongst the first to attempt to understand the drivers and systemic solutions to infrastructure expansion from a combined biophysical and political economic perspective, albeit in a single country for a single infrastructure class (housing in England).

1.5 Geographical scope

In this thesis, I investigate a range of biodiversity offsetting systems and policies for addressing the impacts of infrastructure. All my main case studies are situated in the global north (UK, Australia), as is my main case study for understanding the political processes that cause infrastructure expansion. This thesis asks questions which relate to the design of biodiversity offsetting mechanisms and regulatory markets, and how much infrastructure stock is required to satisfy everyone's fundamental needs. The necessary precursor for these questions to be relevant is that the geographical region has advanced environmental governance capabilities, offsetting is one tool within a diverse policy mix for governing society's impacts on biodiversity, and infrastructure stocks are sufficiently expansive to already be satisfying a large proportion of society's infrastructure needs. In the global south, several of the premises behind these questions may not hold. For example, nature conservation may be so underfunded that the revenues generated by offsets might be essential to national biodiversity conservation spending, which leaves questions about trading rules and mechanism design less important than the fundamental premise of generating increased revenues for conservation. In Madagascar, the Ambatovy mine represented the single largest ever investment in the country, which enabled the mine's offset to be sufficiently large to fund and manage an entire new protected area (Devenish et al. 2022). Additionally, there may be fundamental infrastructure deficiencies (i.e. total infrastructure stocks at <50 t/capita), which generally makes further infrastructure expansion necessary to satisfy unmet human needs. My discussion of addressing the processes that cause excessive infrastructure expansion would be mostly irrelevant in such a context. As such, the conclusions in this thesis are broadly more relevant to the global north.

1.6 Thesis structure

This thesis is structured as follows. Chapter 2 reviews both the extent of the global infrastructure boom, and the global distribution of biodiversity compensation policies, ultimately reviewing the policy transformations that must be undertaken for these policies to genuinely achieve no net loss in biodiversity (zu Ermgassen et al. 2019b). Chapter 3 reports the result of a rapid evidence assessment to assemble the peer-reviewed literature reporting empirical outcomes of biodiversity offsets or no net loss policies (zu Ermgassen et al. 2019a). Identifying a serious shortfall in high-quality evidence, in chapter 4 I then conduct an impact evaluation of one of the world's oldest biodiversity offsetting systems in Victoria (Australia) using a quasi-experimental design. Identifying that the outcomes of the system are ambiguous but most likely indicate that the policy did not achieve its no net loss objective, in chapter 5 I then explore an important mechanism that might underpin the limited successes we observe in jurisdictional offsetting policies – contradictions between the way that biodiversity needs to be treated and conceptualised in order to facilitate the application of market-like instruments, and achieving robust ecological outcomes (zu Ermgassen et al. 2020). In chapter 6 I then evaluate the preliminary outcomes of England's new biodiversity compensation system "Biodiversity Net Gain". I identify serious governance gaps and disadvantages of the policy's chosen biodiversity metric which will most likely lead to the policy not delivering its promised biodiversity outcomes. Given that the prevailing policy for mitigating trade-offs between infrastructure and ecology in England is likely to fall short, in chapter 7 I then conduct an interdisciplinary analysis of the housing crisis in England, which quantifies the ecological harms associated with business-as-usual housing policy, and explores the underlying political economy and economics which underpin the fundamentally unsustainable choice of policy for resolving England's housing crisis. I then review policy solutions for satisfying society's fundamental housing needs without transgressing national sustainability policy targets. I conclude in chapter 8, ultimately arguing that postgrowth economics might be a vital discipline for the analysis of problems in conservation science.

Chapter 2 The role of 'No Net Loss' policies in conserving biodiversity threatened by the global infrastructure boom

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2.1 Abstract

Over US$60 trillion is predicted to be spent on new infrastructure globally by 2040. Is it possible to meet UN Sustainable Development Goal 9 (develop infrastructure networks) without sacrificing Goals 14 and 15 (ending biodiversity loss)? We explore the potential role of No Net Loss (NNL) policies in reconciling these SDGs. Assessing country-level overlaps between planned infrastructure expansion, infrastructure-threatened biodiversity, and national biodiversity compensation policies, around half of predicted infrastructure and infrastructure-threatened biodiversity falls within countries with some form of mandatory compensation policy. However, these policies currently have shortcomings, are unlikely to achieve NNL in biodiversity, and could risk doing more harm than good. We summarise policy transformations required for NNL policies to mitigate all infrastructure impacts on biodiversity. To achieve SDGs 9 alongside 14 and 15, capitalising on the global coverage of mandatory compensation policies and rapidly transforming them into robust NNL policies (emphasising impact avoidance) should be an urgent priority.

Keywords: No net loss, biodiversity offsets, infrastructure expansion, environmental impact assessment, Sustainable Development Goals (SDGs), biodiversity compensation, conservation policy

2.2 Biodiversity impacts of the global infrastructure boom

The UN Sustainable Development Goals (SDGs) lay out society's ambition to deliver social and economic prosperity for all, while conserving nature on land and sea (SDGs 14 and 15 respectively). However, 'business-as-usual' approaches to solving social and economic development challenges may compromise our ability to achieve the SDGs that are focused on eliminating our impacts on species and ecosystems (Spaiser et al. 2017; Hickel 2019). One of these potential contradictions relates to infrastructure: is it possible to rapidly expand the world's built infrastructural networks (SDG 9) without harming non-human life on Earth (SDGs 14 and 15)? At this key juncture for the future of biodiversity, the development of the post-2020 framework for the Convention for Biological Diversity (CBD), this is a crucial question to consider.

We are currently experiencing the most rapid expansion of built infrastructure in history ('the basic physical and organizational structures and facilities (e.g. buildings, roads, power supplies) needed for the operation of a society or enterprise'; Lexico Dictionaries) with over US$60 trillion of infrastructure spending predicted between 2019-2040 (estimated for 56 countries totalling 88% of global GDP)(Global Infrastructure Hub 2017, 2018). It is projected that an additional 1.2 million km^2 of land will be urbanised between 2000-2030 (185% increase; Seto et al. 2012), and an additional 3-4.7 million km of roads added to the global network by 2050 (22-34% increase; Meijer et al. 2018). In a high-profile example, the ongoing Chinese 'Belt and Road Initiative' might be the most ambitious infrastructure drive in history (Ascensão et al. 2018). The programme aims to link 65 countries, representing two-thirds of the global population, in a network of transport and energy infrastructure, spatially overlapping with 1,700 sites with conservation designations (Ascensão et al. 2018).

Infrastructural expansion can be an important mechanism for alleviating poverty and delivering economic growth (Agénor & Moreno-Dodson 2006; Donaldson 2018), but when unaccompanied by strong environmental safeguards it is also a key global driver of biodiversity and ecosystem service loss (Laurance et al. 2015; Maxwell et al.

2016). Major extractive, transport and energy-production infrastructure projects are planned within some of the world's most biodiverse and carbon-rich regions, including the Congo Basin, the Amazon and Borneo (Laurance et al. 2015; Latrubesse et al. 2017; Alamgir et al. 2019a). Infrastructure can impact on biodiversity in multiple ways, including direct habitat loss within the built infrastructure footprint, alteration of ecosystem properties or fragmentation (Torres et al. 2016; Tulloch et al. 2019), and exacerbation of biological resource consumption (Laurance et al. 2015) by facilitating further economic activity (through e.g. improving road access). At global scales, one third (9,053/27,159) of all assessed threatened species (categorised as Critically Endangered, Endangered or Vulnerable; assessed 14/6/19) on the Red List are threatened by infrastructure, including around half of all threatened amphibians and birds (55% and 46% respectively; IUCN 2019). Transport, energy, and residential infrastructure are also key contributors to climate breakdown (Laurance et al. 2014; Tong et al. 2019), another important driver of biodiversity loss. In addition to the considerable biodiversity implications, much planned mining, transport and urban infrastructure is also predicted to impact heavily on areas of global ecosystem service importance (Seto et al. 2012; Laurance et al. 2014; Harfoot et al. 2018), further exacerbating major environmental challenges including climate breakdown.

2.3 Regulation of infrastructure impacts on biodiversity

In committing to SDGs 14 and 15, the international community committed to 'sustainably manage and protect marine and coastal ecosystems to avoid significant adverse impacts', and 'protect and prevent the extinction of threatened species by 2020'. Given infrastructure's role in driving biodiversity loss, it is worth asking: how close are we to achieving this aspiration for infrastructure, and what else could be done? This perspective extends the conceptual framework of a 'global mitigation hierarchy' outlined in Arlidge et al. (2018), focusing specifically on mitigating the biodiversity impacts of infrastructural expansion.

NNL policies are an increasingly influential set of policies that have emerged specifically with this aspiration at their core, to fully mitigate the biodiversity impacts of infrastructure and, in some cases, land use change. First rising to prominence in response to widespread wetland losses in the USA and loss of natural landscape aesthetic in Germany (Hough & Robertson 2009; Wende et al. 2018), idealised NNL policies are based on the principle that biodiversity is as a minimum left no worse off after development than before (Box 1). NNL is commonly operationalised through the application of a mitigation hierarchy to development impacts (e.g. avoid, minimise, restore, offset; Bennett et al. 2017) and predicated on a strict preference for the first stage (to avoid biodiversity impacts wherever possible). Most commonly implemented through environmental impact assessment (EIA) frameworks, NNL policies considerably strengthen the treatment of biodiversity in traditional EIA. Traditional EIAs aim to assist with decision-making for developments by providing information on the predicted environmental impacts of development and potentially exploring options for mitigating some of these environmental impacts to 'acceptable' levels, but it is uncommon for EIAs to address impacts on biodiversity per se in quantitative terms (Bigard et al. 2017). In contrast, NNL policies set a clear overall goal for biodiversity, and following the application of the mitigation hierarchy, set out in quantitative terms what actions need to be taken in order for the expected residual losses from the development to be at least matched through compensatory actions including biodiversity offsetting. They explicitly define which aspects of biodiversity are considered priorities and how they are to be measured, and quantitative targets can then be set to assess whether or not these priorities have been achieved (Bull et al. 2013a). Additionally, if ecological theory determines that NNL in biodiversity cannot be achieved in a given context, NNL policies give a concrete rationale to when projects should not be permitted to go ahead (Pilgrim et al. 2013; Phalan et al. 2018). However as explored later, these core principles often fail to be respected in practice, and the quantitative nature of NNL does not free it from the influence of uneven power dynamics or vested interests (Carver & Sullivan 2017). Additionally, one of the main ways that principles of NNL are applied around the world is through the creation of biodiversity compensation policies, which often fall far short of the idealised application of NNL outlined above because of a lack of adherence to the mitigation hierarchy (especially avoidance; Phalan et al. 2018).

Box 1. Key terms
Biodiversity compensation – actions taken to compensate for negative impacts to biodiversity caused by developments, which may include financial compensation for affected stakeholders. Compensatory actions generate gains that are not necessarily quantified, or equivalent in type or magnitude to losses, and as such are more general than 'biodiversity offsetting'. Biodiversity offsetting – actions taken to compensate fully for the residual impacts of development following the quantitative assessment of biodiversity losses; gains must be of equivalent or greater ecological value to losses. Offsetting is a 'specific and rigorously quantified type of compensation measure' (Bull et al. 2016). No Net Loss policy – policy applied at various spatial scales aiming to achieve a minimum of no net loss in biodiversity across all impacts of development. NNL policies are often operationalised in practice through application of the 'mitigation hierarchy'. Mitigation hierarchy – a framework for mitigating biodiversity losses from development by sequentially avoiding biodiversity impacts wherever possible, minimising impacts where impacts are unavoidable, restoring following the impact if impacts are time-bound, and finally offsetting any residual impacts to biodiversity (Gardner et al. 2013).

2.4 Current uptake of biodiversity compensation policies

To assess progress in achieving NNL of biodiversity from new infrastructure, we first explore the global extent of more general biodiversity compensation policies. Whilst much past research on compensation has focused on outcomes at local scales (Lindenmayer et al. 2017; Thorn et al. 2018; zu Ermgassen et al. 2019a), the global implications of compensation policies are only just beginning to emerge. For example, taking just the subset of compensation represented by biodiversity offsets, an estimated $153{,}679^{+25{,}013}_{-64{,}223}$ km^2 of biodiversity offsets were (as of 2018) in the process of being implemented to offset infrastructure and land use change impacts globally, which when summed make the area of biodiversity offsets approximately equivalent in size to a country as large as Bangladesh (Bull & Strange 2018). Recently, the IUCN and collaborators assembled a global database on biodiversity compensation policies, which documents at country-level (covering 197 countries accounting for 98% of global GDP) the degree to which compensation policies (including but not restricted to offsets) are referenced and embedded into overarching national environmental or

EIA legislation (Box 2). This database details that compensation policies including offsetting policies are significantly more widespread than previously reported (Madsen 2011): 37 countries representing 72% of global GDP represented in the database have mandatory compensation policies for at least certain infrastructure sectors or habitat types (Figure 2(A)), with a further 64 countries providing guidance on compensatory measures or enabling offsets as voluntary practice ('precursor policies'). Despite widespread criticism of offsetting policies (Moreno-Mateos et al. 2015; Maron et al. 2018), this global policy adoption indicates that compensation policies could have an important role to play in minimising the biodiversity impacts of the ongoing global infrastructure boom (Quintero & Mathur 2011).

Box 2. The Global Inventory of Biodiversity Offset Policies

The Global Inventory of Biodiversity Offset Policies (GIBOP) is an open-access global database summarising the degree to which biodiversity compensation policies (including offsetting policies) and the mitigation hierarchy are embedded within national environmental policy frameworks. The database was assembled through an analysis of 197 countries' national environmental or EIA legislation, allocating each country a score representing the 'strength' of biodiversity compensation legislation. Whilst this score was allocated using a standardised process across each country, there remains an unavoidable interpretive element. Scores are defined as:

0) no mention of compensation;

1) countries at an early stage of policy development (minimal regulatory provisions on offset or compensation);

2) countries enabling the use of voluntary offsets (scheme acknowledged in regulatory framework);

3) countries requiring mandatory biodiversity compensation in at least some circumstances.

More information about methods and limitations can be found at https://portals.iucn.org/offsetpolicy/.

Worldwide, the dominant infrastructural threats to biodiversity are residential and commercial development, followed by mining and extraction and then other infrastructure types (linear infrastructure and energy production; Figure 2(B)). According to the Global Infrastructure Hub, US$46 trillion of infrastructure investment by 2040 (74% of predicted infrastructure investment for the 56 countries in the database) is predicted to occur in countries with mandatory compensation policies for at least some infrastructure classes or habitat types (Figure 2(C)) (Global Infrastructure Hub 2017, 2018; GIBOP 2019). These countries are associated with an estimated 568,000 km^2 in additional urban areas (2000-2030; 47% of global total (Seto et al. 2012)) and over 1.5 million km of new roads (by 2050; 42% of global total (Meijer et al. 2018)). Consequently, around half of the world's new infrastructure up to 2040 can be expected to fall within countries with some existing form of mandatory compensation policy, and this is likely to increase as adoption of compensatory policies including biodiversity offsetting continues to spread globally. If all countries currently enabling (but not requiring) the use of various forms of biodiversity compensation as part of their impact mitigation strategies (n=64) moved to mandatory policies, this coverage would increase considerably (e.g. an additional 35% of projected global road expansion would fall within these countries).

Beyond being applicable in countries in which around half of the world's projected infrastructure will be constructed, compensation policies also cover a sizeable proportion of the world's biodiversity features threatened by infrastructure. We assessed the spatial overlap between infrastructure-threatened bird species extant ranges (N=593, Red List accessed 14/6/19) and regions under different compensation policy strengths (Box 2), using birds to minimise assessment biases between species (Butchart et al. 2004). The mean percentage of each species' range falling in countries with mandatory compensation policies is 47%, and a further 25% falls under 'precursor' policies (Figure 2(D)). We note here that we are simply describing broad spatial overlaps, and not speculating about causal relationships between biodiversity and compensation policy adoption. Additionally, at the national scale the particular infrastructure impacts threatening these species may not fall under the jurisdiction of current compensation policies (e.g. if the impacts are generated by an industry which is not regulated). Nevertheless, this high-level coverage of threatened biodiversity demonstrates that compensation policies are likely to play a key role at the global scale in the conservation of biodiversity threatened by infrastructure expansion.

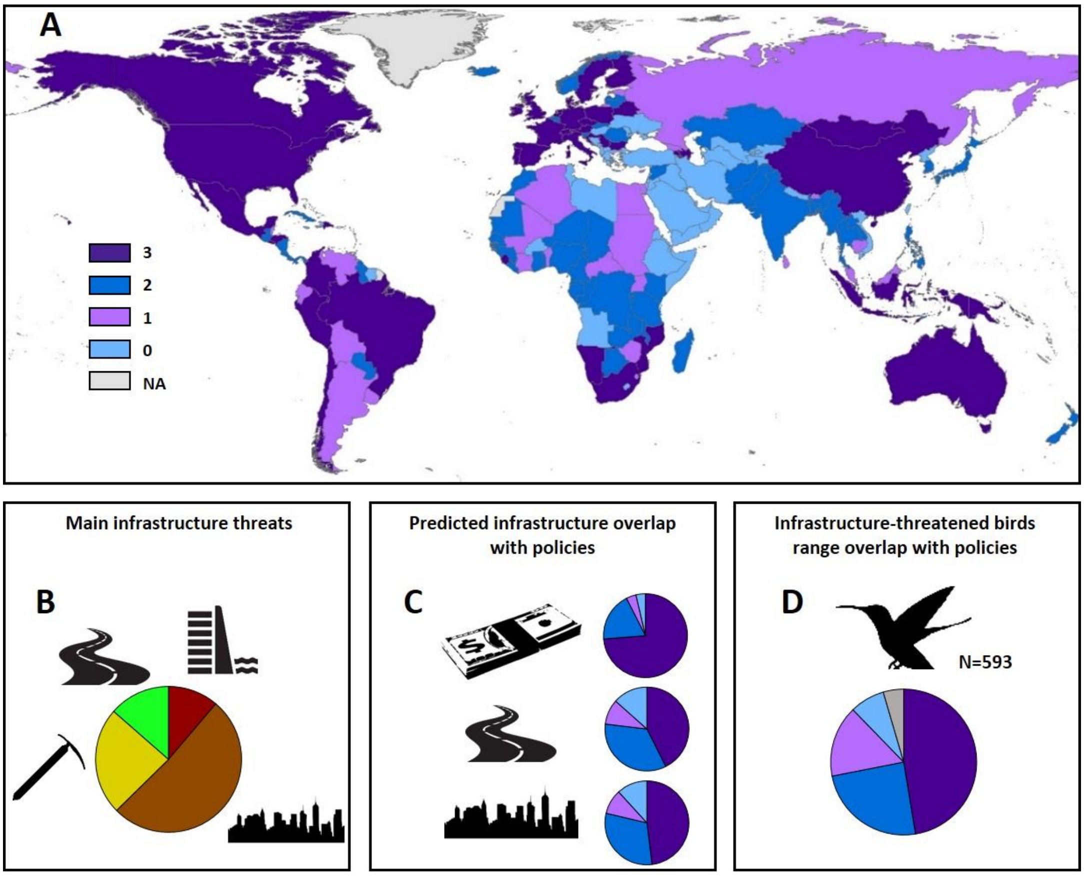

Figure 2. Infrastructure-related threats to species and global coverage of biodiversity compensation policies. Policy scores (see Box 2): 3 = mandatory compensation in some contexts; 2 = enable voluntary offsetting; 1 = minimum regulatory provisions for compensation; 0 = compensation not mentioned in national policy. A) Global map of compensation policy strength (GIBOP 2019). B) Breakdown of the main source of infrastructural threats facing all infrastructure-threatened (CR-VU) species on the IUCN red list (N=9,059 species; pie-chart comprised of 11,475 threats, some species double-counted if facing multiple types of infrastructural threat (IUCN 2019)). Main threats, clockwise from top: dams, residential and commercial development, mining and energy production, transport and transmission networks. C) Overlap between compensation policies (GIBOP 2019) and different indicators of global infrastructural expansion. Top: distribution of predicted infrastructure spending 2019-2040 for 56 countries accounting for 88% of global GDP (Global Infrastructure Hub 2017, 2018). Middle: distribution of predicted road expansion by 2050 for 164 countries (Meijer et al. 2018). Bottom: distribution of predicted urbanisation 2000-2030 for 189 countries (Seto et al. 2012). D) Mean overlap between extant distribution of infrastructure-threatened birds on the Red List (N=596) (BirdLife 2018) and biodiversity compensation policies.

2.5 Moving from biodiversity compensation to No Net Loss

The widespread integration of biodiversity compensation requirements with national policy frameworks around the world demonstrates policy recognition of the impacts of infrastructural expansion. However, biodiversity compensation policies need to be carefully designed in order to stand a chance of achieving NNL consistent with the aspirations of the SDGs (Maron et al. 2018), and current biodiversity compensation policies often fall far short of this aspiration. The GIBOP database shows that only 23% of the countries enabling or requiring (scores 2-3) biodiversity compensation (including offsets) require that compensation be used strictly as a 'last resort' after the rest of the mitigation hierarchy, and of these 101 countries, only 10% apply international best practice principles (BBOP 2013). These shortcomings have several implications. Using offsets or other forms of compensation without sequentially implementing the rest of the mitigation hierarchy risks permitting the loss of irreplaceable biodiversity such as slow-recovering or old-growth ecosystems or threatened species (Pilgrim et al. 2013; Reside et al. 2019). Additionally, it risks facilitating increased damage to natural systems under the logic that offsets might be marginally cheaper than avoidance, trading certain biodiversity losses for uncertain gains (Spash 2015). If NNL is to realise its potential to mitigate the impacts of the global infrastructure boom, an essential first step is therefore to transform existing biodiversity compensation policies into true NNL policies through mandatory application of preceding stages of the mitigation hierarchy, and implementation of offsets in line with social and ecological best practice rather than more general biodiversity compensation (BBOP 2013; Griffiths et al. 2018).

Such an ambition is not unattainable. Best practice NNL policies applying the mitigation hierarchy already exist in 10 countries, and a substantial amount of international infrastructure investment also falls under the scope of NNL policies through safeguards associated with multilateral development financing, such as the International Finance Corporation's Performance Standard 6 (NNL for impacts to Natural Habitat and Net Gain for impacts to Critical Habitat) and World Bank's Environmental and Social Standard 6. Similar requirements apply in the safeguard frameworks of the Asian Development Bank, Intra-American Development Bank and the African Development Bank (Himberg 2015). As an example of the extent of this financing, between 2015 and March 2019, the World Bank committed US$83 billion to built infrastructure development projects, of which 81% was invested in countries without mandatory NNL policies (data from World Bank 2019). Major infrastructure projects funded by the World Bank are required (at least in theory) to meet ecological outcomes which are 'materially consistent' with their own NNL policies (World Bank 2018). In addition to multilateral financing, major private financing sources mandate NNL implicitly under the Equator Principles (a risk management framework for managing socio-environmental risks of project finance, adopted by 97 financial institutions worldwide), which commits them to the International Finance Corporation performance standards including Performance Standard 6 (IFC 2018). Eighty percent of project finance transactions in emerging markets are now associated with banks that have adopted the Equator Principles (IFC 2018), although considerable further reforms are needed to enhance implementation of the principles (Wörsdörfer 2015).

The combination of national compensation policies and multi-lateral policy coverage indicate that enhancing biodiversity compensation policies to aim for NNL could provide a key tool for mitigating the impacts of the global infrastructure boom. But we argue below that if even existing 'best-practice' NNL policies are to fulfil their potential there is need for a rapid, transformational improvement in their application and effectiveness, or they risk undermining biodiversity conservation outcomes overall.

2.6 Expanding the scope of No Net Loss policies

Many NNL policies have historically failed to achieve their intended overarching policy aim (zu Ermgassen et al. 2019a): shortcomings are embedded into multiple stages of the NNL policy implementation process from policy down to project scales (Figure 3). Perhaps the most important limitation to most existing NNL policies is that the total infrastructural impacts under their jurisdiction tend to be highly constrained – often the majority of impacts fall outside the scope of existing regulation (referred to by Maron et al. (2018) as Type 2 impacts; Table 2; Figure 3). If NNL is only applied to a subsection of impacts, then even if project-scale mitigation is achieved the policy will inevitably oversee landscape-scale declines in biodiversity (Maron et al. 2018; zu Ermgassen et al. 2019a). There are two main sources of unmitigated infrastructural impacts: deliberate policy choices that leave particular sets of impacts either entirely unaddressed or granted special exemptions from regulation, and illegal, uncompliant or unreported impacts.

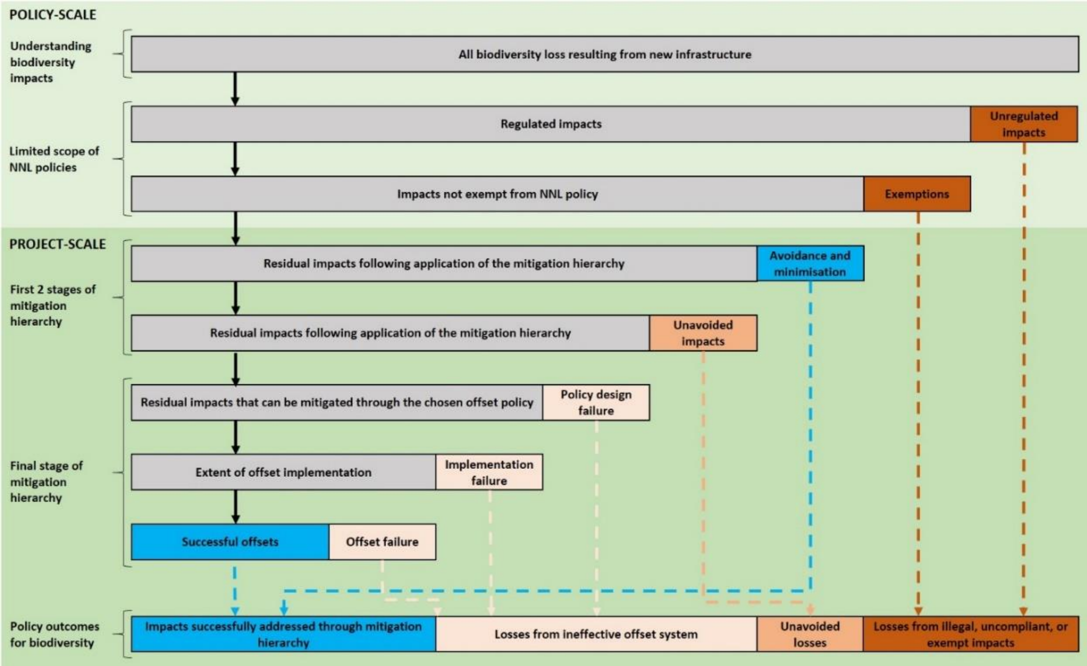

Figure 3. Schematic diagram of the embedded failures to address biodiversity losses from new infrastructure in each implementation stage of the mitigation hierarchy as currently applied in NNL policies. Light green box (top) denotes failures to address the full suite of infrastructure impacts on biodiversity impacts at the policy-scale, darker box (bottom) outlines failures to address biodiversity loss embedded at project-scale applications of the mitigation hierarchy. Type 2 impacts as referred to by Maron et al. (2018) are impacts which do not come under the scope of existing NNL policies, reflected by the 'unregulated impacts' and 'exemptions' categories. The size of the boxes is arbitrary and likely highly context-specific, so we have insufficient information to demonstrate the relative importance of each of the shortcomings in NNL application at this time

Case study	**Policy context**	**Total impacts captured by NNL**
Wetlands in Florida, USA (2001-2011) (Levrel et al. 2017)	National policy goal of no net loss in 'wetland acreage and function' (EPA 2008). Compensatory mitigation allows for compensation for wetland impacts (Hough & Robertson 2009). Mitigation banking is the legislatively favoured and most widely used compensation mechanism	Mitigation banking (which captures most but not all wetland compensation) restored 58,575 ha across the study region, but overall Florida experienced a net loss of over 56,000 ha wetlands across the study period
Wetlands in 20 counties in North	As above	4,591 ha and 68 ha of wetlands were restored and created

Carolina, USA (1994-2001) (Carle 2011)		respectively across the study period, whilst the net loss of wetlands was 25,303 ha
Habitat suitable for threatened endemic the southern black-throated finch (*Poephila cincta cincta*), predominantly in Queensland, Australia (2000-2016) (Reside et al. 2019)	National Environmental Protection and Biodiversity Conservation (EPBC) Act aims to protect 'Matters of National Environmental Significance', which includes threatened species. Where an action might impact on 'Matters of Environmental Significance', a referral to regulators is necessary, and if found to have a significant impact, offsets may be mandated. Simultaneously, Queensland has the Vegetation Management Act (VMA), which aims to maintain biodiversity and ecological processes through regulation of vegetation clearing	631,000 ha of potential black-throated finch habitat (which should have counted as a 'Matter of Environmental Significance' because of the finch's threat status) was cleared across the study period. Of this, 502,391 ha was not associated with a known referral under the EPBC act, despite that the majority was likely cleared for pasture and thus subject to a referral
Native vegetation in New South Wales, Australia (2005-2015) (Gibbons et al. 2018)	Aim of New South Wales Native Vegetation Act is to 'prevent broad-scale clearing unless it improves or maintains environmental outcomes'. Offsetting is one mechanism mandated by the policy	Policy included exemptions that enabled circa 87% of vegetation clearing to occur uncompensated

Table 2. Case study examples of the disparity between total infrastructure or land use change impacts and those impacts which are subject to NNL (indicated in the above cases by the degree of offsetting relative to habitat loss)

All biodiversity impact mitigation policy has limitations to its coverage: mitigation policy commonly applies to either a subsection of biodiversity (i.e. only particular habitat types or legal designations: e.g. Indonesian forest policy requires compensation for losses from deforestation of state forests), or a subsection of industries (e.g. Mongolia requires compensation for damages associated with mining, petroleum and mineral extraction projects). However, as the evidence grows for the biodiversity and ecosystem service value of habitats that have not classically received much protection, such as isolated habitat fragments (Wintle et al. 2019), urban nature (Goddard et al. 2010) and abandoned land (Navarro & Pereira 2012), allowing unmitigated biodiversity loss across any habitats now seems increasingly incompatible with achieving a minimum of NNL of biodiversity at landscape scales (Bull et al. 2020). Additionally, even when regulation should in theory apply, many regions grant exemptions for specific infrastructure developments deemed to be strategically important, reflecting an underlying political prioritisation of economic over biodiversity values. For example, numerous national governments have circumvented the EU Habitats Directive's nominal NNL policy for the Natura 2000 network of protected areas by arguing that the associated infrastructures are in the 'overriding public interest', granting them an exemption even though the justifications for this designation often fall far short of what is legally required (Krämer 2009). Additionally, many impacts are implicitly exempted from policies if they are deemed not to exceed certain impact 'significance' thresholds, which can often be arbitrary or overruled on arbitrary grounds (Jacob et al. 2016; Murray et al. 2018). According to government consultation documents, the proposed approach to mandate Biodiversity Net Gain in England comes close to covering all infrastructure impacts (Defra 2019a). Under the proposals, developments will be required to deliver an improvement in biodiversity (as measured by the UK Department for Environment, Farming and Rural Affairs biodiversity metric; Defra 2018a) consistent with good practice principles (Baker et al. 2019). However, even this policy acknowledges that certain developments are, at this stage, exempt such as 'nationally significant infrastructure' and 'permitted development' (Defra 2019a). These developments will still adhere to existing UK laws to protect biodiversity, but these laws give consent for developments to proceed with biodiversity loss.

The second major reason why biodiversity loss from infrastructure falls outside the jurisdiction of NNL policy is that many impacts are illegal or unreported. For example, in Queensland, Australia the majority of potential black-throated finch habitat cleared between 2000-2016 was not associated with a referral under the Environmental Protection and Biodiversity Conservation Act (a prerequisite to the application of the mitigation hierarchy), implying that landholders were not reporting their land clearing (Reside et al. 2019). In the Brazilian Amazon, approximately 80% of roads are constructed without government approval, and are therefore not subject to environmental regulations (Brandão Jr & Souza Jr 2006). Improving compliance with and enforcement of environmental regulation is a monumental task, which is far from limited to NNL policies (UNEP 2019).

2.7 Project-scale implementation and compliance challenges

Even if all infrastructure impacts were fully captured within NNL policy, biodiversity still falls through multiple cracks in the application of the mitigation hierarchy at project scales, both in the implementation of the avoidance and minimisation steps, and the design and implementation of offsetting policies (Figure 3). One overarching technical issue is the choice of biodiversity metric to use in impact assessment processes: metrics are simplified representations of the complex phenomenon of biodiversity, and so aspects of biodiversity that are not explicitly integrated into the metric risk falling outside the project planning process (reviewed comprehensively elsewhere; Quétier & Lavorel 2011a; Bull et al. 2013a; Bas et al. 2016).

The avoidance step is widely considered the most important, yet understudied, step of the mitigation hierarchy (Hough & Robertson 2009; Phalan et al. 2018). Empirical evidence for the effectiveness of avoidance is severely lacking (but see Pascoe et al. 2019), and empirically challenging because in some systems much avoidance occurs through unobservable informal communications between developers and regulators, and so the final number of development permits accepted or rejected is a misleading proxy for effectiveness (Sinclair 2018). However, it is clear that many infrastructure projects that receive approval and proceed would not pass simple cost-benefit tests if all negative, long-term, direct and indirect social, environmental and maintenance costs were accounted for (Laurance et al. 2015). Furthermore, proper application of the mitigation hierarchy implies that any impacts to irreplaceable biodiversity must be avoided (Pilgrim et al. 2013); yet, some NNL policies continue to facilitate the clearance of threatened species habitat even when it simply cannot be justified on conservation grounds because it is non-offsettable and risks causing local extinction (Reside et al. 2019).

Avoidance fails to be implemented satisfactorily for many reasons (reviewed in Phalan et al. 2018), including capacity shortages in public bodies responsible for assessing alternative options, and political prioritisation of economic development over environmental outcomes that often renders ‘no project’ scenarios politically undesirable and undervalues long-term socio-environmental costs (Clare et al. 2011; Phalan et al. 2018). Compounding this, EIA processes are often implemented too late in the project planning process to exert significant influence over key aspects of project design such as location, as considerable project costs and planning effort have already accrued (Cashmore et al. 2004; Arts et al. 2016). Corruption and uneven power dynamics can also play a role (Carver & Sullivan 2017; Williams & Dupuy 2017). Situations where groups with a vested interest in development proceeding hold undue influence over the mitigation hierarchy process are commonplace in EIAs through which many NNL systems are implemented (Walker et al. 2009). For example, in some countries companies commissioning EIAs from consultants are permitted to withhold payment until the EIA is delivered, thus holding leverage over consultants to incentivise favourable EIA reports that underestimate negative biodiversity impacts and thus the degree of avoidance required (Williams & Dupuy 2017). Application of avoidance can also be suppressed by governments if they perceive strong geopolitical incentives to promote infrastructure development. For example, dam construction in the Brazilian Amazon cannot be reconciled with achieving NNL in biodiversity (Forsberg et al. 2017; Latrubesse et al. 2017; Jones & Bull 2019), however, the government perceives access to hydroelectric energy to be a geopolitical priority that supersedes avoiding impacts to irreplaceable biodiversity (Fearnside 2016; Gerlak et al. 2019).

Once the avoidance and minimisation steps of the mitigation hierarchy have been applied, any residual impacts of infrastructure on biodiversity are then mitigated through offset policy, with any failures to apply the first two stages of the hierarchy adequately manifesting in additional residual impacts. Losses continue to occur under offsetting policies because of poor offset policy design (Maron et al. 2018), failure to implement the required offsets (Bezombes et al. 2019), and finally through failures of the offsetting interventions themselves (zu Ermgassen et al. 2019a). There are multiple design issues that can embed biodiversity losses into NNL policies (reviewed in Maron et al. 2018), for example when unrealistic counterfactuals are used which imply that unfeasibly high rates of loss would have happened in the absence of the policy (Bull et al. 2014a; Maron et al. 2015), when offsets do not provide any additionality (Thorn et al. 2018), or when there is a lack of accounting for time lags between development losses and offset ecological improvements (Gibbons et al. 2018).

However, even NNL policies that adequately address the theoretical ecological requirements for achieving NNL risk suffering from a number of implementation problems that plague many environmental policies and conservation interventions. A key difficulty is that offsets are often very challenging to organise logistically and contractually (Evans 2017). Habitat-based offsets often require the acquisition or conservation management of land that would otherwise not have been contributing to conservation to the same degree. Offsets may be hard to find because landholders are unwilling to restrict their management rights (Vaissière et al. 2018), or because enough suitable land is simply unavailable (e.g. in Sabah, Malaysia (von Hase & Parham 2018) or France (Guillet & Semal 2018)), and instances of land scarcity are likely to increase in the future. This may drive greater emphasis in future on non-site based offsets (e.g. behaviour change interventions to reduce biodiversity loss). Whether site-based or not, offsets have tended to suffer from persistent implementation failures, related to weak compliance or regulatory enforcement, and inconsistencies within interacting governance arrangements (Evans 2017; Bezombes et al. 2019). At global scales, there are considerable gaps between offset policy and implementation: in 60% of countries that have some form of mandatory biodiversity compensation policy there is no documented evidence of a single offset yet being implemented according to the world's most comprehensive global offset database (Figure 4; Bull & Strange 2018). In these countries, ecosystem loss continues to proceed without proper compensation. Lastly, even if conservation interventions are implemented in line with offset obligations, incomplete understanding of restoration ecology or the effectiveness of the implemented offset actions can lead to a failure to achieve NNL in biodiversity or ecosystem function (Lindenmayer et al. 2017; Theis et al. 2019; zu Ermgassen et al. 2019a).

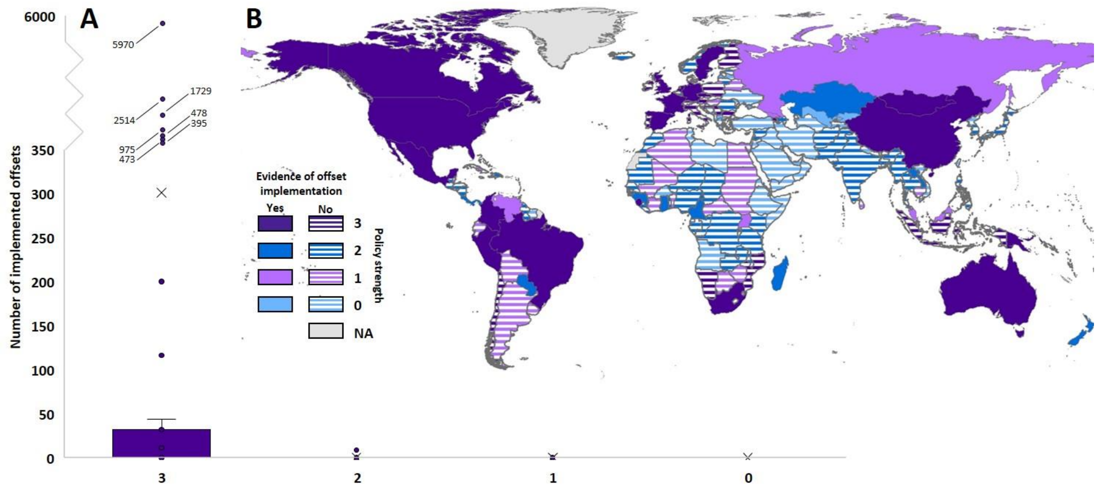

Figure 4. Global disparities between biodiversity compensation policy commitments and offset implementation (Bull & Strange 2018; GIBOP 2019), with the boxplots denoting the total number of offsets recorded as implemented in each country, and the map highlighting countries with no recorded implementation of offsets despite policy commitments. Policy scores (see Box 2): 3 = mandatory compensation in some contexts; 2 = enable voluntary offsetting; 1 = minimum regulatory provisions for compensation; 0 = compensation not mentioned in national policy. A) Box and whisker plots showing upper and lower quartiles and medians of the number of offsets implemented globally under different policy strengths. Crosses denote sample means (adjacent to x-axis for policy strength values 2-0). Whiskers denote the minimum/maximum values that fall within the lower/upper bound of the interquartile range -/+ 1.5 interquartile range. Outliers falls outside that range. B) Map of global biodiversity compensation policies strengths and evidence for offset implementation (defined as the presence of at least 1 offset or a non-zero area of offset implementation in-country from the most comprehensive global offset implementation database (Bull & Strange 2018)). Note that offset implementation displayed may be the result of national policy, voluntary commitments or international financing requirements*

2.8 The future of No Net Loss

Over the last decade, there has been fierce debate about the merits of NNL and biodiversity offsetting and the degree to which it can help achieve or potentially unintentionally undermine conservation outcomes (Moreno-Mateos et al. 2015; Maron et al. 2016; Apostolopoulou & Adams 2017). Empirical explorations of unintended outcomes remain scarce and largely inconclusive so far (e.g. no evidence for 'license to trash' in Levrel et al. (2017) or Gibbons et al. (2018)); nevertheless, there is clearly in some contexts merit to the idea that NNL and offsetting policies have been designed by policymakers and influenced by the private sector to ‘sell’ the narrative that infrastructural expansion and environmental protection can go hand-in-hand (Walker et al. 2009; Calvet et al. 2015), without deep reflection on the considerable barriers to achieving true NNL in practice or the place-based nature of biodiversity and cultural value (Moreno-Mateos et al. 2015; Apostolopoulou & Adams 2017). There are also legitimate concerns that governments may use offset systems as excuses to reduce their own spending on conservation ('cost-shifting'(Maron et al. 2016)); and that offsetting masks the fundamentally political assertion that infrastructure expansion is desirable even in wealthy countries despite that we already risk overshooting on planetary boundaries and that further economic expansion does not necessarily yield wellbeing increases (Apostolopoulou & Adams 2017; Jebb et al. 2018). The social justice of current NNL policies has also been rightfully questioned, with evidence that the most marginalised people tend to be those who bear the largest livelihood costs and see fewest benefits from offset delivery (Bidaud et al. 2018) – for offsets to be ecologically successful and socially defensible, these shortcomings must be addressed through improved legitimate community participation in both infrastructure and offset planning and negotiation processes (Griffiths et al. 2018). These criticisms point to the risk that poorly designed and implemented NNL and offsetting policies could do more harm than good for conservation and people. However, enthusiastic uptake of compensation policies by policymakers does create a large opportunity for conservation globally: if implementation is improved and the benefits of NNL can be maximised, then NNL is potentially an avenue to mitigating damage on natural systems caused by trillions of dollars' worth of infrastructure, in addition to efficiently addressing global gaps in conservation financing through 'polluter-pays' (Calvet et al. 2015). To achieve this potential, the points of failure in each stage of the infrastructural impact mitigation process need to be addressed.

In order to make progress towards achieving NNL at policy scales, the jurisdiction of NNL policies must be expanded across all impacts (converting Type 2 into Type 1 impacts; Maron et al. 2018) and exemptions from NNL requirements eliminated. As a first step, we recommend that countries audit their recent infrastructure impacts, assess what proportion of these came under NNL policy, and identify the main reasons for disparities between total and potentially mitigated impacts. This can help highlight the exact policies and exemptions that facilitate the loss of biodiversity from infrastructure development. The enduring problem of limiting illegal infrastructure and biodiversity impacts is key. This remains an enormous challenge, but emerging technologies allowing for near real-time monitoring of land use change may be an important component of the solution (Finer et al. 2018).

NNL may be intrinsically unfeasible for projects that damage invaluable or irreplaceable biodiversity (Pilgrim et al. 2013). NNL policies thus need to define 'no go' situations, and ensure that these are integrated with, and do not undermine, existing strict protections (although in practice, such protections are often over-ridden where projects are considered economic or political imperatives: e.g. dams in megadiverse tropical forest regions; Jones & Bull 2019). It is necessary to enhance macro-scale avoidance through strengthening Strategic Environmental Assessment, integrating development objectives and systematic conservation planning to clearly highlight where impacts to biodiversity must be avoided, such as in South Africa's planning policy and biodiversity offsetting implementation strategy (Brownlie et al. 2017). Additionally, there are ecosystem-specific constraints on whether policies requiring NNL at project scales can achieve NNL at the landscape level. In biodiverse, spatially-constrained regions undergoing rapid infrastructure growth there may simply be insufficient space for the offsets required (von Hase & Parham 2018). NNL at the landscape level requires habitat restoration to compensate for project damage, so may also be unachievable in ecosystems where restoration is very slow or otherwise unfeasible (Gibbons et al. 2016). In such situations, policies can nevertheless set project compensation requirements so that biodiversity remains above a set threshold at the landscape level (Simmonds et al. 2019; Maron et al. 2020).

At project scales, NNL will only be achieved if the incentives of the actors in the system are aligned. NNL needs to be set as a project deliverable from the start of the project lifecycle and the project designed in ways that make tangible, measurable and meaningful outcomes for both biodiversity and for people (Baker et al. 2019). Governments need to set clear and well-enforced NNL legislation, to ensure that developers seeking to deliver NNL are not undercut by competition. Developers need to be incentivised to achieve NNL by being convinced that positive biodiversity impacts do deliver social license to operate and competitive advantage. Commissioners of new infrastructure must demonstrate that they truly value those biodiversity outcomes.

Unfortunately, in many countries these conditions are not present. Central to the misapplication of NNL policy is the underlying political philosophy that short-term economic and security considerations outweigh long-term environmental ones. It is hard to address this in democracies through improved regulatory procedures or transparency; political philosophies will only shift when underlying cultures – voters and their values – change to demand these alternative priorities. However, good policy can help constrain gross violations by setting clear boundaries that cannot be overstepped without triggering comprehensive public scrutiny. NNL policy can potentially play an important role by clarifying what is and is not acceptable at both the avoidance and offsetting stages. For example, the IFC's guidance note for Performance Standard 6 very clearly states that no financing will be permitted for projects that impact UNESCO World Heritage Sites, or sites fitting the designation criteria of the Alliance for Zero Extinction (IFC 2019). Clear boundaries such as these should help constrain some of the worst potential outcomes of NNL policies if implementation standards still fall short.

There are multiple more specific policy enhancements that could help deliver NNL across infrastructure impacts. To improve implementation of the first step of the mitigation hierarchy, more resources are needed for planners, with an amelioration of power imbalances that distort planning processes. This is politically challenging, but simply providing environmental information consistent with the 'rational decision-making' model is unlikely to deliver adequate avoidance (Cashmore et al. 2004): more systemic changes to planning systems are necessary. These include ensuring that information on biodiversity risks is genuinely provided early enough in the project planning process for 'no-project' to be a seriously considered option; severing the leverage of developers over the assessment of potential impacts (potentially through the establishment of independent public impact assessors; Murray et al. 2018), and improving resourcing for planning departments so that they can cope with their case load in areas of rapid development (Laurance & Arrea 2017). To improve the capacity of planners overseeing NNL systems, a portion of offset financing should be reinvested in strengthening institutional capacity and developing the biodiversity information base (including high-quality baseline biodiversity data), helping improve the effectiveness of biodiversity planning and NNL policies over time.

Finally, there are many ways to improve design of offset systems, so as to mitigate the residual impacts of infrastructure expansion. It is necessary to design policy so that NNL is at least theoretically achievable at programme and landscape, not just project scales (Maron et al. 2018), which requires integrating state-of-the-art understanding of multipliers, time lags, biodiversity metrics, and cumulative impacts (not just cumulative impacts of portfolios of infrastructure projects, but also considering the way that infrastructure might interact with other drivers of biodiversity loss such as climate breakdown; Quétier & Lavorel 2011a; Bull et al. 2014; Sonter et al. 2017; Gibbons et al. 2018). Gaining the acceptance and support of local communities is essential to the success of conservation interventions, and offsetting is no exception: ecological and social outcomes would be considerably improved if offsets ensured that nobody affected by the initial development and paired offset was worse off as a result of the development-offset pairing than in their absence (Griffiths et al. 2018). Using the best available evidence for the success of the implemented offset interventions is also essential to achieving NNL, and resources for supporting local-scale evidence-based restoration initiatives are growing (e.g. Conservation Evidence (www.conservationevidence.com)). Monitoring and evaluation should be central to offset systems, with outcomes fed back into processes for synthesising evidence so that the effectiveness of ecological enhancement and restoration can be improved over time. Additionally, measures must be put into place to address the identified global gap between the policy and implementation of biodiversity offsets (Figure 4). Again, an important solution may well be capacity-building and enhanced powers and independence of regulatory bodies. There are very few recorded examples of developers receiving financial penalties for failing to achieve their biodiversity offset obligations (Hahn & Richards 2013). Thus, a simple step likely to improve compliance would be to increase the powers of regulators to prosecute non-compliance. In the context of other environmental policies this is shown to improve compliance not just within the firms prosecuted but more broadly across polluting industries (Gray & Shimshack 2011).

If expanding the world's infrastructure networks is socially desirable, can it be done in a way that meets SDGs 9, 14 and 15 simultaneously? Not if business-as-usual environmental practices continue during the ongoing expansion of the global infrastructure networks. However, existing biodiversity compensation policies could feasibly be transformed into robust NNL policies to close this gap. Enthusiastic policy uptake globally has created an opportunity to limit further extensive damage to biodiversity, if policy design and implementation can be improved. Transforming the scope and implementation of biodiversity compensation policies (and especially emphasising avoidance of irreversible impacts) should therefore be considered a global policy priority, with potential for integration into the post-2020 framework of the CBD.

Chapter 3 The ecological outcomes of biodiversity offsets under 'no net loss' policies: a global review

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3.1 Abstract

No net loss (NNL) biodiversity policies mandating the application of a mitigation hierarchy (avoid, minimise, remediate, offset) to the ecological impacts of built infrastructure are proliferating globally. However, little is known about their effectiveness at achieving NNL outcomes. We reviewed the English-language peer-reviewed literature (capturing 15,715 articles), and identified 32 reporting observed ecological outcomes from NNL policies, including >300,000 hectares of biodiversity offsets. Approximately one third of NNL policies and individual biodiversity offsets reported achieving NNL, primarily in wetlands, although most studies used widely-criticised area-based outcome measures. The most commonly cited reason for success was applying high offset multipliers (large offset area relative to the impacted area). We identified large gaps between the global implementation of offsets and the evidence for their effectiveness: despite two-thirds of the world's biodiversity offsets being applied in forested ecosystems, we found none out of four studies demonstrated successful NNL outcomes for forested habitats or species. We also found no evidence for NNL achievement using avoided loss offsets (impacts offset by protecting existing habitat elsewhere). Additionally, we summarised regional variability in compliance rates with NNL policies. As global infrastructural expansion accelerates, we must urgently improve the evidence-base around efforts to mitigate development impacts on biodiversity.

Keywords: biodiversity offsets; compliance; conservation outcomes; mitigation hierarchy; multipliers; no net loss; policy effectiveness

3.2 Introduction

We are living in an age of both severe biodiversity declines and unprecedented global expansion of built infrastructure (Laurance et al. 2015; IPBES 2019). Approximately a quarter of all species from red-list assessed groups are threatened with extinction, with the unmitigated impacts of infrastructure a major driver (Maxwell et al. 2016; IPBES 2019). These impacts are expected to intensify over the coming decades, with dramatic increases in our transport networks, urban footprint, and energy production facilities already under way (Steffen et al. 2015). Mitigating these impacts is therefore an urgent global priority. Currently, one of the most widely-used tools for addressing the environmental impacts of infrastructure is No Net Loss (NNL) policies (Bennett et al. 2017), which mandate that a mitigation hierarchy (MH) is applied to sequentially avoid, minimise, remediate, and offset the biodiversity impacts of new developments (Bennett et al. 2017), with some variation amongst policies (e.g. US mitigation sequence: avoid; minimise; compensate).

NNL policies are proliferating around the world (Bennett et al. 2017), reflected in the widespread implementation of biodiversity offsets (Bull & Strange 2018). Throughout, we use the term 'biodiversity offsets' to refer to all offsets implemented as the final stage of NNL policies, as nearly all policies focus on achieving outcomes that are related to or underpinned by biodiversity. However, the exact ecological characteristics for which these policies aim to achieve NNL vary considerably (e.g. US wetland compensatory mitigation protects 'wetland acreage and function' (EPA 2008). There is a notable lack of evidence regarding the actual outcomes of NNL policies because of the relative immaturity of many policies, a lack of data transparency surrounding NNL implementation (Bull et al. 2018), and challenges evaluating largely unobservable outcomes of the MH process (e.g. identifying avoided impacts; Sinclair 2018). Much of the evidence of NNL effectiveness comes from individual offset case studies or simulation studies (e.g. Sonter et al. 2017; Thorn et al. 2018). In the absence of a coherent body of evidence regarding actual outcomes, many theoretical criticisms and defences of NNL have been discussed in the literature. Criticisms revolve around the ecological feasibility of restoration (Maron et al. 2012), choice and definition of biodiversity 'units' (Bull et al. 2013a), perverse incentives to game offset policies through manipulation of counterfactuals (Gordon et al. 2015), ethics of biodiversity trading (Ives & Bekessy 2015), and the weakening of institutions which safeguard the environment (Walker et al. 2009). In response, defences of NNL acknowledge that well-targeted infrastructural expansion can deliver considerable wellbeing benefits, and when applied according to best practice (Bull et al. 2013a; Bennett et al. 2017), NNL can facilitate this without damaging biodiversity overall. Furthermore, NNL buffers impacts on biodiversity that would most likely occur anyway in the absence of NNL policy (von Hase & ten Kate 2017). Additionally, the organisation and financing of offsets may make avoiding impacts initially more favourable to developers (Calvet et al. 2015). However, without an empirically-grounded evidence base, it is unclear which arguments dominate in practice.

Evidence from case studies shows that NNL policies result in both successes and failures (Quigley & Harper 2006a). As with any conservation intervention, developing evidence about the contextual factors that predict NNL success is essential. Additionally, researchers have reviewed and tested the major indicators of biodiversity proposed for use in NNL and evaluated whether they provide useful approximations of biodiversity changes (Bezombes et al. 2018). However, little work has synthesised which indicators are used in the practical implementation of NNL globally.

Several high-profile NNL policies have now been implemented for sufficient timescales for a preliminary understanding of outcomes to emerge (e.g. Gibbons et al. 2018). However, there remains no synthesis of all the information available on the actual observed outcomes of NNL policies from around the world (i.e. whether they have demonstrably achieved NNL of their ecological characteristic of interest). Addressing this, we reviewed the global literature on the outcomes of NNL policies to synthesise literature gaps and coverage, summarise the state of the knowledge on the determinants NNL outcomes, assess the biodiversity metrics used in practice, assess regional compliance with NNL policies, and evaluate the validity of the existing literature. For clarity, our study addresses both the effectiveness of NNL policies (i.e. the application of the MH to development impacts under jurisdiction of a NNL policy) and individual biodiversity offsets (i.e. whether or not offsets achieve NNL in chosen biodiversity indicators at project scales).

3.3 Methods

3.3.1 Review protocol

We conducted a rapid evidence assessment (Khangura et al. 2012) of peer-reviewed literature on NNL outcomes. Our search term (Supporting information) comprised a set of strings linked by Boolean operators describing:

- alternative offset types (e.g. 'environmental')
- 'offset' and commonly-used alternatives (e.g. 'compensat*')
- impact evaluation (e.g. 'outcome*')
- and excluding nuisance terms (refined by identifying unrelated papers in the first 200 hits of our WoS review; e.g. 'gas mitigation')

Performing the same search in Web of Science (WoS) and Scopus databases (final search date 13/3/19), we removed repeats and then reviewed the remaining studies using the 'metagear' package in R (Lajeunesse 2016; R Core Team 2018). We conducted a first assessment of potentially relevant literature by selecting all studies mentioning NNL policies or offsets in their abstracts, then read the full papers to identify whether our inclusion criteria were met. We limited our review to studies published from 2003-2019, to account for the major reforms to the effectiveness of US wetland mitigation policy introduced by the National Wetlands Mitigation Action Plan in December 2002 (Hough & Robertson 2009). We restricted our search to English-language articles from relevant topic categories (Supporting information). Previous research has shown that English captures most literature on offsets tied to international funding requirements, studies from North America and Oceania, and a substantial proportion of European literature, so our findings should be representative of the global literature (Bull et al. 2018; Bull & Strange 2018). Additionally, we searched through all reference lists in papers meeting our inclusion criteria for additional literature.

3.3.2 Data extraction

Papers were included in our database if they reported observed (i.e. not simulated) ex-post ecological or land cover-related outcomes of polices with an explicit NNL-or-better objective for aspects of biodiversity. We limited our search to peer-reviewed publications only (including conference proceedings and book chapters) to attempt to overcome the data quality issues highlighted by other reviews of offset studies which include the grey literature (Theis et al. 2019), but recognise that the majority of NNL implementation occurs outside academic evaluation. Papers reporting evaluations of individual offset projects were included if they specified the impacts (as a minimum defining the impacted habitat and area) associated with the offsets, thus allowing for a rudimentary assessment of biodiversity losses and gains. These papers compared biodiversity at offsets with either biodiversity at the impacted site (pre-initiation of impacts), or with a biodiversity reference site (Table 12 for studies considered but ultimately rejected). Notably, whilst we included only these studies that allowed for a site-specific estimate of biodiversity losses and gains and thus a basic evaluation of whether NNL was achieved, some key NNL policies do not assess biodiversity losses and gains in this way (e.g. US wetland mitigation policy mandates that compensation sites achieve benchmark ecological criteria rather than explicitly achieving the same level of ecosystem functioning as impacted wetlands). Therefore, such NNL policies may in theory achieve full compliance but not NNL.

For each study/individual offset project where possible we extracted information regarding the:

- type of biodiversity outcome variable used to assess losses and gains
- magnitude of the outcome variable at the offset and impact/control site
- affected type of biodiversity (e.g. forest, species)
- location

- mean offset age (mean time between offset initiation and outcome evaluation)
- spatial scale (Table 3)
- whether or not NNL was achieved for the outcome variable of interest
- article author's explanations for why / why not (including only reasons that addressed the specific outcome variable used)

For each reported outcome variable, we assigned it the appropriate level for four descriptive categories (Table 3). If a paper reported multiple ecological indices or outcome variables, we recorded them all. For individual offsets which presented time-series outcomes, we recorded the outcome variable at the latest time-period to allow the maximum time for ecological recovery in the offset-control comparison. For NNL policies presenting time-series outcomes, we took the sum of the outcome variables across time periods to capture the policy's impact across the entire evaluation period. We recorded information about the policy outcomes across its entire geographical jurisdiction (i.e. if a paper reported localised habitat losses but NNL overall (e.g. across an entire state), we recorded that NNL was achieved). We extracted data from figures using WebPlotDigitizer (Rohatgi 2015). We recorded the raw values for outcome variables and used them to infer NNL outcomes, except for papers that compared outcomes between offset and impact/reference sites using statistical tests, where we used the test's outcome to inform NNL designation. When studies reported that outcomes for some of the projects they evaluated was unknown, we recalculated the percentage of projects reporting successes and failures restricting the total sample to only projects for which the outcome was known. For offset project studies that reported per-unit-area values for a given outcome variable, we multiplied the outcome variable for the offset site by the offset ratio so that the final comparison between biodiversity at the impact and offset sites accounted for differences in area between the two. Therefore for project-scale evaluations we did not include area as an outcome variable, but for programme and landscape-scale evaluations habitat area was included as an outcome. Additionally, we noted two important aspects of offset design: whether or not the described offsets referenced the additionality of their associated conservation actions (i.e. whether the biodiversity gains at the offset were additional to what would have been present in the absence of the offset), and whether losses/gains were evaluated against a static or dynamic counterfactual (McKenney & Kiesecker 2010; Bull et al. 2014a).

Category	**Groupings**	**Inclusion criteria**
Scale	Landscape	Assess changes in the total area of a particular land cover type regulated under a regional NNL policy (although note that some individual impacts within the geographical jurisdiction of the policy will not be regulated by the policy because of legal exemptions or illegal impacts)
	Programme	Assess the outcomes of a defined portfolio of offsets without necessarily comparing them with their associated impacts
	Project	Report the results of individual impact/reference and offset pairs
Offset type	Creation	Result in the creation of new habitat where none existed previously
	Restoration	Restoration or enhancement of degraded habitats; may or may not result in additional habitat area
	Protection	Protection of existing habitats, may or may not involve conservation management. No additional area for conservation
Data type	Ecological site-based	Primary data collected on site
	Expert judgement	Judgement about outcomes elicited from experts
	Official documentation	Data retrieved from official documentations such as mitigation permit files or offset registries
	Remote sensing	Use remote sensing to assess changes in habitat extent
Outcome variable type	Community indices	General indices used to describe ecological communities (e.g. species richness; Simpson index). Do not account for species identity
	Community densities	Indices showing the abundance of an aspect of biodiversity per unit area (e.g. g/m^2 fish biomass)
	Habitat area	Area of habitat
	Habitat quality	Quality of habitat (e.g. percentage coverage of vegetation types associated with the offset habitat type)
	Indices of biotic integrity	Indices of biotic integrity (Karr 1981), partially account for changes in species identity

	Regulatory compliance	Degree to which a given compliance criterion has been met (compliance does not necessarily demonstrate the achievement of NNL)
	Species population proxy	Direct monitoring or species proxy monitoring methods targeting a particular individual or set of species (e.g. population abundance; environmental indicators of species activity levels)

Table 3. Categorisation of information from each study evaluating outcomes from biodiversity offsets or NNL policies

We also assessed the internal validity of site-based assessments of individual biodiversity offsets, paying particular attention to potential selection bias and performance bias (Bilotta et al. 2014). We recorded information about the:

- study design (e.g. before-after-control-impact);
- control used (e.g. either impact-site or reference-site);
- sampling methods and whether those descriptions were sufficiently randomised or open to selection bias; and,
- number of time periods sampled and whether this was sufficient to capture inter-temporal ecological dynamics.

3.4 Results

3.4.1 Overview of studies

Our searches returned 15,715 articles once duplicates were removed. After screening abstracts for relevance, we fully assessed 418 articles for inclusion (Table 11). Twenty-nine studies met our inclusion criteria (7% of potentially relevant studies), with a further three identified via in-article citations, leaving 32 studies from 5 countries (Table 4; Figure 5). Our database includes four landscape-scale, 18 programme-scale, and 10 project-scale studies (covering 26 projects) and accounts for a minimum of 300,000ha of offsets and 180,000ha of impacts, representing approximately 2% of the global area of spatially explicit known offset implementation (Bull & Strange 2018). In total we identified 121 outcome variables (column 11, Table 4) from 48 NNL policies or individual offsets (1-44 outcomes per study, mean=3.75). NNL assessments could not be made for eight studies, as the sole ecological outcomes they reported related to whether regulatory compliance standards were met (e.g. percentage invasive species plant cover), which often do not explicitly aim to achieve NNL of biodiversity per se at project scales (Sudol & Ambrose 2002). When treating each offset or NNL policy independently (N=48), NNL was achieved for 17 assessments, not achieved for 15 assessments, and both successful and unsuccessful depending on the choice of outcome variable for eight assessments. No studies demonstrated the achievement of NNL in forested ecosystems or for avoided loss offsets (Figure 6).

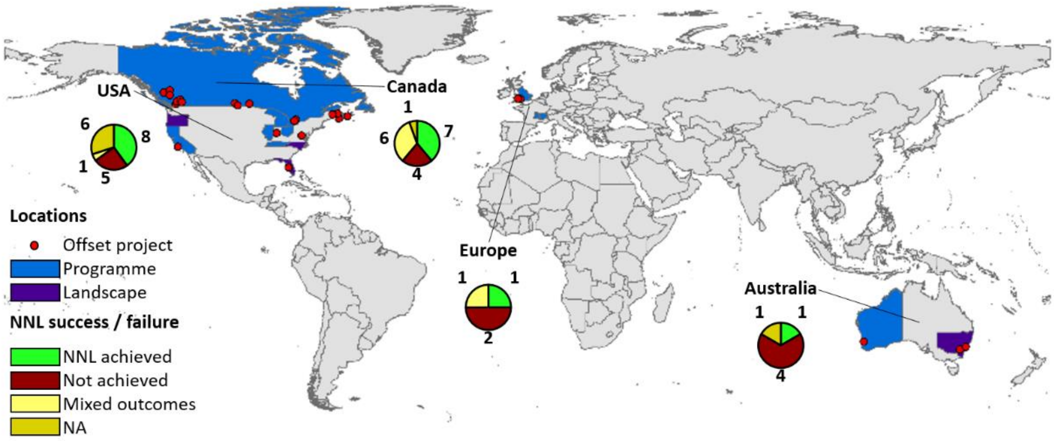

Figure 5. Map of all of study and project areas included in our review. Pie-charts indicate the number of projects/studies by region reporting achieving NNL, failing to achieve NNL, achieving a mixture of outcomes for different outcome variables, and for which no NNL designation could be made because the outcome variable was a measure of regulatory compliance

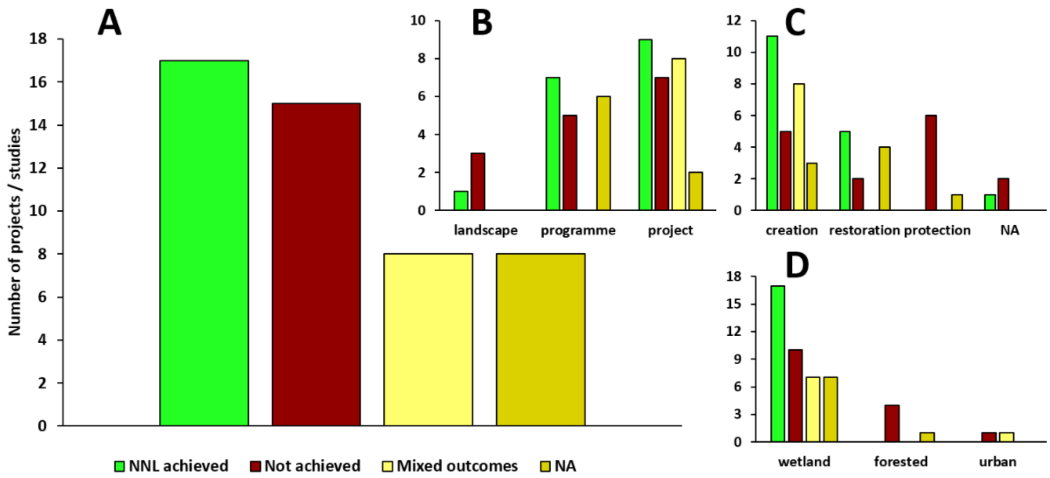

Figure 6. A) Total number of studies/projects within our database achieving NNL. The number of studies/projects is disaggregated by spatial scale (B), offset type (C), and biodiversity type affected (D). NA represents either studies which presented outcome variables from which a NNL designation could not be determined (A), or studies where information on offset type was not provided (C). Studies evaluating the outcomes of bat mitigation actions aiming to achieve NNL in bat population status are categorised as 'urban' (D)

Study	Location	NNL policy	Mean offset age	Scale	Affected habitat / species	Dominant offset type	Total offset area (ha)	Total impact area (ha)	Data type	Outcome variables	NNL achieved?
Gibbons et al. 2018	Australia, New South Wales	New South Wales native vegetation act	10	programme	forested ('native vegetation')	protection	83459	21928	remote sensing	Native vegetation cover	No
Lindenmayer et al. 2017	Australia, New South Wales	Hume Highway environmental assessment	1.5	project	forest species	restoration	NA	NA	ecological site-based	Superb parrot and squirrel glider % nest box / tree hollow occupancy	No
Levrel et al. 2017	USA, Florida	US Clean Water Act section 404	10	landscape	wetland	NA	58575	114575	remote sensing	Wetland area	No
Van den Bosch & Matthews 2017	USA, Illinois	US Clean Water Act section 404	14	programme	wetland	restoration	NA	NA	ecological site-based	% of sites meeting various compliance performance standards	NA
May et al. 2017	Australia, Western Australia	Western Australia state Environmental Protection Act	11	programme	forested	protection	2841	NA	official documentation	% of implemented, evaluated offsets that successfully purchase land for conservation, or achieve their completion criteria	NA
Drielsma et al. 2016	Australia, New South Wales, Lower Murray Darling catchment	New South Wales native vegetation act	7	landscape	forested	protection	107994	40458	remote sensing	Natural vegetation area	No
Goldberg & Reiss 2016	USA, Florida, Lower St John's River Basin	US Clean Water Act section 404	7	programme	wetland	protection	11123	1412	official documentation	Wetland / upland area	No

Fickas et al. 2016	USA, Oregon, Willamette Valley	US Clean Water Act section 404	12	landscape	wetland	NA	NA	NA	remote sensing	Wetland area	Yes
Murata & Feest 2015	UK, Wales, Cardiff Bay	Cardiff Bay Barrage Environmental Statement	7.5	project	wetland	restoration	273	207	ecological site-based	Various ecological indices inc. species richness and Simpson index	Yes
Hobbs & MacAller 2014	USA, California, Santa Maria River	US Clean Water Act section 404	2	project	wetland	restoration	52	52	ecological site-based	Meeting performance standards for native species richness and cover, and non-native cover	NA
Stone et al. 2013	UK, England	EU Habitats Directive; English Nature Bat mitigation guidance	NA	programme	urban species	restoration	NA	NA	official documentation	% change in bat presence/absence and abundances post-development	No
Hill et al. 2013	USA, North Carolina	US Clean Water Act section 404	7	programme	wetland	restoration	8000	NA	ecological site-based	% of wetland and stream components achieving regulatory compliance	NA
Pickett et al. 2013	Australia, New South Wales, Sydney	Sydney Olympic Park development licence	10	project	wetland species	creation	6.4757	0.3351	ecological site-based	Population size	Yes
Kozich & Halvorsen 2012	USA, Michigan, Upper Peninsula	US Clean Water Act section 404	3.5	programme	wetland	creation	75.1	28.8	ecological site-based, official documentation	% of wetlands meeting invasive species performance criteria, wetland area	Yes (area), NA (compliance criteria)

Carle 2011	USA, North Carolina	US Clean Water Act section 404	7	landscape	wetland	NA	NA	NA	remote sensing	Wetland area	No
Reiss et al. 2009	USA, Florida	US Clean Water Act section 404	6	programme	wetland	restoration	24014	NA	expert judgement	% of banks achieved or on course to achieve final regulatory performance criteria	NA
Robertson & Hayden 2008	USA, Illinois, Chicago district	US Clean Water Act section 404	NA	programme	wetland	restoration	1209.12	448	official documentation	Wetland area	Yes
Kettlewell et al. 2008	USA, Ohio, Cuyahoga River watershed	US Clean Water Act section 404	NA	programme	wetland	creation	NA	14.95	ecological site-based	Wetland area	Yes
Breaux et al. 2005	USA, California, San Francisco district	US Clean Water Act section 404	11.5	programme	wetland	restoration	135.2	40	ecological site-based	Wetland area, % of projects meeting all regulatory performance criteria	Yes (area), NA (compliance criteria)
Teels et al. 2004	USA, Virginia, Warrenton	US Clean Water Act section 404	10	project	wetland	creation	7.3	6	ecological site-based	Species richness, index of biotic integrity	No
Thorn et al. 2018	Australia, Western Australia, Beeliar regional park	Western Australia state Environmental Protection Act	NA	project	forest species	protection	523	97.85	ecological site-based	Area of high quality Carnaby's cockatoo and red-tailed black cockatoo habitat, number of Quenda diggings	No
Harper & Quigley 2005	Canada	Fisheries Act, Habitat Policy	3.6	programme	wetland	creation	102.0388	41.9562	official documentation	Fish habitat area	Yes
Quigley & Harper 2006b	Canada	Fisheries Act, Habitat Policy	4.4	programme	wetland	creation	NA	NA	ecological site-based	% compliance with biological	NA

										performance criteria	
Quigley & Harper 2006a	Canada, Manitoba; British Colombia; Nova Scotia; Ontario; New Brunswick	Fisheries Act, Habitat Policy	5	16 projects	wetland	creation	1.6781	1.9606	ecological site-based	Periphyton biomass, invertebrate abundance, coverage of riparian vegetation, fish biomass	Mixed
Morgan & Roberts 2003	USA, Tennessee, NA	US Clean Water Act section 404	NA	programme	wetland	protection	77.7	38	ecological site-based	Wetland area, % of sites meeting all regulatory performance criteria	No (area), NA (compliance criteria)
Hegberg et al. 2010	USA, Indiana, Indianapolis	US Clean Water Act section 404	3	2 projects	wetland	creation	11575 feet	NA	ecological site-based	Various ecological indices inc. index of biotic integrity	No
BenDor et al. 2009	USA, North Carolina	US Clean Water Act section 404	NA	programme	wetland	restoration	NA	NA	official documentation	Wetland area, stream length	Yes
Shafer & Roberts 2008	USA, Florida	US Clean Water Act section 404	21	programme	wetland	creation	NA	NA	ecological site-based	% of sites likely to meet regulatory permit criteria	NA
Shea et al. 2007	USA, Florida, Hillsborough County	US Clean Water Act section 404	1	project	wetland	creation	400	190	ecological site-based	% of samples meeting permit criteria for wetland plant coverage and nuisance species	NA
Garland et al. 2017	UK, Somerset, Bath	EU Habitats Directive; English Nature Bat mitigation guidance	2	project	urban species	creation	NA	NA	ecological site-based	Presence of brown long-eared bat and common pipstrelle roosts	Yes (long-eared bat), No (common pipstrelle)
BenDor et al. 2007	USA, Illinois, Chicago	US Clean Water Act section 404	NA	programme	wetland	restoration	1053.6	617.6	official documentation	Wetland area	Yes

Bezombes et al. 2019	France, Auvergne-Rhône-Alpes, Isère	French law on biodiversity no. 2016-1087	4	programme	wetland, wetland species	protection	182.76	59.79	ecological site-based	% offsets where compensation habitat or species present	No

Table 4. Outline of all studies included in our review

3.4.2 Outcomes of programme- and landscape-scale evaluations

Four studies conducted landscape-scale evaluations of the area of land cover changes under the jurisdiction of NNL policies, with three finding that NNL was not achieved by area (Figure 6). No causal interpretation should be given to these results as other conservation policies may have been implemented simultaneously with NNL policies. Levrel et al. (2017) and Carle (2011) focused on Florida and 20 counties across North Carolina respectively. Both found that total wetland area decreased over their study periods (2001-2011 and 1994-2001), despite considerable restoration efforts attributable to wetland mitigation policy. Drielsma et al. (2016) evaluated the Southern Mallee Guidelines scheme in western New South Wales, Australia. The authors modelled biodiversity change attributable to the scheme, concluding that it broadly achieved the aim of maintaining or improving native vegetation. However, discounting modelled outcomes, the observed outcomes of the scheme were that over 40,000 hectares of vegetated grazing lease were cleared and 'offset' through the protection of other areas, leading to an overall net loss in vegetated habitat area. Lastly, Fickas et al. (2016) found that NNL in wetland area in Willamette Valley (Oregon) was achieved since the formal adoption of the national No Net Loss policy goal and major clarifications to Section 404 of the Clean Water Act in 1990.

Of the 12 programme-scale evaluations in the literature that included outcome variables from which NNL assessments could be made, seven reported achieving NNL (Figure 6). All seven used change in habitat area as outcome variables, and reported results from offset programmes focused predominantly on habitat creation and restoration (Breaux et al. 2005; Harper & Quigley 2005a; BenDor et al. 2007; Kettlewell et al. 2008; Robertson & Hayden 2008; BenDor et al. 2009; Kozich & Halvorsen 2012). The other three studies also using area as their outcome variables that failed to achieve NNL were all reporting results from offset systems based predominantly on avoided loss offsets (Morgan & Roberts 2003; Goldberg & Reiss 2016; Gibbons et al. 2018). The remaining studies evaluated the success of bat mitigation in the UK under the objective of 'NNL in local bat population status', and the percentage of offset sites in Isère, France, where the required offset habitat type or species was present. Here, NNL was not achieved for both bat presence and abundances post-mitigation (categorised as 'urban' in Figure 6; Stone et al. 2013), and offset habitat/species presence varied from 61-73% (Bezombes et al. 2019).

3.4.3 Outcomes of biodiversity offsets

Twenty-six biodiversity offsets from 10 studies were included in our database, of which we could make NNL designations for 24. Of these, nine achieved NNL for all given outcome variables, seven failed to achieve any, and eight achieved NNL for some outcome variables but not for others (Figure 7). There was not enough identifying variation in the data to statistically explore whether specific aspects of offset design, type or ecology predicted the achievement of a higher percentage of total outcome variables. Nevertheless, it is noteworthy that 64% (7/11, Figure 7 (C)) of projects with offset ratios >1 achieved NNL for all of their associated outcome variables compared with 17% for offsets with ratios ≤1 (2/12, Figure 7 (B)).

There was nominally variation between outcome measures when comparing outcome values between offset and impact/reference sites (Figure 8), although an insufficient data volume to explore statistical differences. On average, assessments of habitat quality tended to find that the quality of offset sites was lower than that at impact sites.

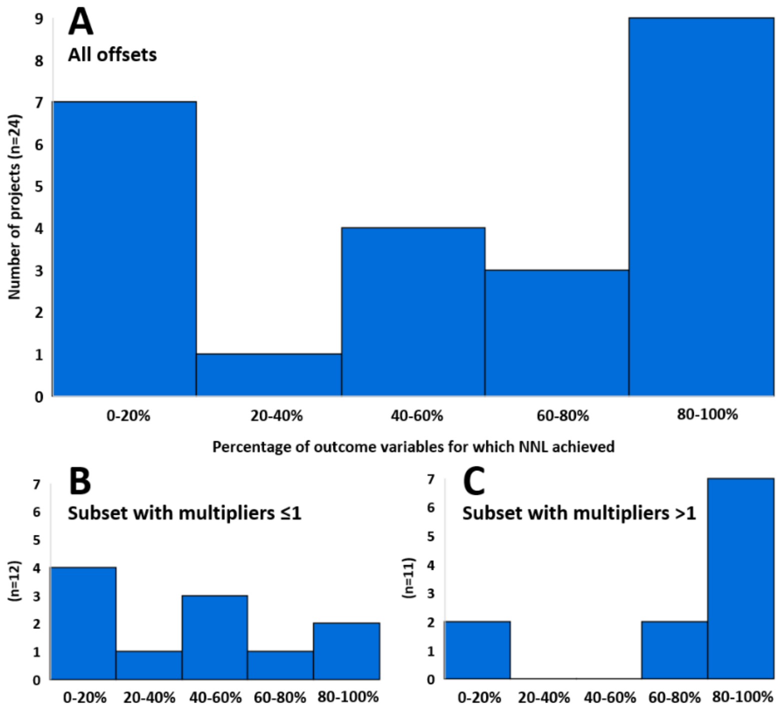

Figure 7. A) Frequency distribution of the percentage of outcome variables achieved for each offset project in our sample where a NNL designation could be made (including one avoided loss offset which is excluded from B and C (Thorn et al. 2018)). B) For all creation/restoration offset projects with a multiplier ≤ 1. C) For all creation/restoration projects with a multiplier > 1.

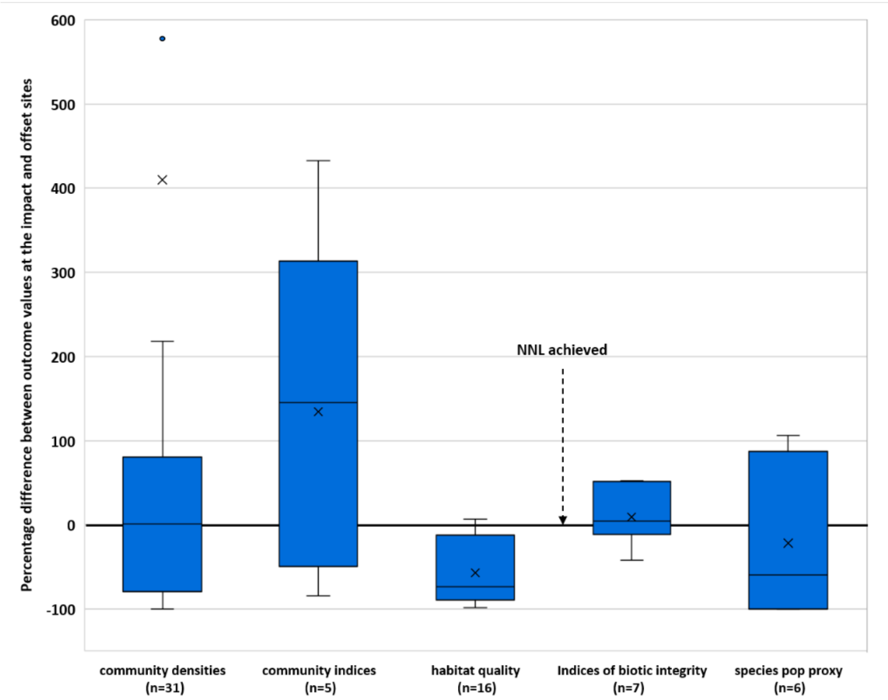

*Figure 8. Box and whisker plots showing the upper and lower quartiles and exclusive medians of the percentage difference between outcome values at offset sites relative to impact/control sites, with outcome variables grouped into categories. Whiskers indicate the maximum/minimum values that fall within +/- 1.5*inter-quartile range. Values >0 indicate that the value at the offset site exceeded that at the impact site. Four outliers (represented by dots) not shown: for the 'community densities' column, outliers occurred at 1469, 3093, 3426 and 4348. Outliers are likely explained by Quigley & Harper (2006a) containing several projects with unusually high offset ratios at several of the sites, and the use of stochastic community density-based outcome measures (e.g. number of invertebrates sampled/m²). Crosses denote the sample mean.*

For the eight project-based studies where offsets were ecologically compared with either their impact sites or reference sites, three met all our criteria for study validity (Teels et al. 2004; Garland et al. 2017; Thorn et al. 2018). Two sampled control/offset sites at a single time-point and thus were unable to account for natural ecological variability in outcomes (Quigley & Harper 2006a; Hegberg et al. 2010, but see justification in Quigley & Harper 2006a), one did not report its sampling protocol and is thus open to sampling bias (Hegberg et al. 2010), and four used controls for their NNL assessments which were collected ≥5 years before data at the offset site (Hegberg et al. 2010; Pickett et al. 2013; Murata & Feest 2015; Lindenmayer et al. 2017).

3.4.4 Outcomes of studies evaluating compliance

Ten studies evaluated the degree to which NNL implementation was meeting regulatory compliance standards at programme scales (Figure 9). Compliance across NNL programmes was imperfect, with no compliance rates exceeding 75% (Hill et al. 2013).

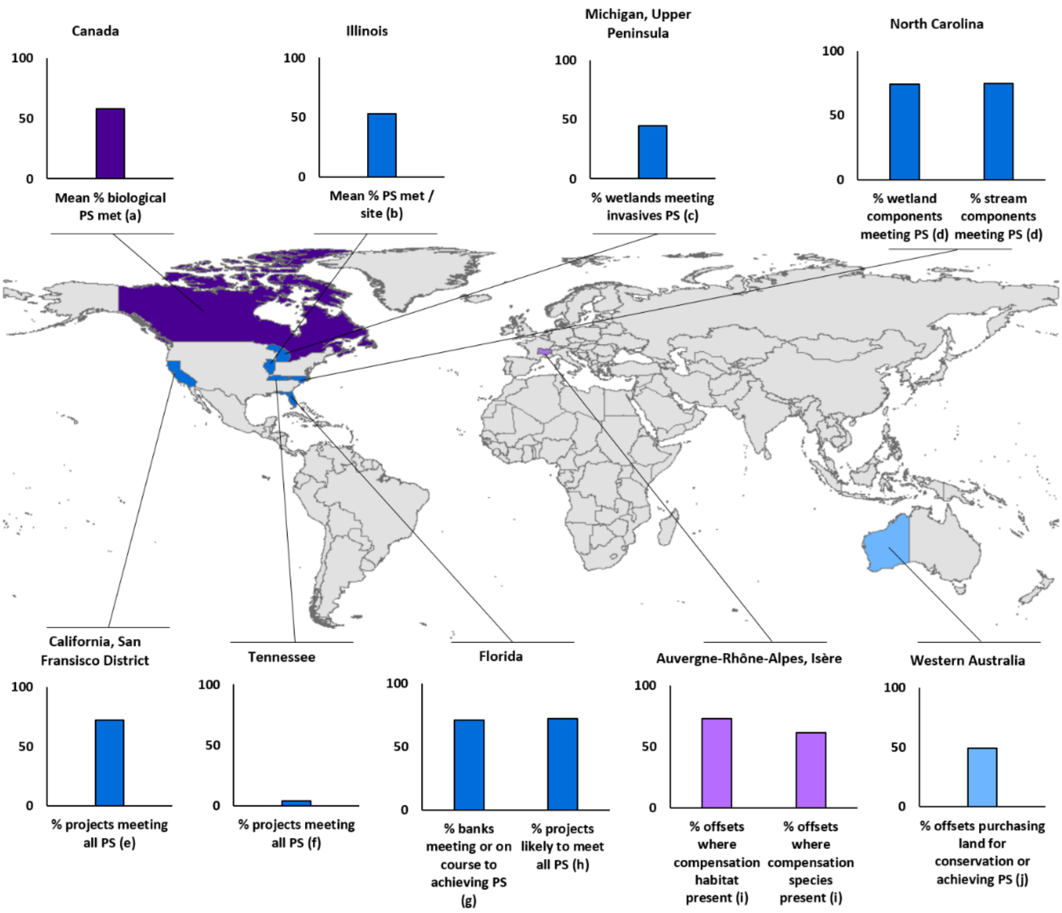

Figure 9. Percentage compliance and compliance criteria reported for regions in our dataset, with bar chart colours corresponding to the region providing the compliance values. Note that the type of reported compliance standards varies between studies, so rates are not comparable. PS denotes 'performance standards'. a) (Quigley & Harper 2006b); b) Van den Bosch & Matthews (2017); c) Kozich & Halvorsen (2012); d) Hill et al. (2013); e) Breaux et al. (2005); f) Morgan & Roberts (2003); g) Reiss et al. (2009); h) Shafer & Roberts (2008); i) Bezombes et al. (2019); j) May et al. (2017)

3.4.5 Reasons for NNL achievement or failure

The two most commonly cited reasons for a lack of NNL success were: failure of the specific conservation interventions applied by the offset (e.g. the offset species failing to respond as expected to the offsetting intervention); and offset implementation failures (Table 5). The most commonly cited reason for success was having high offset ratios. Additionally, Fickas et al. (2016) noted that NNL policy internalised impacts on wetlands that were previously not subject to regulation, thus potentially disincentivising habitat conversion.

NNL outcome	Reason	Scale	References
NNL / offset failure, failure to achieve compliance	Avoided loss leading to an overall loss in area of natural habitats	Programme	Morgan and Roberts 2003; Gibbons et al. 2018
	Compliance standards unrelated to ecological outcomes	Programme	May et al. 2017
	Conflict with development	Programme	Shafer and Roberts 2008
	Conservation intervention failure	Programme; project	Quigley & Harper 2006b; Stone et al. 2013; Lindenmayer et al. 2017; Garland et al. 2017; Bezombes et al. 2019
	Contradictions within permit requirements	Programme	Quigley and Harper 2006b
	Failure to consider landscape context	Programme	Van den Bosch et al. 2017
	Illegal trespassing	Programme	Hill et al. 2013
	Insufficient offset ratios	Programme; project	Goldberg and Reiss 2016; Stone et al. 2013; Quigley and Harper 2006a
	Invasive encroachment without management	Programme	Van den Bosch et al. 2017
	Lack of additionality	Project	Thorn et al. 2018
	Lack of contingency measures in case of offset failure	Programme	May et al. 2017
	Lack of data to demonstrate outcomes	Programme	May et al. 2017; Stone et al. 2013; Bezombes et al. 2019
	Lack of ecological equivalence	Project	Thorn et al. 2018; Teels et al. 2004

	Lack of ecological suitability of creation offset site	Programme	Hill et al. 2013; Kozich and Halvorsen 2012; Quigley and Harper 2006a; Shafer and Roberts 2008
	Lack of monitoring	Programme; project	Quigley and Harper 2006a; Quigley and Harper 2006b
	Lack of offset expertise	Programme	Quigley and Harper 2006b
	Offset implementation failure	Programme	Quigley and Harper 2006b; May et al. 2017; Morgan and Roberts 2003; Shafer and Roberts 2008; Bezombes et al. 2019
	Temporal lag	Programme; project	Quigley and Harper 2006a
	Unregulated impacts	Landscape; programme	Goldberg and Reiss 2016; Carle 2011
NNL / offset success	Bringing impacts under regulation	Landscape	Fickas et al. 2016
	High offset ratio	Programme; project	Pickett et al. 2013; Robertson and Hayden 2008; Harper and Quigley 2005
	Simple biodiversity metric	Project	Pickett et al. 2013

Table 5. List of reasons cited for NNL policy / offset success or failure. The number of citations per reason should not be taken to indicate the importance of that reason, as there was variation between papers in the depth of their discussions of potential explanations

3.5 Discussion

Our review reveals important insights about the state of the evidence-base for NNL and biodiversity offsetting. We provide preliminary indications that: NNL has historically been more successful in wetland than forested ecosystems; avoided loss offsets are particularly risky; evaluations have so far predominantly used area-based outcome measures; there are potential problems with the validity of studies evaluating offset outcomes; and the most common reason for offset success appears to be the implementation of high offset ratios.

We identify a substantial gap between the global implementation of NNL and the evidence base concerning ecological effectiveness. Sixty-seven percent of the world’s offsets are applied in forested ecosystems (Bull & Strange 2018), yet our review reveals that only four studies have assessed NNL outcomes from offsets applied to forest ecosystems or wildlife. Of these, none demonstrated that their associated NNL targets were achieved. Similarly, 20% of the world's offsets entail some form of protection or avoided loss (Bull & Strange 2018). Yet, only six studies have assessed NNL outcomes from this common offset type, and none found that NNL was achieved.

3.5.1 Exploring unsuccessful outcomes of avoided loss and forest offsets

Avoided loss offsets appeared to be unsuccessful for multiple reasons. Critically, they necessarily lead to an immediate net loss in habitat area (Gibbons et al. 2018). This can be justified as a mechanism for preventing biodiversity loss if the background rate of biodiversity loss is sufficiently high. However, in the studies included here and the wider literature, it is evident that assumed rates of background declines are commonly higher than the actual rate, superficially justifying the use of avoided loss offsets when in reality gains only accrue many decades into the future (Gibbons et al. 2018; Reside et al. 2019). This issue is compounded if the 'protection' afforded by offsets does not actually reduce the probability of loss, most commonly when sites which are not under threat of development receive 'protection' (e.g. Thorn et al. 2018). Drielsma et al. (2016) justify the use of avoided loss on the grounds that biodiversity improvements on newly protected sites could offset the losses attributable to the reduction in overall habitat extent. Whether or not these condition gains are achieved in reality is questionable, especially considering the consistent ecological or implementation failure of conservation management interventions associated with offsets in our sample (Lindenmayer et al. 2017; Bezombes et al. 2019).

Many of the same reasons apply to explain the apparent failure of offsets focused on forest biodiversity, although identifying explanations unique to forests is challenging as 4/5 forest offset studies are also avoided loss offsets. Additionally, all forest studies came from Australia, where native vegetation offsets based predominantly on avoided loss have been criticised for facilitating high rates of deforestation and species declines (Reside et al. 2019). Nevertheless, both studies evaluating interventions aiming to offset impacts on forest species found that the interventions failed to deliver ecological equivalence, providing either lower quality or less-utilised habitat than that impacted by development (Lindenmayer et al. 2017; Thorn et al. 2018). On the planning side, May et al. (2017) identify a number of shortcomings hindering Western Australia's native vegetation offset policies from achieving NNL, including a lack of contingency planning in the case of offset failure, insufficient reporting of offset outcomes, offset performance criteria being disconnected from actual ecological outcomes, and poor compliance.

3.5.2 Compliance with NNL policies

May et al.'s (2017) findings are indicative of the rest of the evaluations of compliance in our dataset, with variously defined compliance rates ranging from 4-75%. Imperfect compliance rates per se do not guarantee failure of NNL policies from an ecological perspective, as the effects of compliance failure might be outweighed by offset multipliers (Bull et al. 2017a). However, a recent global review including grey literature demonstrated that compliance with offset permit criteria often considerably exceeds the ecological functional performance of those offsets, indicating that achieving compliance is often insufficient to achieve NNL (Theis et al. 2019). Additionally, low compliance rates do indicate that regulatory enforcement of offset outcomes is often lacking, potentially demonstrating limited institutional interest in the true outcomes of offsetting, thus weakening the probability of NNL outcomes (Walker et al. 2009). There are rarely legal mechanisms for imposing financial penalties for non-compliance (Hahn & Richards 2013). Improving monitoring alone will not guarantee improved outcomes (Kozich & Halvorsen 2012): compliance likely requires strict enforcement, with regulators empowered to impose punishments when permits are violated (Gray & Shimshack 2011). Such pecuniary enforcement measures have been demonstrated in the context of other environmental policies to have direct and indirect benefits, such as both increasing compliance rates within punished firms, and inducing spillovers improving compliance within unpunished firms (Gray & Shimshack 2011). Whilst improving compliance is likely key, if NNL policies fail to use an appropriate reference system (either the pre-impact site or control site) to define the compliance criteria for offsets, then even achieving full compliance may well fail to achieve NNL of biodiversity across the paired impacted and offset sites (Theis et al. 2019).

3.5.3 Achieving No Net Loss: true success or methodological artefact?

Despite little evidence for the effectiveness of some common offset types, a third of all projects or studies in our database reporting achieving NNL. All but one of the successful NNL outcomes occurred for wetland habitats or species, with 50% of wetland projects/studies where a NNL designation could be made achieving NNL. Additionally, all of the successful NNL outcomes occurred for creation or restoration offsets. We speculate that wetland restoration offsets might have higher NNL rates than other offset types in our dataset for two main reasons: firstly, wetlands display higher rates of ecological recovery than many other habitat types (Jones et al. 2018), and this recovery is more likely to reach reference conditions if the impacted wetland was itself degraded (relatively likely in areas undergoing development or construction). Secondly, the two main wetland offsetting policies covered by our dataset are Section 404 of the Clean Water Act in the USA and the Canadian policy of NNL in productive fish habitat. These rank amongst the oldest NNL policies, and both have undergone numerous refinements during their implementation (Hough & Robertson 2009; Rubec & Hanson 2009), thus their effectiveness might exceed that of younger offset policies elsewhere.

An additional key reason for biodiversity offset success appears to be high offset ratios. This finding should be considered in the context of recent literature encouraging practitioners not to simply rely upon high multipliers to solve all offset implementation problems (Bull et al. 2017a). However, within our database, high multipliers appear to be a predictor of NNL success. For individual species-based offsets, this may be because high offset multipliers can be a useful mechanism for increasing habitat availability for the offset species and thus easing density-dependence constraints within the re-establishing population (Pickett et al. 2013). For habitat-based offsets, high multipliers might promote the achievement of NNL if best-practice biodiversity metrics which account for both habitat extent and condition are used (Bezombes et al. 2018), although care must be taken to constrain trades between habitat condition types to avoid trading large extents of biodiversity-poor habitat for small extents of valuable habitat (Carver & Sullivan 2017).

However, it is unclear to what degree these perceived predictors of success (wetlands and high multipliers) reflect true trends, or whether these reflect the choice of outcome variables used to assess NNL. At programme scales, 7 of 10 wetland studies where a NNL designation could be made found that NNL was achieved, but all studies used area as an outcome variable. At landscape scales, 2/3 wetland studies found that NNL was not achieved, and again all used area as their outcome variables. At project scales, 9/21 offsets achieved NNL, yet for seven of these successes the outcome variables were community densities. Six of these successes came from Quigley & Harper (2006a), who calculated whether or not NNL was achieved for community density outcomes whilst accounting for the offset multiplier (i.e. to infer whether the overall abundance of the community group in question was higher for the offset than the impact site). Thus, these successful NNL outcomes are also linked inextricably to offset area. Therefore, with our current dataset we cannot definitively answer the question of whether true NNL in biodiversity is more likely for wetlands than other habitat types, because many of the current metrics used to assess NNL in the literature are confounding offset area (and the offset multiplier) with increases in biodiversity. This is problematic because habitat area alone does not necessarily reflect habitat quality or community composition (Dale & Gerlak 2007), and is thus widely recognised as an unsatisfactory biodiversity metric (Quétier & Lavorel 2011b). Additionally, this review cannot indicate the direction of causality – projects with larger offset ratios might be more likely to be successful, but plausibly larger offset ratios might merely be more strongly embedded into older NNL policies.

3.5.4 Influence of spatial scale on No Net Loss outcomes

The perceived discrepancy in outcomes between landscape-scale and programme-scale evaluations of NNL is likely because programme-scale evaluations only account for registered offsets/impacts: yet unregulated or exempt impacts may well make the difference between achieving NNL or not (Maron et al. 2018). For example, in Florida between 2001-2011, mitigation banking restored 58,575 ha of wetlands, yet across the state a net 5600 ha/year were lost during the same time period (Levrel et al. 2017), which is possibly because the Clean Water Act applies only to 'jurisdictional wetlands', thus many wetland impacts escape regulation. Discrepancies between the apparent success of programme-scale area-based evaluations and landscape-scale ones indicate that NNL policies are likely undermined if some impacts are unreported or otherwise exempt from regulation (Gibbons et al. 2018; Reside et al. 2019). Thus, the scope of impacts falling underneath these policies should be widened to include all impacts and minimise opportunities to avoid NNL legislation.

3.5.5 Outcomes of individual biodiversity offsets

For individual offsets, the outcome variables used were more complex than merely habitat area, and generally adapted to the particular contexts of their associated NNL policies (e.g. Quigley & Harper (2006a) used indicators of habitat productivity (variables representing habitat quality and community densities) to assess whether offsets achieved their policy target of NNL of productive fish habitat). Notably, we found only three studies that attempted to assess whether offset and impact sites were ecologically equivalent at the community level. For offsetting to be demonstrably ecologically equivalent, it should capture aspects of species identity or community composition: two studies accounted for community composition by using indices of biotic integrity (Teels et al. 2004; Hegberg et al. 2010), and one by assessing whether habitat type, quality and structure was similar to that at the impact site (Thorn et al. 2018). Given the strong emphasis in best-practice principles on achieving ecological equivalence (McKenney & Kiesecker 2010; Quétier & Lavorel 2011b), the lack of empirical evaluations demonstrating equivalence is a clear gap.

Additionally, we found a number of methodological issues with offset studies, with 3/8 studies conducting site-based ecological assessments of biodiversity losses and gains meeting our criteria for study internal validity. Alongside opportunities for selection bias and the measurement of biodiversity at a single time-point that does therefore not account for ecological dynamics, the most common issue was the use of controls that are open to potential performance bias (Bilotta et al. 2014). Four studies used controls from ≥5 years before measuring biodiversity at the offset site, which can be justified on the grounds that development projects take years to be implemented, but it cannot be ruled out that other factors influenced changes in biodiversity over this time, thus obscuring the true impact of the NNL policy on biodiversity. Additionally, although not identified in these studies, evaluators should beware pseudoreplication when assessing whether NNL is achieved across multiple sites.

Combined, these points emphasise the need for higher-quality evidence to understand when NNL is defensible as a conservation strategy. Our review identified just one study meeting our inclusion criteria which compared NNL outcomes with a robust counterfactual (Gibbons et al. 2018). Generally, the quality of impact evaluations for NNL appear to be lagging behind those applied in other areas of conservation and environmental policy, such as payments for ecosystem services (Pynegar et al. 2018), protected areas (Miteva et al. 2012), commodity sustainability certification (Carlson et al. 2018), and forest policy (Simmons et al. 2018). Recognising that the true causal impact of conservation policies can be confounded by biases in those receiving conservation treatments, there is an increase in applications of experimental, quasi-experimental and matching methods to improve our causal understanding of policy effectiveness (Ferraro & Hanauer 2014a). The first study of this kind assessing the effectiveness of NNL-related policies focused on species conservation banks (Sonter et al. 2019), but most of these do not have NNL requirements, and to date there remain no NNL evaluations using advanced causal inference. This is therefore a vital area of future research.

Biodiversity offsets receive disproportionate attention compared to the other stages of the MH (Hough & Robertson 2009). However, the effectiveness of NNL is fundamentally reliant on robust implementation of avoidance and minimisation measures (von Hase & ten Kate 2017; Phalan et al. 2018). Our current understanding of the effectiveness of these stages is limited. The major difficulty in evaluating avoidance is that only part of the process of avoidance is observable: permit denials and evaluations of alternative impact sites common to major infrastructure projects. The evidence from these stages would imply that avoidance is weakly applied, as numerous studies have demonstrated low rates of project rejection on environmental grounds and weak justifications for why final project sites were chosen (Clare et al. 2011; Phalan et al. 2018). However, recent work from South Africa has found these observable characteristics to be imperfect reflections of the actual avoidance embedded in the planning process, as many decisions on avoidance happen through informal consultations with regulators in advance of project proposal (Sinclair 2018).

3.5.6 Policy implications

Finally, are the findings of this review generalizable and of policy relevance? Our search language is a limitation, and whilst there is evidence that English captures most of the literature on NNL implementation globally (Bull & Strange 2018), NNL systems in countries like Germany or Brazil may not have been captured in our review. Furthermore, our sampling strategy is biased away from the grey literature. However, the direction of this bias is unclear (Theis et al. 2019) – plausible arguments could be made both for a selection bias towards publishing unsuccessful NNL results in the academic conservation literature, and towards not publishing unsuccessful results in the grey literature because of a fear of criticism for legislators or vested interests. Additionally, although our review was global, the evaluations of actual NNL outcomes identified in our review are biased towards high-income countries with strong institutions. Thus, it is possible that our review may overestimate the probability of achieving NNL outcomes in countries with weaker environmental legislation. However, strong institutions far from guarantee a successful NNL policy – details of NNL design are vitally important (Maron et al. 2018). Therefore, without overstating our findings, we feel there are generalizable recommendations that can be derived from our review:

- policymakers should be aware that without significant improvements to existing policies, NNL policies in forested habitats or utilising avoided loss offsets are unlikely to achieve NNL;
- improving compliance with NNL policies is essential for achieving improved ecological outcomes (which may come from mandating some form of penalty for non-compliance); and,
- it is important to move beyond area-based outcome measures when implementing NNL.

With $60-70 trillion dollars committed to infrastructural expansion by 2030 (Laurance et al. 2015), it is essential that we develop solutions that fully address the unmitigated biodiversity impacts of infrastructural expansion. If we are to achieve NNL of biodiversity, it is an urgent priority to develop the evidence base to understand what works, and when.

Chapter 4 Evaluating the impact of one of the world's oldest biodiversity offsetting systems on native vegetation

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4.1 Abstract

Biodiversity offsetting is an influential policy mechanism for reconciling trade-offs between development and biodiversity loss. However, there is little robust empirical evidence on its effectiveness. We conducted the first quasi-experimental impact evaluation of a jurisdictional offsetting policy (Victoria, Australia). Offsets under Victoria's Native Vegetation Framework (2002-2013) aimed to improve biodiversity through preventing losses and degradation of remnant vegetation, and generating gains in vegetation extent and quality. We categorised offsets into those with near-complete baseline woody vegetation cover ("avoided loss", 2702 ha) and with incomplete cover ("regeneration", 501 ha), and evaluated these offsets' impacts on native vegetation extent from 2008-2018. We used two alternative counterfactual estimation approaches: one using statistical matching on biophysical covariates, and another comparing changes in offsets with changes in sites that became offsets in the future. This latter approach may partially account for self-selection bias, whereby landholders opting into the programme may be less likely to have removed their remnant vegetation anyway. We used means-comparisons to evaluate whether avoided loss offsets prevented native vegetation clearance from 2008-2018, and a difference-in-differences framework to evaluate the impact of regeneration offsets. Whilst the impact of avoided loss offsets was zero compared with both controls, the impact of regeneration offsets varied from increasing woody vegetation extent by ~2.8%/year (i.e. ~138ha from 2008-2018) to zero depending on the choice of control and whether offsets burned by wildfires were excluded from the sample. We cannot conclusively demonstrate whether no net loss was achieved as impacts on vegetation condition could not be included in the analysis and the amount of clearing associated with these offsets is not publicly available. However, in the subsequent three-year period 774 ha was cleared under this offset policy. Our results also offer preliminary evidence that self-selection bias might be undermining the outcomes of biodiversity offsetting regulatory markets.

4.2 Introduction

The last half-century has seen a vast expansion in the material demands of the global economy, driving increasing resource consumption and accelerating infrastructure expansion (Steffen et al. 2015a; Krausmann et al. 2018). The land use change accompanying increasing consumption is the predominant driver of biodiversity and ecosystem service loss (Maxwell et al. 2016; Díaz et al. 2019; Marques et al. 2019; WWF 2020). Policy instruments have emerged to attempt to govern potential trade-offs between land use change and biodiversity (zu Ermgassen et al. 2019b; Bull et al. 2020). Among the most influential is biodiversity offsetting, which is being applied in a growing number of jurisdictions globally, as well as under major multilateral bank biodiversity safeguard policies (Bull & Strange 2018; zu Ermgassen et al. 2019b). Offsetting has also received much attention in national and international policy discussions for its perceived promise as a scalable mechanism for attracting private finance into addressing global shortfalls in biodiversity funding (Deutz et al. 2020).

4.2.1 The evidence underpinning biodiversity offsetting

Robust impact evaluation methods are increasingly being applied in other domains of conservation science, such as protected areas (Geldmann et al. 2019), forest policy (Simmons et al. 2018), and commodity certification (Santika et al. 2019). Experimental (e.g. randomised-controlled trials) or quasi-experimental designs (e.g. analysis following statistical matching) improve evaluations of impacts compared with traditional between-group comparisons because they generate more credible counterfactuals against which to assess the 'true' impact of policy (Maron et al. 2013). So far, just two studies have used quasi-experimental approaches to evaluate the outcomes of biodiversity compensation policies. Sonter et al. (2019) used statistical matching coupled with a means comparison to evaluate the effect of Californian species conservation banks on land use change, and showed these banks perversely averted considerable gains in natural habitats relative to counterfactual sites (although note that species conservation banks rarely have an explicit NNL objective and therefore are not true offsets; Gamarra & Toombs 2017). Most recently, Devenish et al. (2022) evaluated the impact of the Ambatovy mine's offsets in Madagascar using matching and fixed effects regression, and show that the associated offset is on track to successfully achieve NNL of forest because it likely will prevent as much deforestation as the mine caused.

However, this preliminary promising result of Devenish et al. (2022) cannot be generalised to jurisdictional offsetting systems (offsetting systems embedded in national or regional policy, often associated with regulatory markets). While the Ambatovy offset was a well-resourced, high-quality and bespoke offset, jurisdictional offsetting policies function differently from direct voluntary offsets (Koh et al. 2019). They are typically implemented as part of the planning process, and as such the policies are designed to satisfy multiple, sometimes conflicting objectives, such as streamlining planning processes whilst simultaneously achieving biodiversity outcomes (Evans 2017; Damiens et al. 2020; zu Ermgassen et al. 2021b). These and other governance challenges have led to, or risk leading to, systemic implementation failures or widespread data unavailability in multiple jurisdictions (Quétier et al. 2014a; Evans 2017; Bull et al. 2018; Bezombes et al. 2019; Samuel 2020; zu Ermgassen et al. 2021b).

Even if offsets are implemented, one major yet underexplored challenge to jurisdictional offset systems is self-selection bias (Jack & Jayachandran 2019). Offsets can only be additional and therefore legitimate compensation for biodiversity losses elsewhere if they induce conservation actions that result in gains that would not have happened in the absence of the offset transaction (Maron et al. 2013). Central to the market-like logic behind offsets, is that sellers would not have implemented conservation in the absence of their offsetting payment. However, previous qualitative work with offset-adopters demonstrates that there are various motivations for implementing offsets on private land, many of which are non-financial and tied to landowners values and attitudes towards nature (Selinske et al. 2016, 2022; Brown et al. 2021; Groce & Cook 2022). This indicates there is a risk of self-selection bias in offsetting systems, with programmes enrolling landholders who might already have been implementing nature-friendly management practices or unlikely to clear existing native vegetation on their land, thereby undermining the additionality of receiving offsetting payments and the achievement of NNL of biodiversity. This is a well-recognised issue in conservation incentive schemes (e.g. PES) (Jack & Jayachandran 2019).

Possibly the most robust evaluation of a jurisdictional offsetting policy to date is Gibbons et al. (2018). They evaluated the outcomes of the New South Wales (Australia) offsetting system, which is predominantly based on 'avoided loss' (i.e. an offset system that compensates for biodiversity losses by reducing the threats to existing habitats so future losses are avoided). They estimated that it will achieve NNL 146 years into the future, as this is the duration required for existing offsets to avoid as much deforestation as that caused by the associated developments. The estimated counterfactual deforestation rate (i.e. what would have happened in the absence of the intervention) is based on the long-term deforestation rate for the region, which leaves opportunities to improve on the methodology by using quasi-experimental methods to generate more context-specific counterfactuals.

Impact evaluations require the estimation of a counterfactual against which to measure the impacts of the intervention. In practice there can be multiple justifiable counterfactuals for a given conservation intervention (Bull et al. 2014b, 2021; Desbureaux 2021) with widely differing effects on policy outcomes (Sonter et al. 2017). Statistical matching is one increasingly-used method for identifying appropriate controls, based on minimising the differences in covariates known to be predictors of the outcome and treatment assignment (Stuart 2010; Schleicher et al. 2019). In conservation, statistical matching is often implemented based on a set of biophysical and economic covariates known to be related to biodiversity outcomes, such as topography or access to roads. However, recent work has identified that a key predictor of the implementation of conservation actions on private land is the psychosocial characteristics of the landowners themselves (Archibald 2020; Brown et al. 2021; Simmons et al. 2021), and data on these are often absent and prohibitively costly to collect. Failing to capture them might lead to biased results. One approach to overcome this limitation is to compare outcomes within the sample of treated units, based on the assumption that landholders participating in the program are likely to be better controls for other landowners participating in the program than landholders which are well-matched on biophysical covariates but do not participate (Tabor et al. 2017).

4.2.2 Victoria's native vegetation framework

Australia has lost one third of all its native vegetation since European settlement. Victoria introduced the Native Vegetation Framework (hereafter 'the Framework') in 2002 with an overall goal of 'A reversal, across the entire landscape, of the long-term decline in the extent and quality of native vegetation, leading to a Net Gain' (Department of Sustainability and Environment 2002). One of the environmental measures introduced by the Framework was native vegetation offsetting, whereby clearing of native vegetation was to be compensated through conservation actions aimed at improving the extent and/or condition of native vegetation elsewhere (see Appendix). Over the following years, the government created a regulatory market in offsets whereby land clearers could purchase credits to offset their native vegetation liabilities. The first offsets implemented under the Framework entered the system in 2006. Offset agreements last 10 years with sites then theoretically protected in perpetuity (but see Damiens et al. 2021). The policy goal and gain scoring methods were altered in 2013 following regulatory reform.

We evaluate the impacts of 'completed' offsets from the native vegetation offsetting system in Victoria, Australia, one of the oldest jurisdictional offsetting policies in the world. We use a robust counterfactual-based design to evaluate whether, and to what extent, Victoria's first tranche of offsets under the Framework resulted in improvements in the extent of native vegetation relative to control sites.

4.3 Methodology

Under the Framework, offsets were considered to generate four types of biodiversity gain (the sum of which can be sold as credits): 'prior management gain', 'security gain', 'maintenance gain' and 'improvement gain', (DSE 2006; Table 6). The currency used to quantify losses and gains is 'habitat hectares' (Parkes et al. 2003): a composite indicator combining habitat area with condition. Condition is measured by comparing the value of a range of ecological attributes with those of intact reference sites for the same habitat type (see Appendix). If offsets are effective, these gains collectively should mean smaller reductions in woodland cover and condition and greater increases in woodland extent or quality, than would otherwise have occurred (although see critiques of the habitat hectares approach; McCarthy et al. 2004).

Gain category	Explanation	Avoid condition losses	Generate condition gains	Avoid losses in area of vegetation	Increase vegetation area
Prior Management	Landholders are awarded units as incentive to participate in the scheme				
Security	Landholders implement legal mechanisms to protect native vegetation from anthropogenic conversion (e.g. enter into management agreement)	✓		✓	
Maintenance	Landholders implement management measures to maintain the current condition of native vegetation over time (e.g. invasive plant removal, stock control)	✓		✓	
Improvement	Landholders implement management measures to improve the condition or extent of native vegetation (e.g. active planting)		✓		✓

Table 6. Summary of the different categories of biodiversity gains achievable according to the Native Vegetation Framework (DSE 2006) and their impact on observable outcomes

The policy goal of offsetting under the Framework was to achieve NNL in habitat hectares of native vegetation despite some permitted native vegetation clearance. We could not conclusively evaluate whether this goal was achieved, as there are no publicly-available data on the clearance events associated with each of the native vegetation offsets in our dataset. Additionally, vegetation condition as measured using habitat hectares is based on site-based attributes that can only be assessed through site-visits (e.g. ground flora, dead wood) and cannot be effectively captured via satellite data such as NDVI measures without comprehensive model testing and validation against site-based data (which is rarely publicly available). However, we could evaluate whether and how much a key component of habitat hectares was generated at offset sites through prevented losses and increases in the extent of woody native vegetation. Gains additional to those achieved at otherwise-similar sites would indicate at least partial compensation for losses elsewhere. We used satellite data on vegetation cover to estimate additional gains in native vegetation extent occurring at offset sites between 2006-2018.

We used two alternative approaches to estimate the counterfactuals. For one approach to estimating the counterfactuals we compared native vegetation outcomes in offset sites registered between 2006-2008 (hereafter "early offsets") with non-offset land parcels not used as offsets ("non-adopters") that were matched on biophysical and land cover variables (Schleicher et al. 2019). We refer to this as our "matched" set of controls. For the second approach to estimating the counterfactuals we compared native vegetation outcomes observed on these early offsets with those on offset sites registered at the end of our evaluation's time series (2017-2019; hereafter "future offsets"). This set of controls therefore comprises land parcels that were not matched on biophysical covariates, but where landowner psychosocial characteristics were more likely to be similar (Simmons et al. 2021).

4.3.1 Data preparation

We obtained shapefiles of all offsets registered on the Native Vegetation Offsets Register from 2002-2019 from the Victorian Department for Environment, Land, Water and Planning (DELWP). The database captures 398 offsets for the years included in our analysis (2006-2008, 2017-2019) covering 5,377 ha. There could be multiple offsets within the same overall landholding, as areas of different native vegetation classes may require different offset management regimes and separate contracts. To match offsets with land parcels not under offset management, we used state-wide land use maps for 2006 (coincident with the system's first registered offsets) which included the spatial boundary and land use information for every land parcel in the state of Victoria (DELWP 2022).

Our woody vegetation cover outcome dataset was derived from the National Forest and Sparse Woody Vegetation Data produced by the Australian government, a Landsat-derived raster with $25m^2$ spatial resolution (Department of the Environment and Energy 2019). Cells were classified into three categories: no woody vegetation, sparse woody vegetation (5-19% canopy cover) and complete woody vegetation (>20% canopy cover, vegetation >2m tall) for an annual time series from 1998-2018. Positional accuracy was estimated at 10m. Whilst offsets in Victoria span various habitat types including grasslands, we restricted our analysis only to offsets containing ecological vegetation classes which, when in good condition, would be expected to be classified as complete woody vegetation cover (i.e. good condition examples of this ecological vegetation class are >2m tall and with >20% canopy cover in each pixel; Supporting Information). Our outcome variable was the proportion of the total number of pixels in each offset/land parcel classified as complete woody vegetation cover in each year, calculated in QGIS (version 3.20.3). We excluded offsets smaller than 10 pixels ($6250m^2$; n=15 early adopters, n=105 late adopters) as the proportion of vegetation cover was sensitive to small changes in these parcels.

For our statistical matching we used a suite of geographical predictors both theoretically and empirically linked with forest loss/gain in multiple contexts (Table 15; Eklund et al. 2016; Simmons et al. 2018; Geldmann et al. 2019; Sonter et al. 2019; Negret et al. 2020). Our predictors of agricultural opportunity costs and ecological productivity included mean rainfall, slope, elevation, temperature, soil water, soil carbon, and baseline woody vegetation cover; predictors of human pressure and accessibility include remoteness, distance from roads and distance from the nearest protected area. We included the land use of each land parcel in 2006, and tested for spatial autocorrelation using a Mantel test to determine whether to include X and Y coordinates of the parcel centroid to address spatial autocorrelation. Given the offsets predominantly mapped onto agriculture, forestry, and conservation area land uses in the state-wide land use dataset, we restricted our potential matched controls to landholdings from these three land use types. We also collated data on all bushfires detected from 2008-2018 from Ward et al. (2019). The fire data was used to test whether the evaluation results are explained by differences in burning between offsets and controls. Information about data sources is in the Appendix.

4.3.2 Analytical approach

The distribution of our outcome variable (proportion woody vegetation cover) at baseline across our offset sites was skewed: many offset sites had proportion woodland cover at or approaching 1 (i.e., the upper bound of our outcome variable) in our evaluation's baseline year (2008; Figure 22). We subset the data into two main categories of offsets—offsets focusing predominantly on avoiding losses of native vegetation and condition that had a proportion baseline woody vegetation cover greater than 0.95 (henceforth **"avoided loss"** offsets, N early=142, N late=81); and those predominantly aiming to achieve increases in native vegetation cover and condition, whose proportion woody vegetation cover in the baseline year 2008 was less than 0.95 (henceforth **"regeneration"** offsets, N early offsets=54, N late offsets=121). We chose 0.95 as the threshold for our core analysis as it retained a sufficient sample size for the statistical analysis of both pools of offsets (i.e., lower thresholds substantially reduced the sample size for regeneration offsets, e.g. threshold 0.9, N early offsets=37). We varied this assumption and evaluated the impact on our results as a robustness check. We analysed both sets of offsets separately. This analytical approach matched important features of the policy: avoided loss offsets act through different mechanisms and often different management regimes from offsets targeting improvements in woody vegetation cover (Table 6), and including both within the same regression framework would constrain our ability to evaluate the effectiveness of the various mechanisms and management measures underpinning the different offset types.

For both of these sets of offsets, we then compared outcomes with those observed in land parcels selected using our two alternative approaches to estimating the counterfactual (i.e. comparing early offsets with matched non-adopters, and future offsets; Figure 10). The advantages and disadvantages of the two approaches to estimating the counterfactual are given in Table 7.

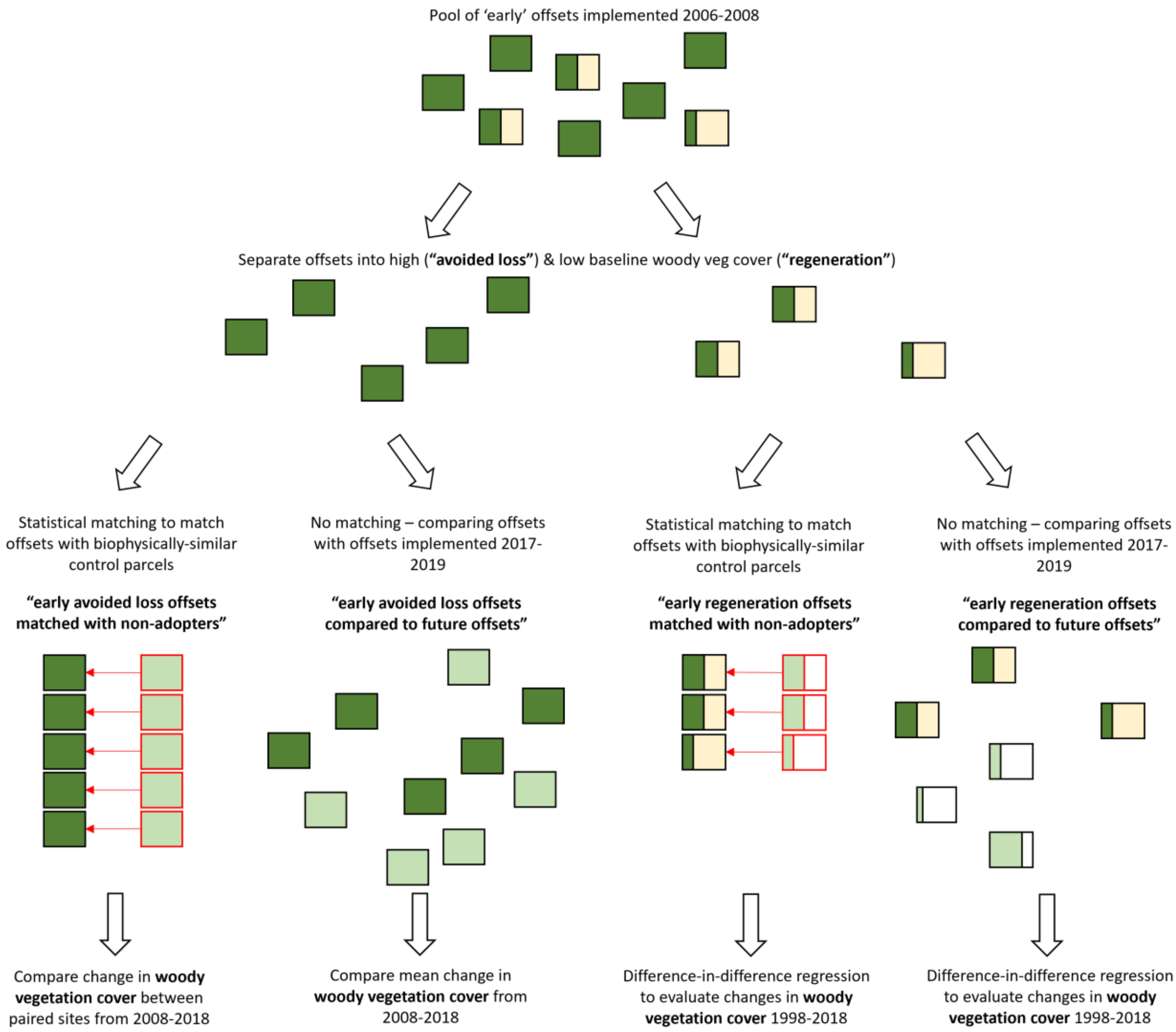

Figure 10. An overview of our methodological approach

	Matching with non-adopters	**Comparing with future offsets**
Advantages	Control parcels were selected to be as similar as possible to offsets according to numerous biophysical and land cover covariates known to affect changes in woody vegetation cover.	Control land parcels were likely to be managed by 'offset-adopters', who are more likely to possess similar psychosocial characteristics to landholders in the offsets sample. Approach therefore potentially reduces self-selection bias. There is evidence of self-selection bias documented by qualitative studies into the

		drivers of conservation on private land in Australia (Selinske et al. 2016, 2022).
Disadvantages	No data were available on the owners of land parcels so may have different environmental attitudes between offset and control samples. This risk of self-selection bias could lead to an overestimate of the treatment effect.	Compared to out of sample matching, the control parcels were less similar to the offsets with regards to biophysical covariates, so multiple factors could be explaining differences in outcomes between offsets and counterfactuals. Land ownership and economic incentives may have changed during the time series, and early-adopters will not be a perfect psychosocial match for late-adopters, so self-selection bias is only partly accounted for.

Table 7. The advantages and disadvantages of both of our approaches to estimating the counterfactuals used in this study

4.3.3 Statistical matching

We used statistical matching to generate one of our two samples of control land parcels (Stuart 2010; Schleicher et al. 2019). We ran the matching separately for both our regeneration and avoided loss subsets. Our pool of potential control parcels was every land parcel in the state within the area range and land use categories of our offsets (N=364,290). We implemented matching in R using the MatchIt package (Ho et al. 2011). Following the protocol of Schleicher et al. (2019), we ran alternative matching specifications (Mahalanobis distance matching, propensity score matching, varying calipers) and inspected the results for evidence of differences in performance using Love plots (Supporting information).

4.3.4 Statistical analyses

Evaluating differences in woody vegetation cover between avoided loss offsets and controls

We compared changes in woody vegetation cover from 2008-2018 in the early avoided loss offsets with that in our two sets of control land parcel samples (i.e. future offsets, and matched non-adopters). Visual inspection of avoided loss offsets and both sets of controls revealed that the overwhelming majority of sites, both offsets and controls, lost no woodland cover in any of the years along our time series (2008-2018). We ran exploratory linear regressions evaluating the change in woody vegetation cover from 2008-2018 across offsets and controls using the economic and biophysical covariates used in matching as covariates. They identified no significant relationships between being designated as an early offset and changes in woody vegetation cover, but these regressions had little explanatory power (e.g. regression comparing changes in woody vegetation cover between early avoided loss offsets and future offsets had an adjusted R^2=0.098), and diagnostic plots showed non-normality in the distribution of residuals. This was expected as linear regression assumes that the outcome variable is unbounded, whereas most of our observations lay on the bound (i.e. most avoided loss offsets had a baseline woody vegetation cover of 1).

Therefore we ultimately conducted a simple comparison of the mean change in woody vegetation cover from 2008-2018 between our early offsets and both sets of controls. When comparing changes in woody vegetation cover between early offsets and future offsets, we compared the sample means using a two-tailed Wilcoxon rank sum test. To evaluate the impacts of offsets compared to matched non-adopter controls, we conducted a two-tailed Mann-Whitney test comparing changes in woody vegetation cover between each paired offset and control. If the means comparisons found a significant difference between woody vegetation loss between offsets and controls, we multiplied the difference in woody vegetation loss between offsets and controls by the total area of avoided loss offsets in our sample to estimate the total area of vegetation saved from clearance by offsets.

Regeneration offsets

Evaluating changes in woody vegetation cover in offsets relative to controls

The baseline woody vegetation cover of our regeneration offsets was not at the bound of our outcome variable, so we analysed the effects of offsets on changes in woody vegetation cover using linear mixed effects models. Linear models are the most commonly-used methods for evaluating the effectiveness of land management on continuous parcel-level land cover outcomes, even when the outcome variable is bounded (Jones & Lewis 2015; Nolte et al. 2019; Archibald 2020). To evaluate the effectiveness of regeneration offsets, we implemented the generalised difference-in-difference framework developed in Wauchope et al. (2021) on our complete set of offsets and control land parcels, running separate regressions for offsets and each of the two sets of controls. The core regression framework could be expressed as

$$veg\ cover_{i,j} = \beta_0 + \beta_1 BA_i + \beta_2 CI_j + \beta_3 T_j + \beta_4 BA_i CI_j + \beta_5 BA_i T_i + \beta_6 CI_j T_i + \beta_7 BA_i CI_j T_i + \boldsymbol{\beta X}_{i,j} + (1|k) + \varepsilon_{i,j}$$

where *veg cover* is our outcome variable given at time step *i* for offset or control land parcel *j, BA* is a dummy variable representing whether the observation is before or after the offset implementation date, *CI* is a dummy representing whether the time series belongs to the control or offset sample, *T* is the year of the observation centred around the intervention year, and $\boldsymbol{X}$ represents a vector of covariates for each land parcel, and *k* represents the overall landholding ID. The coefficient of interest was the interaction term β_7, which represents the difference in the change in trend in forest cover after the offset implementation between the offset and control parcels. Theoretically, for regeneration offsets, the change in woody vegetation cover over time should be more positive after the offset is implemented than before (meaning woody vegetation cover is increasing at a faster rate), and this before-after change should be greater than in the control (given the lack of an intervention). Further information about the meanings of other coefficients is given in Wauchope et al. (2021); none were of direct interest to our research question.

We set our intervention date at 2008 and therefore implicitly grouped together all early offsets implemented from 2006-2008. The assumption that all offsets are implemented in 2008 would be problematic if we were interested in the immediate change in woodland cover resulting from the intervention (β_5), but we were only interested in the long-term change in trend. Time lags between changes in management and woody vegetation growth mean that we would expect little change in woodland coverage caused by changes in management in offset sites to occur immediately (i.e. between 2006-2008), so we lost little relevant information contributing to the change in long-run trend from this assumption.

To account for repeated observations, heteroskedasticity and non-normality, we used a linear mixed effects model with landholding ID as a random effect, de-meaning the covariates to ensure model convergence and using the lme4 package in R version 4.4.1 (Bates et al. 2014; Wauchope et al. 2019). In addition to the biophysical covariates mentioned above, we included X and Y coordinates as covariates to partially address spatial autocorrelation, and included the baseline proportion woodland cover and its square to account for nonlinear relationships between baseline cover and subsequent changes in cover over time (Love 2022). We checked for collinearity between variables using variance inflation factors, and found four variables (rainfall, soil carbon, elevation, temperature) with variance inflation factors above five and evaluated the effect of dropping these variables on our results, finding their exclusion did not affect our coefficient of interest. Diagnostic plots show there remains some residual heteroskedasticity, but linear mixed effects models are robust to violations of the distributional assumptions (Schielzeth et al. 2020). We compared the model performance to an alternative specification where we used a linear model without random effects, comparing model performance based on AICs.

Difference-in-difference analyses rely on the assumption of parallel trends between the intervention and counterfactual sites in the period before the intervention (i.e. 1998-2008), which we tested for through visual inspections and by regressing the pre-intervention woody vegetation cover data against the interaction between whether the site is from the control or intervention sample, and year (Devenish et al. 2022). If the interaction is significant, it implies that there is a significantly different time trend between the offsets and controls.

Coefficient β_7 can be interpreted as the total change in woody vegetation cover in each year that offsets deliver above that delivered by controls following the date of offset implementation. Therefore to estimate the total area of woody vegetation gain attributable to the implementation of regeneration offsets in each year, we multiplied the total area of regeneration offsets in our sample by $1+\beta_7$. To estimate the total change in woody vegetation across the whole 10-year lifetime of these offsets, we multiplied this by 10.

Evaluating the effect of excluding sites burned in wildfires

Wildfires have the potential to bias our results if they impact early offsets differently from controls, as they could generate a difference in woody vegetation cover trajectories between treatments that is not attributable to our treatment effect (offset management). This especially of concern in Victoria during our evaluation period, as the Black Saturday fires in 2009 burned approximately 450,000 ha of bushland. To identify offsets and controls potentially impacted by wildfires, we visually assessed the time-series woody vegetation cover data for each land parcel for unusual reductions in woodland cover, then cross-referenced the parcel location against spatial data on all bushfires in Victoria occurring from 2008-2018 as assembled in Ward et al. (2019). We reran our core analyses excluding the affected land parcels.

Evaluating sensitivity of the results to the threshold between avoided loss and regeneration offsets

To evaluate the effect of the choice of threshold (proportion woody vegetation cover above/below 0.95) used for classifying offsets into the two offset categories, we reran our core analyses at alternative threshold levels (0.9 and 0.8) and summarised the effects on the results.

Evaluating the effects of local spillovers

To test for local spillovers whereby land conversion was displaced from the offsets into the surrounding landscape inflating the rate of habitat loss (Ferraro et al. 2019), we assessed whether any of the matched land parcels fell within 500m of offset sites, and if so, we reran our outcome regressions excluding all these matched land parcels which fell within 500m of the offset sites and investigated the effects on our coefficient of interest.

4.4 Results

4.4.1 Dataset summary

Our study spanned 196 “early-adopter” native vegetation offsets (total area of 3203 ha), 364,290 “non-adopter” land parcels, and 202 “late-adopter” offsets (total area 2174 ha). On average, early-adopter offsets had higher levels of baseline woodland cover, were larger, and were located in different local government areas from offsets allocated from 2017-2019. Details of the distribution of covariates between the samples and the geographical distribution of early- and late-adopter offsets is in the Supporting information (Figure 22, Figure 23).

4.4.2 Avoided loss offsets

Comparison of avoided loss offsets and matched non-adopters

For avoided loss offsets, our best-performing matching specification (Supporting information) was nearest neighbour matching based on Mahalanobis distances with exact matching for land use and a caliper of 0.25 standard deviations. This specification found matches for 138/142 early offsets with standardised mean differences below 0.25 for all covariates.

The mean change in the proportion of woody vegetation cover was +0.002 in offsets and -0.013 in matched non-adopter controls (no significant difference between paired offsets and controls at the 5% significance level, Mann-Whitney test, p=0.09; Figure 11A). There was no difference in woody vegetation change between early offsets and their paired controls for 65 pairs, and controls outperformed offsets for 31 pairs. Offsets outperformed controls for 42 pairs, with 7 pairs where offsets prevented the loss of >10% woody vegetation cover relative to their matched controls.

Comparison of early avoided loss offsets and future offsets

There was no clear difference in the change in woody vegetation cover in early avoided loss offsets from 2008-2018 compared to future offsets (early mean woody vegetation change=0.002, future offsets=0.001, Wilcoxon rank sum test p=0.99; Figure

11B), indicating that offsets protecting existing woody vegetation did not avoid more woodland loss than controls.

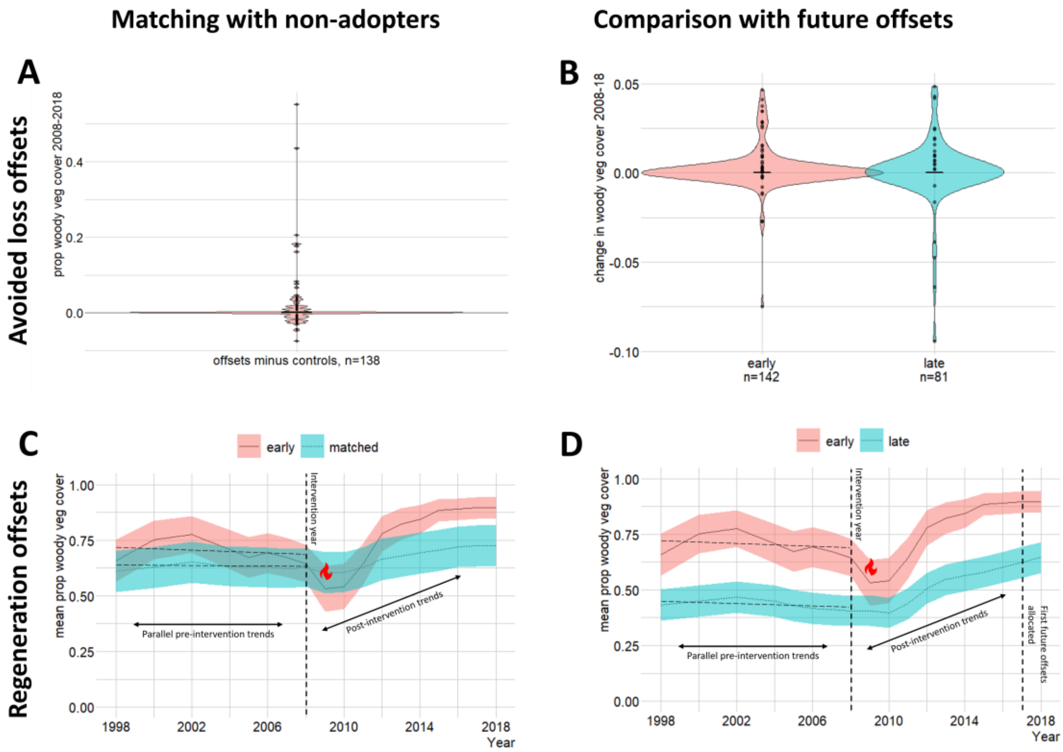

Figure 11. Visual summary of the results of the evaluation comparing native vegetation cover between offsets and controls. A) Combined violin and boxplot of the difference in the change in the proportion of woody vegetation cover from 2008-2018 between early offsets and matched non-adopter control pairs. Positive differences indicate offsets which outperformed their paired controls, negative differences indicate paired controls which outperformed their offsets. B) Combined violin and boxplot of the change in the proportion of woody vegetation cover from 2008-2018 in early offsets compared to future offsets. C) Changes in the mean proportion of sites covered with woody vegetation across regeneration offsets and matched non-adopter controls from 1998-2018. Error bars represent 95% confidence intervals for each year. The flame symbol in 2009 marks the Black Saturday fires, which severely burned one landholding containing 12 offsets. Hatched lines before 2008 represent the pre-intervention trends in the change in woody vegetation cover in the offsets and control sites. Arrows explain the key component of the analysis, comparing the change in trend before and after 2008 between offsets and controls. C) Changes in the mean proportion of sites covered with woody vegetation across regeneration offsets and future offsets from 1998-2018. Error bars represent 95% confidence intervals for each year. Vertical line in 2017 marks the first future offsets entering Victoria's native vegetation offsetting system

4.4.3 Regeneration offsets

Comparison of regeneration offsets and matched non-adopters

For regeneration offsets, our best-performing matching specification was 1:1 nearest-neighbour Mahalanobis distance matching with exact matching for land use and a caliper of 0.25 standard deviations. The standardised mean difference was successfully reduced below 0.25 for all covariates and below 0.1 for 8/13 covariates, indicating high-quality matches (see Appendix). 53/54 offsets from this subset were successfully matched with counterfactual land parcels.

Our test for parallel trends in pre-intervention rates of woodland change held (Supporting information), so we proceeded with the difference-in-difference analysis. Our regression with parcel ID as a random effect yielded a parameter of interest $ß_7$=0.028, p=<0.001, CI= 0.019-0.036 with a model R^2=0.72 (full regression output in Supporting information). This estimate implies that woody vegetation cover increased in offsets by, on average, 2.75% more each year than in counterfactual land parcels post-intervention (Figure 11C). This model also is associated with a lower AIC than our alternative model specifications. Given the area of regeneration offsets was 501 ha, this implies that over the 10-year offset management period regeneration offsets led to a ~138 ha increase in woody vegetation cover relative to controls.

Comparison of regeneration early-adopter offsets and late-adopters

Our test for parallel trends in the pre-treatment period between early and late regeneration offsets found no significant difference in trends (see Appendix), justifying our subsequent difference-in-difference analysis. Our regression model indicated that early regeneration offsets were associated with larger increases in woody vegetation cover than future offsets, indicating that they generated additional gains. Under our core model, our parameter of interest (the coefficient for our interaction terms, $ß_7$) was significant at the conventional 0.05 significance level ($ß_7$=0.0147, p=0.004, CI=0.0037-0.0193, model R^2=0.80). This estimate implies that, for every year after the intervention was implemented, woody vegetation cover rose in regeneration offsets on average by 1.5% in excess of that achieved in counterfactual sites (equivalent to an additional increase in woody vegetation of 74 ha; Figure 11D). This model outperformed alternative specifications according to AIC values.

Effect of removing burned offsets from the analysis

We identified one landholding containing 12 early offsets which was burned completely in the 2009 Black Saturday fires. This reduced the woody vegetation cover of the early offset sample the year after the intervention year of 2008. This led to a subsequent sharp increase in woodland cover as the vegetation regrew, which coincided with the implementation of offset management. When this site was excluded from the sample, our estimate for β_7 changed for both regressions, falling to 0.024 for the comparison with matched non-adopter controls. For the comparison with future offsets, exclusion of these burned offsets led to there no longer being a clear difference in woody vegetation cover trends between offsets and controls ($\beta 7$ = 0.006, p=0.12).

Effect of varying the threshold between avoided loss and regeneration offsets

Varying the threshold (proportion of the site covered by woody vegetation) used to categorise offsets into the avoided loss or regeneration categories had some impact on the magnitude of the impacts of regeneration offsets and avoided loss offsets. The general pattern was for the effect size of regeneration offsets to rise as the threshold fell, whilst this did not impact the outcomes of avoided loss offsets. However, this also led to the classification of fewer offsets (therefore a smaller overall area) as regeneration offsets, which counterbalanced the increase in effect size. For example, if the threshold was set at a proportion woody vegetation cover of 0.8, β_7 rose to 0.04 when offsets were compared with the matched non-adopter controls (equivalent to a 4.04% increase in woody vegetation per year), and 0.02 when compared with future offsets (equivalent to a 2.09% increase in woody vegetation per year. The avoided loss outcomes were unaffected, with neither means comparison yielding a clear difference between offsets and controls. With a threshold of 0.8, the total area under regeneration offsets was 228 ha (avoided loss offset management 2975 ha). Therefore when the threshold was set at 0.8, the analysis implied that over 10 years regeneration offsets led to a gain in 92 ha (compared with matched non-adopter controls) or 48 ha (compared to future offsets) with avoided loss offsets leading to no additional gains in woody vegetation cover.

4.5 Discussion

The results demonstrate that the impact of biodiversity offsets on woody vegetation extent under the Native Vegetation Framework is dependent on the approach used to estimate the counterfactual. When we matched offsets to non-adopter land parcels which were biophysically similar, but did not account for differences in the landholder characteristics, the results suggested that avoided loss offsets had no clear impact on preventing losses of woody vegetation cover and regeneration offsets may have delivered additional increases in woody vegetation cover. If instead we compared early offsets with future offsets (thereby potentially partially capturing self-selection bias; but which were poorly matched on biophysical variables), then avoided loss offsets had no clear impact on woody vegetation cover, and regeneration offsets appeared to have delivered smaller gains in woody vegetation cover. If we further excluded one landholding containing 12 offsets which burned in 2009, there was no clear gain in woody vegetation cover from regeneration offsets (Figure 12).

This evaluation did not permit us to conclude conclusively whether or not offsets under the Framework achieved their policy goal of NNL of habitat hectares as our analysis did not capture impacts on native vegetation condition, and there is little public information on the habitat hectares of native vegetation cleared from 2006-2008 for which the offsets were compensation. One public government document reports that 245 ha of vegetation were cleared under Victoria's native vegetation policy in the year July 2006-August 2007 (Parkes 2007). Other public documentation shows in the 3 years following our evaluation (2008-2011; DSE 2012) 774 ha of native vegetation were permitted to be cleared. Our most optimistic results suggested that avoided loss offsets had no clear impact on woody vegetation cover (i.e. they protected and enhanced the quality of vegetation that would not have been cleared in the absence of offset management), and regeneration offsets led to a 2.8% per year increase in native vegetation cover relative to controls, or a total additional increase of ~138 ha of woody vegetation from 2008-2018. This additional increase is smaller than the known area of losses for a single year out of the three years-worth of offsets included in our evaluation (i.e. offsets allocated from 2006-2008). Our least optimistic results suggest that both types of offsets had no impact on woody vegetation cover relative to controls. These results would mean that no net loss of native vegetation as measured by habitat hectares could only be achieved through large increases in vegetation condition.

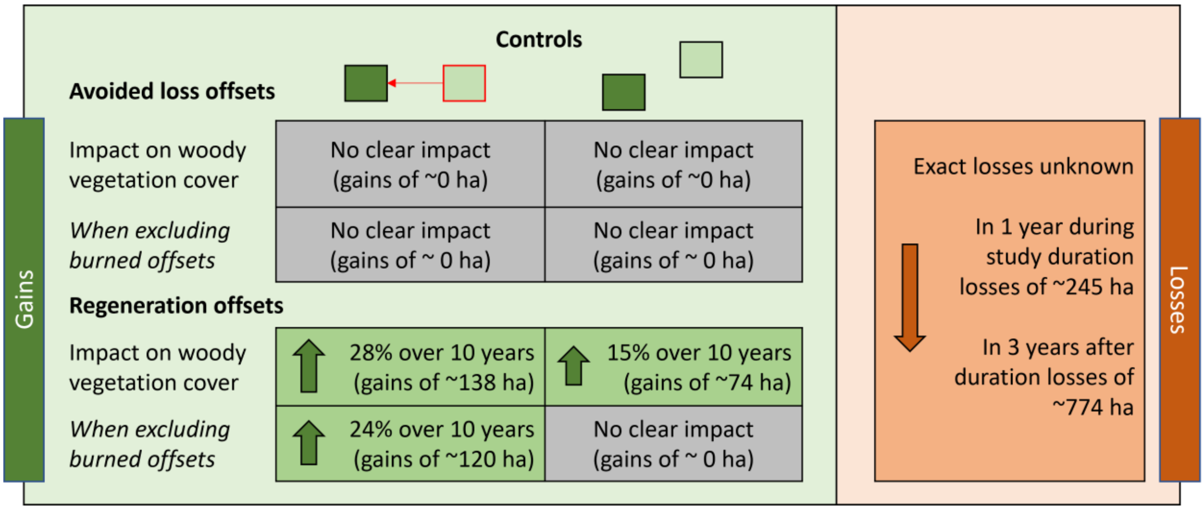

Figure 12. Visual summary of the outcomes of each component of the impact evaluation, and a comparison of woody vegetation losses and gains under the policy.

One limitation is that we evaluate the outcomes of the offsetting system for the duration of the 10-year management contracts. However, in theory offsets in the Victorian system are protected in perpetuity (but see Damiens et al. 2021). Therefore, over longer timescales, as with all avoided loss offsets it may be that the offsets in our sample accrue gains via avoided losses that were not detectable over the time series of this evaluation.

Our analysis focused on the question of whether gains associated with offsets effectively counterbalanced losses, but it should be seen in the context of a recent independent evaluation of Victoria's policies for regulating native vegetation losses on private land which found that native vegetation (habitat hectares) is unambiguously declining across the state, which is not achieving its overall goal of no net loss (VAGO 2022). The evaluation found the primary driver is illegal vegetation clearance for which no compensation is occurring, although various other problems with the offsetting system were identified, including the risk of overallocation of credits from established offsets, and serious governance shortfalls for offsets regulated by councils (which fall under a different referral pathway from the offsets evaluated in this paper).

4.5.1 Implications for offsetting policies

Our analysis has several important implications. Firstly, the evaluation indicates that offsets protecting existing areas with high levels of woody vegetation cover may have done little to protect woody vegetation from clearance, invalidating some of the mechanisms through which security gain and maintenance gain generate credits in Victoria's offsetting system, and threatening the core logic of 'avoided loss' based offset systems (Maseyk et al. 2020). This provides further empirical evidence supporting the already extensive literature showing that the offset multipliers used in offset policies are much lower than the true multipliers required to achieve NNL (Laitila et al. 2014; Gibbons et al. 2016; zu Ermgassen et al. 2019a; Maseyk et al. 2020). A clear implication is the need to increase the size of biodiversity offsetting multipliers used in jurisdictional policies if they are to credibly claim NNL outcomes within a reasonable timeframe.

Another is re-affirming the value of robust study designs and time-series data in impact evaluations. Our results demonstrate that vegetation cover increased in offsets across our time-series, and simple before-after designs would have unambiguously demonstrated that offsets have increased woody vegetation cover. However, the comparison of changes in vegetation against carefully-chosen controls shows that much of this vegetation enhancement would have occurred anyway in the absence of offset management, as vegetation across Victoria recovered following the end of the Millennium drought in 2010. Our analysis adds to the literature highlighting the necessity of applying more robust study designs in conservation science to develop an improved understanding of conservation effectiveness (Ferraro & Hanauer 2014b; Christie et al. 2019; Wauchope et al. 2022).

A third is the complex picture where alternative approaches to estimating the counterfactual yield different answers regarding the policy's outcomes. Offsetting in general has been criticised for its reliance on the construction of complex counterfactuals which are resource-intensive to estimate and often gameable. One plausible alternative is to implement a biodiversity compensation system that does not rely on counterfactual estimation at all, such as the recently proposed 'target-based compensation' approach (Simmonds et al. 2019, 2022). Under the proposals the jurisdiction would set a suite of biodiversity targets outlining the desired future state of biodiversity (e.g. percentage land cover devoted to a set of threatened habitats). Each development which impacted biodiversity would have to compensate through actions that make a contribution towards those pre-agreed biodiversity targets, with the compensation amount a function of the size of the impact, and the disparity between the current state of biodiversity and the desired outcome state (full details in Simmonds et al. 2019). The approach circumvents the need to estimate a context-specific counterfactual.

Perhaps the most novel contribution of this study is the preliminary evidence that self-selection bias may inhibit the additionality of biodiversity offsetting within offsetting regulatory markets, indicated by the generally smaller effect sizes when using the within-sample (early offsets versus future offsets) controls compared with the matched non-adopter controls. The evidence provided here has limitations – our within-sample approach to overcoming self-selection bias relies on the assumption that landholders who opted into offsets in the early and late time-periods share pro-environmental characteristics that affect their entry into the programme. We do not have the data to evaluate how well this assumption holds, or whether the changes in the policy gain-scoring architecture in both 2013 and 2017 substantially changed the economic incentives facing landholders and encouraged landholders to participate who would not have participated in the early enrolment. However, our self-selection bias hypothesis fits with the patterns in our data, the qualitative data from elsewhere in Australia indicating that many landholders enrol in conservation management because they have pro-environmental or land stewardship attitudes (Selinske et al. 2016, 2022), quantitative data showing that vegetation clearance behaviours are partially explained by landholders' psychosocial traits (Brown et al. 2021; Simmons et al. 2021), and empirical work exploring the implications of self-selection bias in PES (Jack & Jayachandran 2019). Ultimately, future work may be able to more rigorously demonstrate self-selection bias by collecting data on landholders' psychosocial traits and modelling their propensity to participate in the offsetting programme as a function of their psychosocial traits (e.g. Archibald 2020; Simmons et al. 2021), then comparing ecological outcomes between participant and non-participant landholders matched on psychosocial traits alongside biophysical covariates.

If we accept that our study design might partially capture the effect of self-selection bias, and that this explains some of the difference in outcomes between our two evaluation approaches, then there would be important implications. Offsetting regulatory markets all over the world select offset sites through a process of voluntary landholder enrolment (Koh et al. 2019), and the gain-scoring methods used to quantify the number of biodiversity credits generated commonly rely on static (e.g. England's Biodiversity Net Gain; zu Ermgassen et al. 2021) or declining (e.g. Victoria, New South Wales; Maseyk et al. 2020) counterfactuals. If a proportion of offsets are delivering gains that largely would have been delivered anyway (i.e. they protect habitat that would not otherwise be under much threat, or lead to biodiversity recovery that would have occurred regardless), this threatens components of the theory of change underpinning jurisdictional offset systems as a mechanism for reconciling development and nature objectives. Jurisdictional offset policies conserve biodiversity through two key mechanisms: 1) they aim to make up for the harm caused by the development project; and 2) they aim to internalise the price of biodiversity loss into the development process, disincentivising damage to areas of high biodiversity in the first place (Calvet et al. 2015; Pascoe et al. 2019b; zu Ermgassen et al. 2020). If the additionality of offsetting actions is questionable, then this partially undermines the first theory of change, and alters the benefits of offsets to more closely resemble those of a direct tax on biodiversity loss, with revenues directed towards agricultural subsidies.

4.5.2 Overcoming self-selection bias in offsetting regulatory markets

In order to overcome the potential problem of self-selection bias undermining the additionality of biodiversity offsets, it would be necessary for offsets to recruit beyond the sample of landholders who have high propensity to participate in the offset programme. Jack & Jayachandran (2019) explored this issue in detail for PES schemes. They demonstrated how enrolment is a function not only of the opportunity cost of conservation (landholders are more likely to enrol if the opportunity cost is low), but also of the enrolment costs (i.e. time and financial costs of bureaucracy and participating in the scheme), and that changing the enrolment costs can alter the characteristics of participants who enrol. To improve enrolment by landholders with low propensity to enrol (who therefore deliver offsets with greater additionality), they explored the idea of manipulating the enrolment costs so that landholders with a low conservation opportunity cost experience high enrolment costs, and vice versa (Jack & Jayachandran, 2019). One mechanism might be to couple environmental payments with incentives that appeal more strongly to landholders with lower propensities to enrol. However, the authors recognise that such an approach has not yet been proven for environmental payments. An alternative would be to make the incentive scheme so large that all landowners with a high propensity to enrol sign up, leaving only landowners with low propensities to enrol (and subsequently higher additionality) to enrol thereafter. We therefore advocate for urgent experimentation in this area to improve the additionality of biodiversity offsets delivered by voluntarily-enrolling landholders.

An additional alternative would be to implement stricter additionality criteria against which to assess enrolment applications, such as asking for a detailed explanation of how the payments from offsetting would allow the landholder to implement conservation measure that they would otherwise not be able to do. We recognise the limitations of such an approach. Regardless of the challenge devising an effective solution, this problem cannot be ignored, as every offset that is not additional fails to compensate for a loss of biodiversity elsewhere thus reinforcing biodiversity loss. A warning from a related policy area comes from recent work evaluating the outcomes of Australia's emissions reduction fund, which has found that a large proportion of the carbon credits from 'human-induced regeneration' were not additional, thereby leaving their associated carbon emissions uncompensated – leaving the public paying several hundred million dollars for no reductions in emissions (Macintosh et al. 2022).

4.5.3 Refocus on the core purpose of offsetting regulatory markets

Biodiversity compensation systems are being increasingly adopted around the world (zu Ermgassen et al. 2019b), and offsets are widely perceived in policy and business circles as a key solution for addressing the global biodiversity finance gap, with Deutz et al. (2020) speculating that offsets might provide up to US$168bn per year by 2030 with favourable public policy. But offsets are not pure conservation funds – they are defensive expenditures (Spash 2015) – each offset is associated with a loss of biodiversity elsewhere. Therefore, each offset that fails to deliver the biodiversity outcomes it was expected to results in harm elsewhere that goes uncompensated for, contributing to further biodiversity declines. In many jurisdictions, biodiversity offsetting regulatory markets have become sizeable industries with their own interests and growth-agendas, estimated at a global market value of $6-9bn (Deutz et al. 2020). But a focus on the market size and performance of biodiversity offsetting fundamentally misidentifies their core policy goal – to deliver as a minimum no net loss of biodiversity. The implications are clear: there is an urgent need to refocus on the real-world ecological outcomes of offsetting and conclusively demonstrate their effectiveness before they are widely adopted as perceived transformational instruments for addressing global biodiversity funding shortfalls.

Chapter 5 The hidden biodiversity risks of increasing flexibility in biodiversity offset trades

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5.1 Abstract

Market-like mechanisms for biodiversity offsetting have emerged globally as supposedly cost-effective approaches for mitigating the impacts of development. In reality, offset buyers have commonly found that required credits are scarce and/or expensive. One response has been to seek improved market functionality, increasing eligible offset supply by allowing greater flexibility in the offset trading rules. These include increasing the size of geographical trading areas and expanding out-of-kind trades ('geographical' and 'ecological' flexibility). We summarise the arguments for and against flexibility, ultimately arguing that increasing flexibility undermines the achievement of No Net Loss (or Net Gain) of biodiversity where high-quality governance is lacking. We argue expanding out-of-kind trading often increases the pool of potentially eligible offsets with limited conservation justification. This interferes with vital information regarding the scarcity of the impacted biodiversity feature, thereby disincentivising impact avoidance. When a biodiversity feature under threat of development is scarce, expensive offsets are an essential feature of the economics of offsetting which communicate that scarcity, not a problem to be regulated away. We present examples where increasing ecological flexibility may be justifying the loss of conservation priorities. We also discuss how increasing geographical flexibility might compromise the additionality principle. We highlight alternative mechanisms for enhancing offset supply without the risks associated with increasing flexibility, including reducing policy uncertainty and improving engagement and awareness to increase landholder participation. Although there are legitimate reasons for increasing offsetting flexibility in some specific contexts, we argue that the biodiversity risks are considerable, and potentially undermine 'no net loss' outcomes.

5.2 Biodiversity offsetting regulatory markets

Biodiversity offsets are a globally significant mechanism for reconciling potential trade-offs between biodiversity and infrastructure expansion or other development projects (Bull & Strange 2018; Shumway et al. 2018; zu Ermgassen et al. 2019a). Offsets are conservation actions taken to compensate fully for the residual biodiversity losses associated with development following the application of a mitigation hierarchy (e.g. avoid, minimise, remediate, offset), with the overall aim of achieving No Net Loss (NNL) or Net Gain of biodiversity (Bull et al. 2013a; Gardner et al. 2013). Within several jurisdictions around the world, offsets are supplied to proponents undertaking development or land clearing through regulatory markets or market-like mechanisms. These are systems characterised by market-like trades between buyers and sellers but are not 'free markets' for various technical reasons, including the buyers being coerced into purchasing through government regulations rather than transactions being voluntary (Koh et al., 2019; Vatn, 2015). Over 81,000km^2 of offsets globally are currently implemented as a result of national biodiversity compensation policies (Bull & Strange 2018).

The ultimate purposes of biodiversity offset regulatory markets are to internalise the value of biodiversity into the land-use planning process and deliver biodiversity gains that fully compensate for losses induced by development activities, in a cost-effective way (Calvet et al. 2015). Commonly, a government regulator sets an overall target outcome (e.g. 'NNL in native vegetation') and facilitates the establishment of trade infrastructure that connects landholders providing offsets with potential buyers, often with the help of brokers (Vatn 2015; Koh et al. 2019). If the market-like mechanism functions effectively, in theory, landholders will compete to deliver the required biodiversity gains at the lowest price – which ultimately allocates the task of biodiversity conservation to the landholder who is able to deliver the required biodiversity most efficiently (Calvet et al. 2015). This relies on the strong and contestable assumption that the landholder does in reality meet their biodiversity obligation (Theis et al. 2019; zu Ermgassen et al. 2019a).

Regulators largely determine the biodiversity outcomes of regulatory offset markets, as they specify the requirements that trades must achieve in order to be compliant. Best practice guidance for voluntary offsets (e.g. BBOP, 2013; IUCN, 2016) suggests that biodiversity trades should be 'like-for-like' or better (i.e. gains or avoided losses should benefit the same biodiversity feature as was impacted, or a biodiversity feature that is more threatened), and should usually occur within the same geographical region. The rationale behind these conventional trading constraints is to maintain the functioning of the impacted ecosystem, and to ensure that the same community of people that loses out on a valuable biodiversity feature maintains access to an equivalent biodiversity feature (Bull et al. 2013a; Griffiths et al. 2019).

Whilst the explicit purpose of many biodiversity offsetting regulatory markets is to achieve NNL of biodiversity, in order to achieve buy-in from regulated industries, in practice offsetting policies are outcomes of negotiation processes among multiple stakeholders (Miller et al. 2015). As a result, offset policies often compromise on ecological theory in order to satisfy other economic or industry objectives (Calvet et al. 2015). This is risky as there is currently limited information on the actual effectiveness of offsetting schemes at delivering appropriate biodiversity gains (Gibbons et al. 2018; zu Ermgassen et al. 2019a). Ways in which economic/cost-reduction priorities interfere with the capacity of offsets to achieve no net loss in biodiversity include situations where policies: a) specify pre-set, often arbitrary multipliers (the ratio of biodiversity gains to losses required by the policy) that are lower than those required to truly achieve NNL (Laitila et al. 2014; Bull et al. 2017a); b) systematically overestimate the counterfactual rates of habitat loss to make offset obligations easier to achieve through 'avoided loss' (offsetting where habitat loss is traded for an increase in the level of protection of existing habitat; Maron et al., 2015); and c) use streamlined and simplified biodiversity assessment methods to reduce transaction burden on developers (Lave et al. 2010; Sullivan & Hannis 2015). There are many mechanisms through which economic considerations can be prioritised over biodiversity. These include pressure from vested interests, and situations where time-stressed under-resourced regulators are implicitly incentivised to rush through approvals without full scrutiny, or deliver outcomes supporting overarching government pro-development priorities over environmental ones (Clare & Krogman 2013; Macintosh & Waugh 2014; Jacob & Dupras 2020).

This paper explores one aspect of offset trades that has so far received relatively little attention in the literature: offsetting flexibility (Habib et al. 2013; Bull et al. 2015; Yu et al. 2018). There are three main categories of flexibility in biodiversity offsetting: ecological, geographical, and temporal (Table 8, see Bull et al. (2015) for additional categories not addressed here). Here, we focus on the implications of ecological and geographical flexibility, as the impact of temporal flexibility (i.e. allowing impacts today to be compensated for through promised biodiversity gains delivered in the future) has been widely discussed and established (Bekessy et al. 2010; Bull et al. 2015; Yu et al. 2018; Weissgerber et al. 2019; Buschke & Brownlie 2020). There is widespread agreement that biodiversity gains achieved in advance of the biodiversity loss associated with development are more likely to deliver NNL and entail better biodiversity outcomes compared to those promised in the future through restoration actions planned over long time horizons.

Regulators set the degrees of flexibility permitted by the policy. Recent evidence (outlined below) suggests that several established offset systems have permitted increasing ecological and/or geographical flexibility over time, consistent with non-ecological objectives such as improving the function of offsetting market-like mechanisms through increasing the ease of trades (Needham et al. 2019). In early-stage offset systems where the regulatory architecture is still under development, such as the UK's Net Gain policy for development activities in England (Defra 2019a), questions surrounding flexibility are fundamental as, once embedded, they determine the future functioning of the policy.

Category of flexibility	Explanation
Ecological (also referred to as 'ecological equivalence')	Biodiversity offset policies have rules that determine which type of biodiversity is considered an acceptable replacement for lost biodiversity. Best practice guidelines promote 'like-for-like or better' trading rules (BBOP 2013; IUCN 2016), whereby lost biodiversity needs to be replaced by the same kind of biodiversity or one that is more ecologically valuable or threatened. As flexibility increases along this dimension, the ecological communities or species targeted by the offset actions can be increasingly different from those impacted by a development activity.
Geographical / spatial	Offset policies normally implement some constraints about where offsets need to be located relative to the impact causing the biodiversity loss. It is widely advocated that offsets should be located as close as possible to the initial impact site, so that people in

	the vicinity retain their access to nature and to improve the chance of ecological equivalence at levels below that of the categorical 'types' of biodiversity (e.g. populations, genes). Commonly, offset policies mandate that trades need to occur within the same administrative unit as the impacts (e.g. Biodiversity Net Gain in England is proposing to penalise trades which do not occur within the same local authority; Crosher et al. 2019a), or within the same defined ecological unit (e.g. compensatory wetland mitigation in the US under the Clean Water Act must occur within the same watershed). As geographical flexibility increases, offset sites can be further from the impact sites.
Temporal (offsets established in advance of biodiversity impacts are often called 'habitat banks')	Offset policies can specify how far in the future biodiversity gains need to be delivered in order to be considered acceptable to compensate for losses today. Some offset policies allow for offsets to deliver gains long into the future (e.g. in the proposed Biodiversity Net Gain policy in the UK, gains can be delivered up to 32 years into the future and count towards acquitting a developers' offset liability; Crosher et al. 2019a), others have constructed systems of habitat banking that ensure that a large proportion of the biodiversity gains are in place before the development impact occurs (e.g. wetland mitigation banking in the US). As temporal flexibility increases, biodiversity gains can be delivered further into the future.

Table 8. Summary of the three main categories of flexibility in biodiversity offset trades

5.3 The arguments for increasing offsetting flexibility

Offset regulatory markets are in general perceived to be inefficient because they are often characterised by low offset supply, high transaction costs (i.e. costs associated with measurement of the value of a trade, search for information, bargaining and decision-making; Cheung 2016) and a low volume of trades for a given credit type (Needham et al. 2019). Some of these transaction costs are essential. Organising offsets (e.g. conducting biodiversity assessments, encouraging landholder participation, monitoring compliance) is time consuming and contractually challenging (Evans 2017; Vaissière et al. 2018); problems which ultimately contributed to the UK's biodiversity offsetting pilot (2012-2014) failing to secure any trades (Needham et al. 2019). These transaction costs can impose additional costs on offset purchasers looking to construct new infrastructure or developments (Buitelaar 2004).

Recent work has outlined that biodiversity offset market-like mechanisms are likely to function most effectively from an economic perspective when they use simple, standardised units of biodiversity, when there is a large offset trading volume, and when there are large geographical trading areas (Needham et al. 2019, 2020). As a result, increasing the geographical flexibility (e.g. Needham et al., 2019) and ecological flexibility (e.g. Habib et al., 2013; Minerals Council of Australia, 2018) of offsets have been proposed as ways of improving the functioning of these market-like mechanisms. The rationale behind this is that flexibility widens the number of offsets that are eligible to compensate for a given biodiversity impact, as the impacted biodiversity feature can be traded for a wider set of potential biodiversity features. Therefore, the supply of potential offsets increases, which reduces prices because competition between landholders to secure a buyer for their offsets increases (although real-world heterogeneity in biodiversity values across jurisdictions may deliver the odd exception, e.g. Needham et al. 2020). Some regulators may also favour flexibility, as it increases the number of eligible offsets sites and therefore may reduce their administrative burden and costs.

There have been various empirical explorations of the potential ecological benefits of offsetting flexibility. Bull et al. (2015) and Habib et al. (2013) have explored the potential biodiversity gains from scrapping the 'ecological equivalence' aspect of offset trades, highlighting that constraining trades to a certain biodiversity feature such as a habitat type might deliver sub-optimal biodiversity outcomes if that feature is common and not considered a local biodiversity priority, and higher priority alternatives are available. Geographical flexibility may also be essential in contexts where impacted biodiversity is highly mobile or migratory, weakening the capacity of equivalent area-based offsets to sufficiently address biodiversity impacts (Bull et al. 2013b). In contexts where the aim is to offset historic habitat loss in highly modified landscapes retrospectively (i.e. after land use change has occurred and with no potential to influence the initial avoidance of impacts), flexibility can be necessary as there may be insufficient remaining appropriate offset sites. For example, Yu et al. (2018) describe an example from the Yellow River Delta in China where the only way that no net loss for each impacted wetland type could be achieved was through expanding the geographical scope of offsetting, allowing for offsetting in neighbouring regions. In some of the world's most prominent offset systems in Australia (Box 3), calls for increasing the flexibility of offset trading certainly resonate with influential vested interests whose activities are being regulated through offsetting policy. For example, relaxing the 'like-for-like' requirements of offsetting policies is the stated preference of some key business stakeholders, such as representatives of extractive industries (Minerals Council of Australia 2018).

Reflecting these issues, there is pressure from regulated industries and deregulation-friendly governments to implement policy changes to reduce transaction costs and stimulate offset supply (Lave et al. 2010; Apostolopoulou & Adams 2017), with environmental regulation perceived as a barrier to development. Pressure from regulated stakeholders to prioritise economic over ecological objectives is to be expected since biodiversity offsetting creates a regulatory framework through which the biodiversity impacts of new developments are internalised and accounted for within the development process. This imposes a cost that regulated industries were previously able to externalise onto society as a whole.

In several Australian offset systems, there is evidence that states are under pressure to increase offsetting flexibility (Ives & Bekessy 2015; Nature Conservation Council of NSW 2016). For example, Queensland recently reviewed their native vegetation offsetting policy and emphasised the ambition to reduce 'green tape' (Queensland Government, 2019a, p11), with the justification that 'some proponents have experienced difficulty addressing impacts for environmental values which cannot be offset'. Victoria's most recent review highlighted the need to 'support the development of the market for low availability offsets' (Department of Environment, Land, Water and Planning 2016), and a major motivation behind New South Wales's biodiversity legislation reform was to 'provide greater levels of flexibility to industry and landholders on how they manage biodiversity, including native vegetation' (Byron et al. 2014). These shifts aim to increase the supply of offset credits (thereby theoretically reducing prices). Consequently, numerous policy statements and modifications have occurred which increase offsetting flexibility (Box 3).

Box 3. Examples of Increasing flexibility in Australian state biodiversity offsetting systems

In Australia, biodiversity offsetting has emerged as a key tool in the policy mix aiming to reduce rapid rates of deforestation and biodiversity loss across the continent (Bradshaw 2012; Miller et al. 2015; Kearney et al. 2019). Australia has lost one third of all its native vegetation since European settlement, and 61% of all 1,136 nationally-listed threatened species are threatened by habitat loss (Kearney et al. 2019; Ward et al. 2019). Partially in response, most states have biodiversity offsetting policies, with the two most well established in the states of New South Wales and Victoria.

The first offsetting system in New South Wales (BioBanking; first offset trades in 2010) specified like-for-like trading requirements for both ecosystem types and threatened species (Department of Environment, Climate Change and Water 2010). Since the introduction of the Biodiversity Conservation Act in 2017, a new level of flexibility has been incorporated into the state's approach to offsetting. Developers now have the choice of passing on their offset liability by paying into the Biodiversity Conservation Trust, a government-run fund. Although a hierarchy of preferred offsetting options is specified (i.e. preference is given to like-for-like or better in the same bioregion) there are no legal restrictions on the Trust offsetting using any habitat types anywhere in the state. As such, the option is open for both ecologically- and geographically-flexible offsetting.

In Victoria, there is also evidence of a trend (albeit less severe) towards flexibility from the original 2002 Native Vegetation Framework (Department of Sustainability and Environment 2002) to today's native vegetation removal regulations. Notably, the policy goal was weakened from 'net gain' to 'no net loss' of biodiversity (Department of Sustainability and Environment 2002; Department of Environment, Land, Water and Planning 2017a). Initially, offset legislation incorporated a graded response, whereby strict like-for-like trades were required for vegetation of 'Very High' conservation significance and progressively weaker rules were allowed for vegetation as conservation significance decreased (Department of Sustainability and Environment 2002). Since 2013 for general offsets (offsets for impacts to native vegetation where there are no threatened species present), there are no like-for-like restrictions. Offsets are required to have at least 80% of the 'strategic biodiversity score' of impact sites, which is a score derived from a systematic conservation prioritisation approach broadly representing habitat condition and rarity as well as the number of threatened species present (Department of Environment, Land, Water and Planning 2017b). General offsets do have a geographical restriction, and are constrained to the same Catchment Management Authority or municipal district. For offsets to threatened species, there is a 'like-for-like' requirement, but no geographical restrictions on where those offsets are located throughout the state.

There are two major underexplored and unquantified risks of increased flexibility that threaten to undermine the desired NNL outcomes of offsetting market-like mechanisms. The first risk associated with increasing the ecological flexibility of offsetting (i.e. relaxing like-for-like) – especially in response to low supply – is the risk of interfering with information regarding the genuine scarcity of the impacted biodiversity feature, potentially disincentivising impact avoidance. The second risk, associated with increasing geographical flexibility, is that larger trading areas have the potential to deliver offsets with lower additionality, undermining the conservation outcomes associated with the offsets.

5.4 Flexibility interferes with information about biodiversity feature scarcity and disincentivises avoidance

Under best practice, biodiversity offsets must be implemented as an option of last resort, preceded by the implementation of the first three steps of the mitigation hierarchy (avoidance, minimisation, remediation). In many cases biodiversity offset systems trade uncertain future biodiversity gains for imminent losses (Maron et al. 2012; Weissgerber et al. 2019). This lack of certainty that the intended biodiversity gains will be delivered in reality means avoiding impacts – step 1 of the mitigation hierarchy – is crucial (Buschke and Brownlie, 2020; Hough and Robertson, 2009; Phalan et al., 2018 but see Bull and Milner-Gulland, 2019). Note here that there is a distinction between avoidance (i.e. avoiding impacts to biodiversity initially, first step of the mitigation hierarchy) and 'avoided loss' offsets, which are offsets that prevent biodiversity losses in an area that likely would have occurred without the offset. Under best practice principles, avoidance should be rigorously applied as the first step of the hierarchy, meaning that promises of compensation should not influence the requirement for adequate avoidance. However in practice, in some offset systems, despite the rhetoric of avoidance, offsetting appears to be the default response to biodiversity impacts (Clare & Krogman 2013; Martin et al. 2016; Samuel 2020). As a result, there is in reality a significant interaction between the offsetting and avoidance steps: a high degree of difficulty, and in particular a high price, associated with acquiring appropriate offsets will incentivise avoiding impacts in the first place (Koh et al. 2017; Pascoe et al. 2019a). In this way, offsets represent a punitive-tax-like incentive to avoid causing biodiversity loss initially.

The major risk therefore with increasing ecological flexibility is that it interferes with communicating the scarcity of the impacted biodiversity feature, thereby potentially disincentivising avoidance of impacts to threatened biodiversity features. Offset requirements are often triggered specifically because the biodiversity feature impacted is threatened and/or scarce. If a threatened or scarce biodiversity feature is required to be compensated by the same biodiversity feature, then it is likely to be difficult or impossible to acquire an appropriate offset (because by definition, appropriate offsets are scarce), therefore incentivising avoiding impacts to that biodiversity in the first instance (Koh et al. 2017). Under conventional economic theory (critically explored in Spash, 2015), we might expect that the scarcity of the impacted biodiversity feature would be communicated via a higher price for appropriate offsets, although evidence from offsetting markets in the US suggests that offset prices do not always predictably follow changes in supply and demand (Robertson 2007, 2008). Nevertheless, the difficulty of acquiring an appropriate offset provides essential biodiversity supply information – as such, it is an *essential feature* of biodiversity offset regulatory markets aiming to achieve NNL outcomes, not an economic inconvenience to be regulated away by increasing the supply of eligible offsets through increased flexibility of trades.

Opening the door to increased flexibility can have very real consequences for threatened biodiversity (Figure 13). For example, in Australian biodiversity offsetting systems there are rarely restrictions that absolutely prohibit impacts on particular biodiversity features, with offsetting usually permitted if a compliant offset can be secured (e.g. Queensland permits the clearance of vegetation in national parks in return for offsetting with a multiplier of 10; Queensland Government 2019b). As a further example, under Western Sydney Growth Centre's biodiversity offset program, flexibility has been built into the offset requirements. The Growth Centre (which includes plans to construct 200,000 new homes) impacts on several critically-endangered ecological communities, including Cumberland Plain woodland and Shale Sandstone transition forest. Since the inception of the Growth Centre in 2007, over 300ha of these two communities have been converted to other land uses (Government of New South Wales 2018). Whilst the scheme has so far achieved its (like-for-like) offsetting 'requirements', the Growth Centre was permitted to proceed with a commitment to avoided loss offsets using a multiplier of 1 (Government of New South Wales, 2006, p.6), which is well below a level sufficient to achieve NNL (Laitila et al. 2014). In effect, this equates to committing to a halving of remaining ecosystems. Regardless, in the event that there is 'insufficient available land' (p.13) for offsetting these threatened habitats, the inbuilt flexibility permits the program to offset using any grassy woodlands within the ecoregion, or indeed any other potential native vegetation. In short: flexibility circumvents the market incentive to avoid impacts to valuable habitats. Indeed, it may well permit considerable losses of these critically endangered habitats. If flexibility were not permitted, the scheme would have to avoid impacts to these ecosystems initially. This may come at some financial cost, but ultimately the policy goal is to achieve NNL of biodiversity and in this case flexible offsets do not facilitate this outcome. Of course, the loss of natural habitats also comes at considerable economic cost which is largely unaccounted for in offset pricing (including both market values such as the traded value of the price of carbon stored within natural habitats, and non-market values such as biodiversity's existence value, and underpinning the resilience of delivery of other ecosystem services). The example from the Western Sydney Growth Centre is not a unique case – similar dynamics have been found for offsetting under Australia's national environmental law, the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act). Instead of disincentivising impacts to biodiversity initially, the inability of proponents to satisfy the 'like-for-like' requirements of the EPBC Act Environmental Offsets Policy has led to instances of flexibility in offset conditions, and an expansion of indirect (e.g. 'out-of-kind') offsetting. Such offsets do not result in a conservation gain for the affected biodiversity, thereby implicitly facilitating the loss of valuable biodiversity (Australian National Audit Office 2020).

5.5 Expanding geographical trading areas may undermine additionality for avoided loss offsets

A second argument for promoting strict trading rules that applies specifically to avoided loss offsets is that offsets within smaller geographic trading regions may yield greater conservation additionality than larger areas (Giannichi et al. 2020). Additionality is a key concept in biodiversity offsetting: for an offset to truly achieve NNL, it must achieve a conservation gain that would not have happened in the absence of the activities associated with the offset (McKenney & Kiesecker 2010;

Gordon et al. 2011; Maron et al. 2013). All things being equal, large trading regions are likely to contain a larger number of potential offset sites which are under limited or no threat of development (Giannichi et al. 2020). The problem with increasing geographical trading areas in the name of improving functioning of the market-like mechanism in theory is that areas under low development pressure tend to 'soak up' the offset obligations of areas with high rates of development because of the lower economic opportunity costs of offset establishment (Giannichi et al. 2020; Kalliolevo et al. 2021; Figure 13). Areas with low development pressure are those that are least likely to be under threat. As a result, offsets located in these areas will offer the least additionality (conservation management will deliver a smaller conservation benefit relative to the counterfactual of what would have occurred without the offset). Similar patterns have been demonstrated to undermine the effectiveness of protected areas (Venter et al. 2018; Geldmann et al. 2019). Relaxing the geographical restrictions on offsets may tend to draw offsets away from areas where they would be more likely to be additional, and drive them towards areas where they offer limited gains relative to the counterfactual, thus undermining NNL outcomes. As a separate issue, smaller trading regions may also be socially desirable because of the potential inequity of reducing affected peoples' access to nature by relocating it further away (BenDor et al., 2007; but see Bateman and Zonneveld, 2019).

This issue is compounded if policies do not have robust methods for assessing offset additionality (Maseyk et al. 2020). In some avoided loss offsetting policies, including Australian native vegetation offsetting systems, the way additionality is operationalised is that sites which are not under formal legal protection are implicitly assumed to be under threat of land clearing or ecological degradation, regardless of their location or threat level (Maron et al. 2015). Hence, the policies assume that simply giving an unprotected site legal protection through an offsetting agreement achieves an outcome that is additional (e.g. in Victoria, this is referred to as 'security gain'; Department of Environment, Land, Water and Planning 2017c). However, working out whether an offset is truly additional requires an analysis of future threats (and probabilities of ecosystem degradation) to the site under the offsetting and counterfactual scenarios, which is rarely done in practice. Protection may well not deliver additional gains in some contexts (Sonter et al. 2020b). For example, if a patch of native vegetation that is not under formal protection is still standing after decades of land management, it is likely that that patch is under limited threat (i.e. because either clearing is uneconomic for landholders, or because the landowners hold pro-environmental attitudes and would likely have maintained that land even in the absence of formal protection; Selinske et al. 2016). Other ways that additionality is commonly operationalised include using offset multipliers or assuming that offset management will improve the future biodiversity value of a site, but neither of these guarantee additional outcomes (Bull et al. 2017a; Dorrough et al. 2019). Further, offsets can only deliver gains due to avoided losses if they protect habitat that is itself not subject to a mandatory offset requirement following clearance (see discussion in Maron et al. (2018) and Maseyk et al. (2020); Figure 13). Type 1 impacts are impacts that would themselves trigger their own offset requirement (e.g. clearance of native vegetation for a new development), and Type 2 impacts are those that would not trigger their own offset (e.g. offsets that prevent a threatening process that is not subject to an offset, such as livestock grazing in areas of native vegetation). In reality, avoided loss offsets preventing Type 1 impacts offer no additionality, because they prevent the clearing of something that would trigger its own offset requirement if cleared. Only avoided loss offsets preventing Type 2 impacts can offer any true additionality. However, avoided loss offsetting systems in operation today in general fail to account for this subtlety, and this remains a fundamental flaw in the way gains due to avoided losses are calculated, undermining the ability of avoided loss offsets to achieve no net loss of biodiversity.

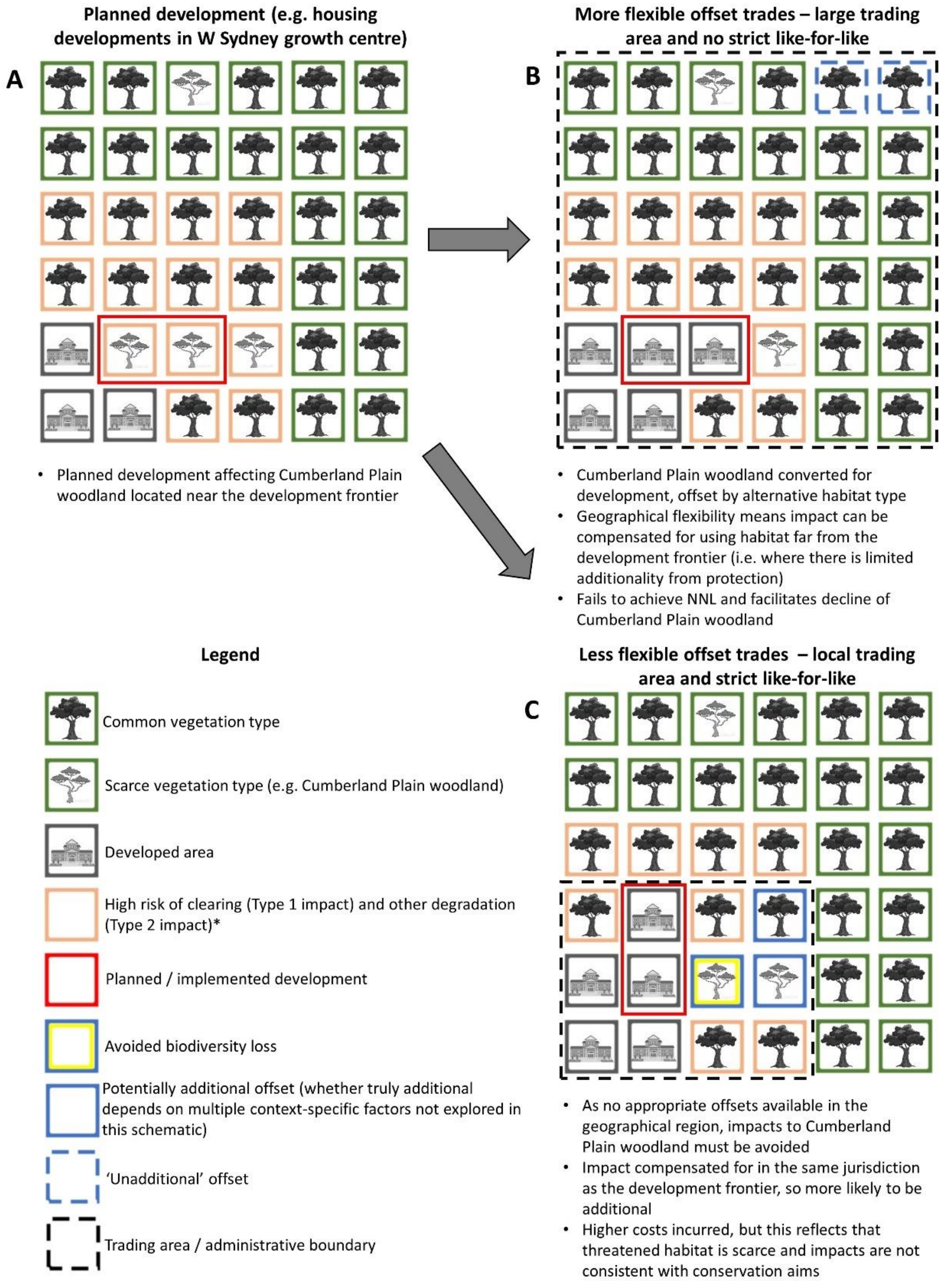

Figure 13. Schematic diagram of the potential biodiversity outcomes associated with flexible, and inflexible offset trading using an illustrative case study (offsetting impacts to Cumberland Plain woodland under the Western Sydney Growth Centre offsetting policy). A) A development is planned, proposing to clear two units of Cumberland Plain woodland. B) Under more flexible trading rules, a different habitat type is permitted to offset impacts to the threatened habitat type, and the offset is located far from the development frontier. This means the incentive to avoid impacts to the threatened habitat initially has been undermined. Additionally, as the offset is located far from threats, the offset is likely to offer less additionality. C) Under less flexible trading rules, offsetting impacts to Cumberland Plain woodland has become challenging or expensive, as available offset sites are scarce.

This incentivises developers to change their development plan to avoid some impacts to the threatened habitat initially. One unit of Cumberland Plain woodland is lost for development, and compensated for with an offset, and another unit of loss is avoided entirely. Whilst this schematic is used here to demonstrate the ecological benefits of rigid trading rules, in the real world the effectiveness of this offset at achieving NNL is dependent on other complexities not explored here. These complexities include the actual offset ratio used for the avoided loss offsets (1:1 as demonstrated here is far too low to achieve anywhere close to absolute NNL in reality); and the degree to which offsets are preventing Type 1 versus Type 2 impacts.

Whilst the risk of geographical flexibility undermining additionality applies primarily to offsetting policy frameworks that permit the preservation or enhancement of existing habitats as offsets, this is non-trivial – avoided loss offsets represent approximately 20% of all of the world's recorded biodiversity offsets by number, covering approximately 50,000km^2 (Bull & Strange 2018). Systems based purely on restoration or habitat creation, such as those in the US and Germany, are less likely to suffer from these drawbacks as additionality is implicit in the act of habitat creation (assuming that habitat would not have passively regenerated under the do-nothing counterfactual; but see Sonter et al., (2017)). This risk would be reduced for offsets which calculate the additionality on a case-by-case basis and integrate this into the calculation of the appropriate offset multiplier (e.g. some large voluntary offsets summarised in Maseyk et al. 2020).

5.6 Improving regulatory market function without inducing the risks of flexibility

As discussed above, flexible trading rules are perceived as solutions to issues relating to 'thin markets' (characterised by low offset supply), including price volatility, strategic behaviour, and market collapse (Adjemian et al. 2016; Needham et al. 2019). However, we contend that there are other mechanisms that can be used to improve the function of offset regulatory markets without introducing offset flexibility that risks undermining biodiversity outcomes. The key point is that the difficulty of securing an appropriate offset trade is a function of two properties: a) the availability of appropriate offset sites containing the impacted habitat type (itself influenced by the absolute scarcity of the threatened habitat type); and b) transaction costs associated with the process of offsetting. The aim of actions to improve the functioning of the market-like mechanism should ideally be to reduce transaction costs, whilst leaving the information about the scarcity of the biodiversity feature intact. Some of the key determinants of transaction costs include a lack of landholder awareness about offset policies, regulatory uncertainty (the regulations surrounding offset policies tend to change frequently), and the degree of trust landholders have in offset administrators (Coggan et al. 2013). These factors can be improved without changing offset trading rules through increasing investments in education and communication about the programme, engagement with previously unreached landholders, and introducing policy stability by committing to keeping the regulation unchanged for a set period of time. We acknowledge these are challenging, but it should not be the default option to increase flexibility and risk the policy's ecological outcomes just because alternative mechanisms for improving market function are difficult to achieve. Additionally, an important driver of price volatility and strategic behaviour between buyers or sellers in thin markets is asymmetrical information (Adjemian et al. 2016), which occurs when one party has better information about the market or the good/service being transacted than the other party. For example, the offset seller is likely to have a better understanding of their true opportunity costs than the buyer, which may permit them to charge higher prices than the seller would in reality be willing to accept. This can be addressed through better public offset registries and data on offset transactions, such as the public offset registries implemented by Western Australia or France (Government of France 2020; Government of Western Australia 2020). All of these actions could be implemented without interfering with the information about the scarcity of the biodiversity feature.

5.7 When flexibility is justifiable

From a biodiversity perspective, we would argue that flexibility is rarely justifiable once real-world implementation issues are taken into account. Institutional factors that influence when flexibility is justifiable include when: a) the offsetting market-like mechanism is embedded within a planning system that includes strict avoidance of threatened biodiversity features; or b) regulatory institutions have the capacity and resources to implement strategic offsetting actions whose biodiversity benefits unquestionably exceed those of like-for-like trading rules. In planning systems with strict avoidance, if implemented effectively, flexibility cannot be used for legitimising losses to threatened biodiversity. For example, the proposed Net Gain policy in England is explicit that offsetting under Net Gain will not weaken existing protections for biodiversity, or be used to justify impacts to irreplaceable habitat (Defra 2019a). This protection is imperfect, and harm to irreplaceable biodiversity can still occur if it is considered to be in the overriding public interest for political or economic reasons. However, in these contexts a bespoke compensation package is agreed and it does not occur within the framework of the offsetting policy. The argument for flexibility being more justifiable where regulators have high levels of capacity is that centralised bodies may able to implement a more systematic and well-planned approach to offsetting that targets local biodiversity priorities than case-by-case offsetting (Habib et al. 2013). However, so far implementation of these approaches has been limited (for example, as of February 2019, of the AUD$9.6 million paid into Queensland's offset fund at the time, only AUD$1.5 million had been committed or spent on offsets; Queensland Government 2019a). Until such systems are demonstrably effective, we suggest that this this approach to enabling increased flexibility through a centralised body will undermine impact avoidance and conservation outcomes (Figure 14).

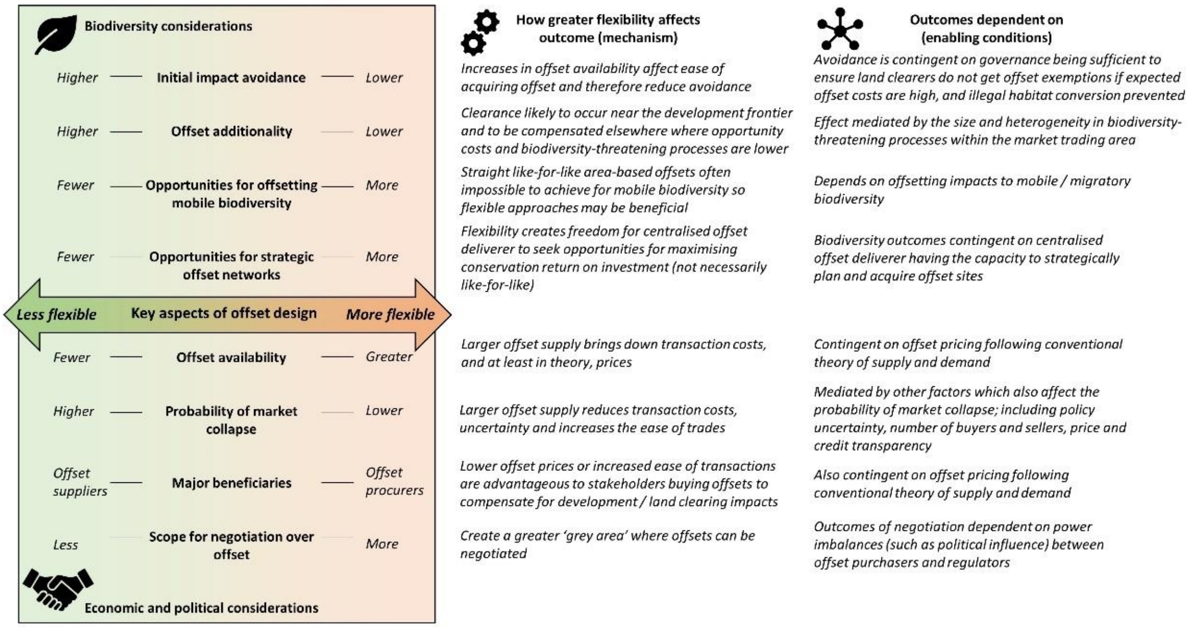

Figure 14. Summary of the advantages and disadvantages of changing the flexibility of offset trades, the mechanisms through which outcomes are achieved, and the factors that those outcomes are dependent on. The top half denotes ecological outcomes, and the bottom half denotes economic and political outcomes.

It is especially important that conservationists are alert to when flexibility is being advocated for purely because appropriate offsets are expensive: indeed, if offsets for a specific biodiversity feature are expensive, this may well be an indication that the biodiversity feature is scarce or threatened and so flexibility might not be justified for that feature. In these cases, the worst outcome from a biodiversity perspective is that regulators deprioritise offsetting exactly because it is expensive – a situation aptly demonstrated by the Warragamba Dam proposal in New South Wales, where a state-owned utility company attempted to reclassify impacts to critically-endangered species habitats as 'indirect impacts' in order to avoid their high offset costs (Hannam 2020; Sanda 2020). Deprioritising offsetting when expensive gravely undermines the economic logic for having offsetting systems in the first place.

5.8 Getting the 'right' level of flexibility

The major difficulty in setting the 'optimal' degree of flexibility that should be permitted in an offsetting system is that ultimately the outcomes of flexibility are mediated by an unobservable characteristic, which is the intention or motivation behind the actor advocating flexibility. Simplistically, the ideal policy from a biodiversity perspective (which is the stated purpose of NNL policies) would allow flexibility when it helps with the achievement of the specific policy goal (i.e. is motivated by achieving NNL or net gain in biodiversity), and restrict it when motivated by other factors which undermine the likelihood of achieving the policy goal, such as simple cost minimisation. In practice, this information is challenging to discern, and so regulators rely on heuristics such as 'like-for-like or better' trading rules, with each policy determining the classifications for what types of biodiversity count as like-for-like (McKenney & Kiesecker 2010). Although 'like-for-like or better' trading rules are widely accepted, it is worth reflecting that, supposedly in the name of NNL, many of these rules permit the loss of threatened biodiversity as long as it is replaced with other types of threatened biodiversity. Such a premise has recently been questioned under the newly-proposed 'target-based ecological compensation' framework (Simmonds et al. 2019), where it has been suggested that 'drawing down' on existing biodiversity should only be permitted if that biodiversity feature is above its 'target level' (i.e. for a species, an appropriate target might be not being classed as threatened on the IUCN Red List; for a habitat type, it might be a target percentage of the historical habitat extent remaining). Similar principles might be used to determine when flexibility is considered acceptable, with the exact threshold tailored to the local policy aim and context.

5.9 Implications for existing and emerging offsetting systems

The main implication is that regulators setting the trading rules of offset policies need to be aware that there are multiple mechanisms for dealing with problems associated with thin markets, each associated with drawbacks and advantages. Although offset policies are in practice always imperfect because they are trying to satisfy multiple objectives (i.e. ecological, economic, political), some changes which are intended to satisfy non-biodiversity objectives can fundamentally undermine the core biodiversity objectives. Additionally, they can somewhat undermine the theoretical strengths of even applying market-like mechanisms to biodiversity management issues in the first place. Changing the level of flexibility inevitably generates winners and losers, and it is always worth questioning who they are and why their interests are being prioritised or deprioritised. In general, we contend that increasing flexibility tends to increase satisfaction of economic objectives and favour the interests of offset procurers (e.g. developers). Given the current generally low capacity of offset system regulators, this often detracts from the ecological objectives of the policy.

In the case of Australian offsetting systems, we would suggest that policymakers need to consider whether overall biodiversity outcomes (the sum of biodiversity impacts avoided - as step one of the mitigation hierarchy - and those successfully offset) are more likely to achieve NNL objectives under flexible or strict trading policies. We would argue that, as it stands, no net loss is more likely to be achieved under strict policies. There are also important lessons for all of the world's many emerging offsetting and biodiversity compensation systems (zu Ermgassen et al. 2019b), as decisions on trading rules embedded at the outset have an overwhelming influence on their biodiversity impacts (Calvet et al. 2015). There are less significant implications for North American offsetting systems, both because the policies already freely allow trades between different types of wetlands (i.e. they are highly ecologically flexible), and because they are primarily restoration-based programs, so it is usually easier to ensure that offsets are truly additional

5.10 Conclusion

The case has previously been made for increasing the flexibility of biodiversity offset trades (Habib et al. 2013), however, here we argue that restricting the flexibility of trades has some highly desirable properties. Most importantly, in offsetting systems where impact avoidance is imperfect and is influenced by the difficulty of securing offsets, like-for-like offsetting drives the unobservable process of impact avoidance (Pascoe et al. 2019a), whereby threatened aspects of biodiversity remain unimpacted because insufficient offsets are available. This process has been largely unaddressed in the offsetting literature (Phalan et al. 2018), even though avoidance is widely considered the most important aspect of the mitigation hierarchy (Hough & Robertson 2009; zu Ermgassen et al. 2019b). Geographical trading restrictions also have the potential to enhance the additionality of offsets, which is a fundamentally important property that defines their associated biodiversity outcomes (Gordon et al. 2011; Maron et al. 2013). To ensure biodiversity offsetting market-like mechanisms are fit to tackle ongoing biodiversity declines we encourage policymakers and practitioners involved in existing offsetting systems and emerging systems around the world to prioritise the biodiversity objectives of these policies. Ultimately, this requires clear thinking about whether increasing flexibility helps to achieve these policies' fundamental biodiversity goals, or hinders them.

Chapter 6 Exploring the ecological outcomes of mandatory Biodiversity Net Gain using evidence from early-adopter jurisdictions in England

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6.1 Abstract

Net outcome-type biodiversity policies are proliferating globally as perceived mechanisms to reconcile economic development and conservation objectives. The UK government's Environment Bill will mandate that most new developments in England demonstrate they deliver a biodiversity net gain (BNG) to receive planning permission, representing the most wide-ranging net outcome-type policy globally. However, as with many nascent net-outcome policies, the likely outcomes of mandatory BNG have not been explored empirically. We assemble all BNG assessments (accounting for ~6% of England's annual housebuilding and other infrastructure) submitted from January 2020-February 2021 in six early-adopter councils who are implementing mandatory No Net Loss or BNG requirements in advance of the national adoption of mandatory BNG, and analyse the aggregate habitat changes proposed. Our sample is associated with a 34% reduction in the area of non-urban habitats, generally compensated by commitments to deliver smaller areas of higher-quality habitats years later in the development project cycle. Ninety-five percent of biodiversity units delivered in our sample come from habitats within or adjacent to the development footprint managed by the developers. However, we find that these gains fall within a governance gap whereby they risk being unenforceable; a challenge which is shared with other net outcome-type policies implemented internationally.

Keywords: Biodiversity offsetting, ecological compensation, environmental governance, environmental policy, impact evaluation, impact mitigation, market-based instruments, nature conservation, biodiversity net gain, no net loss

6.2 The challenge of reconciling biodiversity conservation with infrastructure expansion

Under the Sustainable Development Goals (SDGs), the global community has simultaneously committed to rapidly expanding built infrastructure networks (SDG 9), whilst ending biodiversity loss (SDGs 14 and 15). However, historically the unmitigated impacts of infrastructure have been a dominant driver of biodiversity loss, threatening one-third of IUCN Red List species (https://www.iucnredlist.org/). To reconcile the SDGs, fundamentally new approaches to infrastructure implementation are required (Thacker et al. 2019). A particular class of policies emerging globally to address this focus on achieving No Net Loss (NNL) or Net Positive biodiversity outcomes from new developments (Bull & Strange 2018; Bull et al. 2020; Milner-Gulland et al. 2021). These are predicated on the concept that infrastructure and biodiversity conservation can theoretically go hand-in-hand if infrastructure is planned to avoid and minimise impacts, and residual impacts are compensated for through conservation actions. There is a wide variation in these policies' effectiveness (zu Ermgassen et al. 2019a), with limited systematic understanding of when they work or fail. The most wide-ranging of these policies globally is the proposal, outlined in the UK Government's Environment Bill, for development under the Town and Country Planning Act (i.e. nearly all residential, commercial and mining construction) in England to deliver a mandatory Net Gain in biodiversity. The Environment Bill is expected to be ratified in 2021, with the mandatory requirement for Biodiversity Net Gain (BNG) implemented after a two-year transition period.

Like many densely-populated wealthy nations, England faces interlocking socio-ecological policy challenges: it is ecologically impoverished, with ongoing wildlife declines (State of Nature Partnership 2019). However, it has committed to building 300,000 new homes annually by the mid-2020s (Ministry of Housing, Communities and Local Government 2018), and has promised heavy investments in new infrastructure through its post-Coronavirus recovery strategy (HM Treasury 2020).

Mandatory BNG might partially reconcile these challenges (Defra, 2018, p4), and is globally relevant in the context of finding policy solutions to mitigate the environmental impacts of the global infrastructure boom (zu Ermgassen et al. 2019b).

6.3 Implementation of the mandatory Biodiversity Net Gain requirement

Developers in England will have to demonstrate their proposals achieve a net gain in biodiversity (measured using a government-prescribed biodiversity metric) to receive planning permission from local planning authorities (LPAs), who ultimately assess all of the development plans associated with the site (which can include various economic, social and environmental impact assessments, construction plans, feasibility studies etc.) and decide whether projects have the right to proceed. Currently, BNG assessments align with the ecological impact assessment (EcIA) process, taking information routinely collected during pre-development ecological surveys and feeding this through an Excel-based biodiversity calculator tool, the "Biodiversity Metric 2.0" (Treweek et al. 2010; Crosher et al. 2019a). The Metric is a multiplicative composite indicator converting inputs including the area, habitat condition, habitat distinctiveness, and various multipliers (capturing elements including the risk of project failure, the expected time taken for the proposed habitat to reach its desired condition level, and the landscape-scale ecological importance of the site) for each habitat patch within the development footprint into an overall biodiversity score measured in 'biodiversity units' (Supporting information). The data required from the project site include quantitative data (the area of each habitat patch within the development site and in the proposed post-development plan), qualitative judgements from ecological consultants regarding the habitats' condition and classification, and some landscape-scale information such as whether the project site lies within an area of landscape-scale importance to biodiversity. These data gathered at the project site are integrated in the Metric with other ecological information which is pre-set for each habitat type and condition level based on expert judgement (e.g. each habitat is given a pre-set distinctiveness score within the Metric;

pre-set values capture how long it takes for a given habitat to reach a given condition level under ecological management measures). It calculates the number of baseline biodiversity units within the development footprint plus (where applicable) associated compensation areas owned/managed by the developer, and compares this with predicted post-development biodiversity units. The Metric also provides guidance on whether like-for-like trades should be required for the specific habitat types included in the assessment (e.g. for high distinctiveness habitats), or whether other trading rules are permitted (e.g. for low distinctiveness habitats). The mandatory BNG-requirement necessitates the overall post-development biodiversity score is ≥10% higher than the baseline. If not, the developer must either alter their project plan appropriately or deliver the unit shortfall by offsetting through a payment to the council or a third party (e.g. habitat bank) which is then liable for delivering biodiversity gains elsewhere. If no compensation sites are available within the LPA where the development is planned, then compensation is permitted in other local authorities; but this triggers a spatial multiplier within the Metric which increases the compensatory units required. As a last resort, developers will be able to purchase biodiversity credits from the national government.

The mandatory BNG requirement is expected to deliver conservation benefits by providing a punitive-tax-like disincentive from harming biodiversity initially: developers will incur costs if their project inflicts damage on habitats ('internalising the externalities'; zu Ermgassen et al., 2020). Additionally, where developers are unable to meet biodiversity obligations themselves, the requirement to purchase 'biodiversity units' is viewed as an opportunity to stimulate private sector investment in nature regeneration. There are widespread hopes that this will create a market in 'biodiversity units', attracting private landholders into for-profit biodiversity unit generation (Defra 2019b).

However, the potential impacts of the mandatory BNG requirement have not been empirically evaluated. We collected all the BNG assessments accompanying planning applications submitted from January 2020-February 2021 (the Metric was essentially finalised in December 2019) in six councils who have adopted BNG-equivalent policies in advance of its national rollout (Supporting information; Table 22) into a new database. BNG assessments tend to be provided either as chapters within the proposed project's preliminary ecological appraisal, EcIA, or as standalone documents, and they contain as a minimum copies of the outputs of the Biodiversity Metric Excel tool (at best, they contain habitat plans and descriptions for the site at baseline and post-development). We identified appropriate councils via engagement with representatives from Defra, councils, and industry associations. The database is live, with more councils added when identified. In total, 16 potential councils were identified; but only the six councils included in our database have BNG-equivalent policies (Figure 15). We define these as BNG-equivalent as they all ask applicants to submit BNG assessments utilising the Metric alongside other planning information, and mandate that a net outcome-type target is achieved for each project (either NNL or 10% net gain) like the proposed national policy. We identified 90 projects referencing BNG assessments, of which 55 provided sufficient information for inclusion. We then removed one outlier project (a dwelling overseeing a 30ha estate implementing landscape-scale ecological restoration) as it was evidently not a policy-driven outcome, and six applications which were rejected by the planning authorities. Our sample spans 1000.3ha of development footprint, of which created or enhanced compensatory non-urban habitats comprise 468ha. The previous best academic estimate of England's entire implemented offset area was 53ha (Bull & Strange 2018), demonstrating the upscaling of ecological compensation represented by the mandatory requirement. By comparing the baseline and proposed future biodiversity assessments for developments in our sample, we explore which land cover changes are likely to be driven by BNG, what role off-site biodiversity offsetting will play, and their implications for conservation.

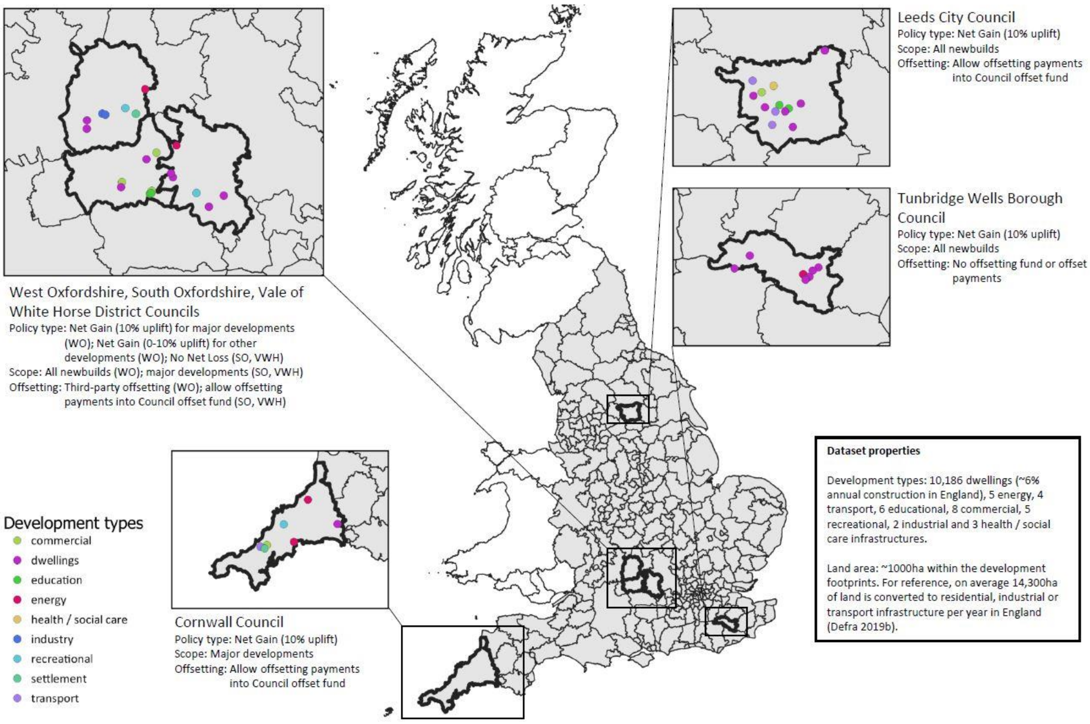

Figure 15. Summary of the BNG dataset, including the development types and locations and details of the six councils' BNG-equivalent policies.

6.4 Early signs that the biodiversity unit market may be smaller than expected

A first finding is that demand for biodiversity units delivered through offset funds or the biodiversity unit market in our sample is low (4.5% of total units); 95% of biodiversity units are to be delivered through the creation and enhancement of habitats within the development footprint or adjacent developer-owned compensation areas. This contrasts with the government's Net Gain impact assessment, which used a central estimate (based on anecdotal responses to the government's Net Gain consultation) for units purchased off-site of 25% (although they model scenarios including 0%; Defra, 2019b). The government has highlighted that developers paying for the off-site delivery of biodiversity units could be an important source of funding for investments within the Local Nature Recovery networks for each LPA (Defra, 2018, p9). The funding provided by these off-site payments might either be collected by the LPAs themselves and invested in a portfolio of biodiversity projects (e.g. enhancement of council-owned land; purchase of private land and its addition to the council's conservation estate) selected by the LPA, or collected by private brokers and invested in habitat banks. Our preliminary results raise doubts about the size of the biodiversity unit market. However, only five of our LPAs provide offsetting options, and the habitat creation market is still immature, so the desirability of purchasing biodiversity units may rise over time.

The number of purchased biodiversity units is low in our sample because 95% of the proposed biodiversity units will be delivered on land owned/managed by the developers. Ninety-one percent of units will be delivered via habitats within the direct development footprints (e.g. recreational grassland areas, tree and scrub establishment along hedgerows and site margins, some projects have dedicated ecological enhancement zones). Whilst small habitat patches within built environments can have ecological value, they are also threatened by high levels of human pressure. For example, 49% of the biodiversity units generated within residential developments in our sample come from on-site grasslands and scrub habitats, representing 27% of the total biodiversity units delivered in the dataset.

6.5 Biodiversity Net Gain will trade losses in habitat area today for promises of future gains in habitat quality

The dataset reveals a 34% reduction in the total area of open greenspace (defined as all non-urban habitats included within the Metric and excluding the units from as-yet-unspent offset funds), despite promising a 20.5% increase in biodiversity units across our sample. These losses in habitat area will be traded for habitats of higher distinctiveness and condition in the future (Figure 16). The pattern of change in habitats in our sample is consistent with a policy of 'trading up', with less distinctive habitats replaced by more distinctive habitats or higher condition levels. The true biodiversity impact of these trends is unclear. Intuitively, the loss of 34% of non-urban habitat area is likely to lead to a reduction of real-world biodiversity. However, improvements to the quality of habitats which increase the ecological resources available to wildlife relative to the baseline state could counteract this. The relative strength of these two factors should be further explored through field-validation of the Metric.

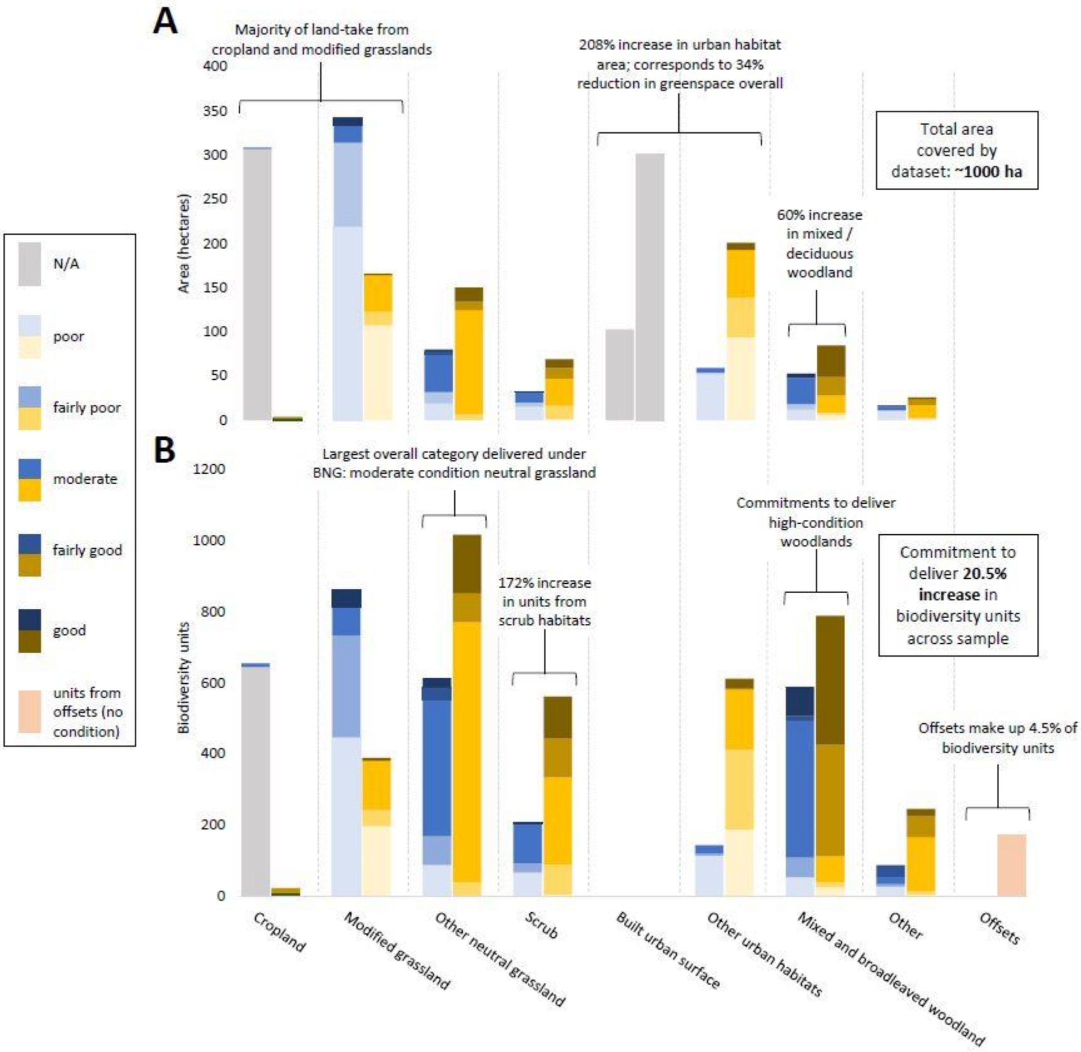

Figure 16. Aggregate ecological changes proposed in our sample of Biodiversity Net Gain assessments, by habitat type and habitat condition. Categories represent all of the relevant habitat types included in the Metric grouped together (e.g. 'scrub' contains the sum of 'mixed scrub', 'bramble scrub' and other related habitats included in the Metric), whilst 'modified grassland', and 'other neutral grassland' each represent a single habitat category in the Metric. A) total area devoted to different habitat types under the baseline (blue), and post-development scenario (yellow/brown). B) total number of biodiversity units delivered under the baseline (blue), and post-development scenario (yellow/brown). Annotations highlight key patterns in the dataset.

Our dataset demonstrates that mandatory BNG will trade biodiversity losses today for uncertain future gains, yielding a classic problem in the offsetting literature (Maron et al. 2012). It is widely recognised that compensating for losses today with promises of future biodiversity gains is risky (acknowledged in the Metric through restoration difficulty and temporal risk multipliers) as compensation measures are subject to implementation and restoration failure, and future political reversals (Maron et al. 2012; Bezombes et al. 2019; zu Ermgassen et al. 2019a). Therefore, conservationists typically prefer for compensation measures to be successfully implemented before associated biodiversity losses. These pre-development gains are commonly delivered through habitat banks. However, when these proposed gains are delivered on-site, they cannot usually be secured in advance of development; here it is essential that appropriate governance exists to ensure promises of future habitat improvements are delivered (discussed below; Damiens et al. 2021). This requires that proposed future gains are ecologically realistic and that modelling of gains is unbiased, so if the specified ecological measures are actually implemented, these gains are likely to be achieved in reality. Secondly, it relies on the appropriate governance being in place for incentivising and regulating real-world implementation.

6.6 How robust and open to bias are habitat condition assessments?

Like many EcIA processes, the Metric requires inputs based on subjective judgements of ecological consultants (although BNG guidance documents underpin these with some objective criteria to guide judgements). The Metric is most sensitive to the identification of habitat type (using the UK Habitats Classification system (https://ukhab.org/)), which determines the 'distinctiveness' score for each habitat, and its condition score. If there is substantial scope for error or bias in the Metric, then the number of units reportedly delivered through the BNG assessment process might be a poor reflection of their true ecological value. For example, under the baseline we find 342ha of modified grassland, a 'low distinctiveness' habitat (distinctiveness multiplier=2). If that same grassland were classified as 'other neutral grassland' ('medium distinctiveness', multiplier=4), all else equal, it would require compensation by double the area of post-development habitats. This highlights the importance of EcIAs (and BNG assessments) being undertaken by suitably-trained professionals, and subject to rigorous assessment by regulators. Leaving such an influential scope for judgements comes with risks, especially if ecological consultants lack sufficient training to conduct the relevant specialised ecological assessments (e.g. grassland assessments), or are implicitly pressurised to report a reduced biodiversity unit obligation by clients (Carver & Sullivan 2017).

To investigate whether the Metric is open to judgement-based variability, we surveyed seven expert grassland ecologists (supporting information). We provided them with all the publicly-available grassland survey information used in the baseline calculation associated with a sample of five BNG assessments (N=13 grassland patches), chosen to represent a range of survey qualities (Supporting information). We removed the final condition scores and habitat type classifications, and asked experts to propose the correct grassland type and condition score, given the information provided. Our specialised expert sample (which required expertise with a new condition assessment process and two habitat classification systems) is too small for statistical inference, but is indicative of whether experts broadly agree with judgements in BNG reports. Our expert sample agreed with both the habitat type and condition assessments 31% of the time, habitat type alone 42%, and condition alone 64% of the time. There was not universal agreement amongst experts regarding the grassland type for any grasslands in our survey (Supporting information), which indicates that less-specialised planners critiquing BNG assessments may find the habitat type and condition assessments challenging to scrutinise. Our survey findings indicate that boundaries between habitat categories are open to interpretation, and that the quality of information provided in BNG assessments is often insufficient to properly scrutinise.

6.7 Major governance gaps risk jeopardising the outcomes of Biodiversity Net Gain

To assess whether appropriate governance is in place to ensure the delivery of promised biodiversity units (a complex challenge that is often unrecognised; Damiens et al. 2021), we reviewed the governance mechanisms proposed in all BNG-related government, parliamentary and industry documentation, highlighting the key points relating to skills, capacity building, monitoring, enforcement, financial arrangements, and legal arrangements (Supporting information). The key finding is that, although there are ambitious commitments to monitoring and implementing offsetting measures delivered into the biodiversity unit market and via the government's stream of 'statutory' biodiversity credits, little attention has been paid to ensuring the delivery of habitats within developer-owned land. Nearly all additional governance mechanisms proposed are aimed at securing 4.5% of the biodiversity delivered through mandatory BNG (although this may rise on implementation of national mandatory BNG). Experience from NNL-type policies around the world shows that governance and implementation issues are essential drivers of their outcomes – often more important than policy-design parameters (Quétier et al. 2014b; Evans 2017; Samuel 2020).

The UK government has committed to resourcing mandatory BNG implementation and developing appropriate industry and regulator skills and capacity, which if implemented may address key problems highlighted in other NNL-type contexts (Quétier et al. 2014b; Samuel 2020). The government has committed to resourcing an additional 1.3 full-time-equivalent (FTE) employees for every higher-tier LPA in England (the largest spatial unit of local government, with 152 across England) to implement mandatory BNG (Defra, 2019b; although these commitments were made prior to the Covid-19 recession which has renewed the government's narrative regarding the need for fiscal prudence). This represents a large increase in capacity given approximately three-quarters of English LPAs currently have no in-house ecological expertise (ENDS report 2019). However, planning policy is often delivered by lower-tier authorities (25 of the higher-tier authorities across England covering >50% of England's land area are comprised of 188 'lower tier' authorities), and we found no formal commitments to increase their resourcing. There are concerns that most councils currently lack the ecological expertise to evaluate net gain assessments (Knight-Lenihan 2020). If unaddressed, this might lead to councils 'accepting' BNG assessments which are ecologically unrealistic (i.e. overpromise on biodiversity units). Additionally, the government commits to resourcing 59 FTE employees across Defra and Natural England to facilitate BNG implementation, focussing on the delivery and monitoring of off-site biodiversity units and local nature recovery networks. The Environment Bill also lays down a policy framework for the delivery of off-site biodiversity units.

However, the documentation reveals a gap with regards to biodiversity units delivered within developers' land. It suggests that existing planning enforcement without modifications is sufficient to secure developer-managed biodiversity delivery, although 'significant' on-site biodiversity gains will need to be secured through a 'suitable mechanism' (Defra, 2020, p179), which although not yet formalised could mean by conservation covenant or section 106 agreement. Given that 95% of biodiversity units in our sample are delivered through developer-managed land, this ambiguity and lack of commitment to enforcement creates risks. Compliance with on-site ecological mitigation and compensation measures in the UK is thought to be low (Drayson & Thompson 2013), yielding concerns that long-term ecological management measures may be insufficiently implemented. Most importantly, the current reactive nature of English planning enforcement is poorly suited to guaranteeing the delivery of high-quality habitats within approved developments. Councils can only take action against known planning violations, with little financing currently available for routine monitoring. Failures of habitat types to reach specified condition levels are unlikely to be reported by the public (although Defra emphasise they would like a transparent system for monitoring implementation of the mandatory requirement; Defra 2019a). Furthermore, the logistical challenges of how to monitor and enforce whether habitats have reached their promised condition levels given that each development is associated with multiple habitats which each 'mature' over different timescales have not yet been addressed (although we expect accelerating discussions about implementation issues as the national policy rollout draws closer). Industry best-practice guidance alludes to this issue by recommending that project proponents produce BNG Management and Monitoring Plans which outline the long-term management and monitoring timetables for their development operations. These should include commitments to adaptive management if monitoring demonstrates that the compensatory habitats are not on track to meet their commitments, and potentially performance-based payment schedules (i.e. so ecological subcontractors would be paid only once given objectives were achieved; Baker et al. 2019). However, potential problems remain: the slowest-maturing habitats in the Metric are assumed to reach their desired condition levels 32 years after project implementation, and assuming that councils will take enforcement action if those habitats fail to achieve their desired condition level decades after the project is constructed seems unrealistic.

Compounding this, even when planning violations are reported, local government guidelines outline that councils are encouraged to only take enforcement action in the case of 'serious harm to a local public amenity' (House of Commons Library 2019). The failure of a habitat to achieve the desired condition risks not satisfying this criterion, leaving them in essence unenforceable – identified as a key driver of failings of the Australian Environmental Protection and Biodiversity Conservation (EPBC) Act and French NNL policy (Quétier et al. 2014b; Evans 2017; Samuel 2020). Therefore, local authorities must rely on developers to implement the actions that are approved in their development applications; but if these actions include costly long-term management measures, they are implicitly incentivised to underinvest in ecological management with little or no oversight, risking long-term biodiversity outcomes.

6.8 Lessons for reconciling infrastructure expansion and biodiversity conservation

The mandatory BNG requirement will join a growing number of national NNL-type policies (zu Ermgassen et al. 2019b). The wide scope of development subject to mandatory BNG has the potential to make it a valuable template for other countries in the midst of international calls to change the functioning of our infrastructure systems in order to address ecological and climate emergencies (Thacker et al. 2019). However, this preliminary evaluation highlights that mandatory BNG as currently implemented at the local level risks poor outcomes for biodiversity when implemented nationally, unless key aspects receive additional attention. Many of these problems are paralleled by those in other biodiversity offsetting systems around the world (Table 9).

Offsetting region	**Problem**	**BNG susceptibility to problem – on-site**	**BNG susceptibility to problem – off-site**
Australia (national policy), France	Capacity shortfalls and inability to enforce lack of compliance (Quétier et al. 2014b; Evans 2017; Samuel 2020)	High susceptibility. Planning enforcement system poorly suited to incentivising compliance, although significant investment committed to improving capacity.	Low susceptibility if all proposed governance measures implemented. Conservation covenants (contracts to protect private land designated for offset sites) expected to come with monitoring schedules and enforcement mechanisms.
Queensland	Inability to find appropriate projects to spend offset funds to generate biodiversity gains. Of the AUD$9.6 million paid into Queensland's offset fund as of February 2019, only AUD$1.5 million had been committed or spent on offsets (Queensland Government 2019a)	-	High susceptibility. Landholders often unwilling to commit to covenants, especially if there is policy uncertainty.

France	Failure to implement compensatory habitats (Bezombes et al. 2019)	High susceptibility. Planning enforcement system poorly suited to incentivising compliance; compliance with ecological mitigation measures is in general imperfect (Drayson & Thompson 2013)	Low susceptibility if all proposed governance measures implemented. Government has proposed an offset register, reporting annually.
Western Australia	Site-level condition assessments are inaccurate and cannot be replicated by independent evaluators (Thorn et al. 2018)	High susceptibility. Expert survey shows information routinely provided in BNG assessments insufficient to eliminate judgement-based variation in condition assessments.	High susceptibility. Expert survey shows information routinely provided in BNG assessments insufficient to eliminate judgement-based variation in condition assessments.
England (offsetting pilots)	Power imbalances between regulators and developers allow developers to argue for cost-reductions to their proposed compensation measures (Carver & Sullivan 2017)	Unknown susceptibility. Power imbalances were shown to influence the outcomes of biodiversity assessments for the offset pilots; mandatory BNG aims to address this by making biodiversity gains mandatory rather than negotiable.	Unknown susceptibility. Power imbalances were shown to influence the outcomes of biodiversity assessments for the offset pilots; mandatory BNG aims to address this by making biodiversity gains mandatory rather than negotiable.
Canada; globally	Low offset multipliers are a key predictor of offset failure (Quigley & Harper 2006a; zu Ermgassen et al. 2019a)	High susceptibility. BNG found to be delivering 34% loss of greenspace area, which if unaccompanied by significant improvements in vegetation condition post-development will lead to a loss of biodiversity.	High susceptibility. BNG found to be delivering 34% loss of greenspace area, which if unaccompanied by significant improvements in vegetation condition post-development will lead to a loss of biodiversity.

Table 9. Problems with compensatory mitigation systems around the world, and the degree to which proposed governance measures for the implementation of the mandatory BNG requirement address these problems

Firstly, it is essential that the appropriate governance measures are in place if the policy is to continue to trade immediate biodiversity losses for uncertain future gains (Damiens et al. 2021); temporal multipliers cannot be relied upon alone (Bull et al. 2017b). The governance of biodiversity units delivered through habitat banking and offsetting have received much attention. But if the majority of biodiversity units are likely to be delivered on site, current planning system mechanisms for monitoring and enforcing compliance are poorly suited for ensuring these materialise in reality.

Secondly, although the responses to the government consultation found broad support from across stakeholders for the majority of biodiversity units being delivered on-site (Defra 2019a), our study suggests this urgently deserves further debate. Our dataset is associated with a 34% loss in open greenspace, coupled with indications that the total level of funding generated through mandatory BNG for off-site, strategic investments in the Local Nature Recovery Networks may be small. Biodiversity enhancements delivered within development footprints risk not materialising in reality, because of governance issues, and these locations being subject to high levels of human pressure and disturbance. Therefore as currently implemented, mandatory BNG risks not only delivering little for biodiversity, but also missing a major opportunity to finance investments in regional biodiversity priorities that can help restore biodiversity at a landscape scale. These risks could be addressed by potentially incentivising the delivery of biodiversity off-site, such as through mandating that a certain percentage of the total biodiversity units delivered by a project must be invested in off-site regional biodiversity priorities or the Local Nature Recovery Network. Another mechanism might be capping how much urban land take is permitted by the policy. When the Metric was first designed, the authors recommended a 1:1 minimum area be established, so that a loss of habitat area could not solely be compensated for through promises of future condition increases (Treweek et al. 2010). On the other hand, a mandatory area target might disincentivise delivering higher condition habitats. It is also worth recognising that a key policy aim of mandatory BNG is improving peoples' access to greenspace (Defra 2019b), which can be used to justify on-site biodiversity enhancements being prioritised. However, this priority risks overwhelming the biodiversity goals of the policy, and potential trade-offs should be explicitly discussed.

Lastly, our study provides yet further evidence that designing governance mechanisms for reconciling infrastructure expansion with biodiversity conservation is deeply challenging. Even ambitious policies are subject to huge uncertainties that risk undermining their biodiversity benefits. The safest mechanism for reducing the biodiversity impact of infrastructure is to avoid impacts to biodiversity initially. In practice, this means redirecting development to previously degraded sites wherever possible. On a deeper level, given the need to transition to an economy that meets the needs of all within the constraints of the Earth system (O'Neill et al. 2018), we must rethink our bias towards finding environmentally-damaging hard infrastructural solutions to societal challenges.

Chapter 7 A home for all within planetary boundaries: exploring pathways for meeting England's housing needs without transgressing national carbon and biodiversity targets

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7.1 Abstract

Secure housing is core to the Sustainable Development Goals and a fundamental human right. However, potential conflicts between housing and sustainability objectives remain under-researched. We explore the impact of current English government housing policy, and alternative housing strategies, on national carbon and biodiversity goals. Using material flow and land use change/biodiversity models, we estimate from 2022-2050 under current policy housing alone would consume 104% of England's cumulative carbon budget (2.6/2.5Gt [50% chance of <1.5°C]); 12% from the construction and operation of newbuilds and 92% from the existing stock. Housing expansion also potentially conflicts with England's biodiversity targets. However, meeting greater housing need without rapid housing expansion is theoretically possible. We review solutions including improving affordability by reducing demand for homes as financial assets, macroprudential policy, expanding social housing, and reducing underutilisation of floor-space. Transitioning to housing strategies which slow housing expansion and accelerate low-carbon retrofits would achieve lower emissions, but we show that they face an unfavourable political economy and structural economic barriers.

Keywords: infrastructure sustainability; biodiversity net gain; material flow analysis; growth-dependence; net zero; financialization of housing

7.2 Housing infrastructure and the Sustainable Development Goals

The Sustainable Development Goals (SDGs) outline humanity's aspirations for achieving high living standards for all without harming nature or modifying the climate system. However, unless the environmental impacts of economic expansion fall at a rate considerably faster than at any point in human history over the coming decades (Hickel & Kallis 2019; Jackson & Victor 2019), then there will be trade-offs between the environmental and economic objectives of the SDGs (Spaiser et al. 2017; Hickel 2019). One such potential trade-off is that between built infrastructure expansion (underpinning multiple SDGs; Thacker et al. 2019) and climate (SDG 13; Müller et al. 2013) and biodiversity objectives (SDGs 14&15; zu Ermgassen et al. 2019). By 2060, an estimated >230 billion m^2 of additional built floor area will be added to the global building stock, equivalent to the built area of Japan each year (UNEP & IEA 2017). The ecological impacts of this unprecedented infrastructure expansion will be profound (Müller et al. 2013; Laurance et al. 2015).

Navigating trade-offs between nature and infrastructure construction is a grand challenge – we need enough infrastructure to meet the transportation, communication, health, energy, production and housing needs of all, but excess infrastructure risks failing to address human needs whilst inflicting damage that threatens the integrity of the Earth system (O'Neill et al. 2018; Brand-Correa et al. 2020; Fanning et al. 2021). Haberl et al. (2019) show that for societies below a threshold of approximately 50 tonnes of concrete per capita (concrete represents 45% of global material stocks by mass), there is coupling between increasing infrastructure and human wellbeing, measured by the social progress index (SPI) (see also Donaldson 2018; Thacker et al. 2019). However, above this threshold, the relationship dissolves. The starkest example is that New Zealand achieves a higher SPI than the Czech Republic with approximately 20% of the per-capita material stocks (Haberl et al. 2019b).

The thorniest problems contain the potential for direct trade-offs between societal priorities – such as between meeting fundamental human needs and remaining within the planet's 'safe operating space' (Fanning & O'Neill 2019). Housing infrastructure represents such a challenge: housing is recognised as a fundamental human right, and in commonly-used needs-based conceptualisations of wellbeing formalised by Doyal & Gough (1984), Max-Neef (1991) and Rao & Min (2018). Housing expansion to address unmet basic human needs is clearly essential. Yet, processes linked to housing provision are, under current production technologies, powerful drivers of both biodiversity loss and climate change. Twenty-four percent of all threatened species on the IUCN Red List are threatened by commercial and residential infrastructure expansion (https://www.iucnredlist.org/), and yet more by construction mineral supply chains (Torres et al. 2021, 2022). Infrastructure's climate impacts come from the greenhouse gas emissions embedded in the production, operation and maintenance of infrastructure - emissions from housing and construction contribute approximately 27% of all annual global carbon dioxide emissions (UNEP 2020).

However, infrastructure and housing construction remain core economic sectors in most advanced economies. Whilst often justified on the grounds of affordability, employment, or providing enabling conditions for increasing productivity (Thacker et al. 2019), the lack of an obvious macro-level wellbeing-infrastructure stock relationship in infrastructure-abundant economies suggests other factors might also help explain the economic salience of specific infrastructure classes. For example, Mattioli et al. (2020) explore the political economy of road infrastructure and car-dependence, and identify a range of socio-political dynamics that lock society into a high car use, high ecological consumption pathway. Political-economic factors might also play an important role in other infrastructure sectors, such as housing, and help partially explain infrastructure proliferation even in cases where the social benefits are unclear.

7.2.1 England's housing and sustainability policy context

This paper explores these issues in the context of England's housing affordability crisis: England represents a particularly salient case study, as it simultaneously has abundant housing stock, unmet housing need, and legally-binding environmental policy goals reflecting national contributions to addressing key planetary boundaries (Steffen et al. 2015b). England has under-occupied housing stock (see Section 2; Mulheirn 2019), but one recent estimate suggests up to 7.9 million people currently experience some symptoms of unmet housing needs (National Housing Federation 2020); predominantly because England has one of the highest rates of housing unaffordability (Downie et al. 2018; National Housing Federation 2020). Additionally, the country's population is still growing. The government's policy response is to build more housing, having committed to supplying 300,000 new homes per year by the mid-2020s (Wilson & Barton 2021).

However, there is limited discussion of the ecological implications of this strategy in policy reports. On the climate side, the government has committed to net zero by 2050, and England's cumulative carbon budget from 2022-2050, compatible with a 50% chance of staying below 1.5°C, is approximately 2.5GtCO_2e (5GtCO_2e is the remaining carbon budget for England implied by the government's Net Zero strategy; Jackson 2021; Supporting information).

The dominant approach to housing sustainability in English policy reports is on reducing the ecological impacts of the existing and future housing stock whilst taking rapid housebuilding rates as given. The overwhelming focus in official government documentation regarding the housing affordability crisis is on building more homes (DCLG 2017; MHCLG 2021a; OECD 2021b). However, home energy and electricity use represents one-fifth of total emissions (CCC 2019, p11). Extensive analyses have demonstrated how to achieve net zero operational emissions across the buildings and residential sector, including retrofitting the existing stock (CCC 2019, 2020a; RICS 2020; EAC 2021; NEF 2021). Policy mechanisms have been proposed to accelerate uptake of energy-saving domestic innovations (e.g. 'green offsets'; 'green land value tax' (Muellbauer 2018; Cheshire & Hilber 2021)). Notably, shifts towards more equitable consumption of floor space/capita are not mentioned in government strategy, despite having been empirically identified as essential to achieving decarbonisation targets (Serrenho et al. 2019; Hertwich et al. 2020; Pauliuk et al. 2021).

However, there have been no reductions in annual emissions from buildings observed since 2015 (Committee on Climate Change 2020, p110). Fifty-four percent of all homes in England have energy performance certificate (EPC) ratings of D or worse, and the Committee on Climate Change recommends all homes exceed this standard by 2028 (EHS 2021). Nearly all require retrofitting to be consistent with the 2050 Net Zero target (EAC 2021). For newbuilds, the percentage possessing an EPC band 'A' has varied between 1-1.5% each year from 2014-2020 (MHCLG 2021b). Homes constructed today which are not compliant with 2050's net zero goal will have to be retrofitted at potentially prohibitively high future cost (Serrenho et al. 2019).

On the biodiversity side, the 2021 Environment Act commits the government to implementing a legally-binding target to halt wildlife declines nationally by 2030, and from late 2023 will mandate that all new developments achieve a 'Biodiversity Net Gain'. Biodiversity Net Gain aims to resolve trade-offs between new construction and impacts on nature. The policy will mandate that all new developments leave biodiversity better off than they found it, as measured using the Biodiversity Metric, a simple habitat-based biodiversity indicator (zu Ermgassen et al. 2021b). However, recent empirical work has demonstrated that the policy's impacts on biodiversity remain ambiguous – planning applications achieving 'net gain' in a set of early-adopter councils were associated with a 34% reduction in the area of greenspace despite claiming a 20% improvement in biodiversity overall, and major governance gaps were identified, risking the successful delivery of these promised compensatory biodiversity improvements (zu Ermgassen et al. 2021b). Given uncertainty about Biodiversity Net Gain's effectiveness, preventing unnecessary land use change consistent with the mitigation hierarchy remains essential (Phalan et al. 2018; Bull et al. 2022).

Whilst supply-side sustainability measures are essential, policy focusing solely on operational impacts might signal that housing proliferation can continue without trading-off against environmental policy objectives or compounding existing decarbonisation challenges in the sector. However, housing proliferation is associated with unavoidable material impacts, including embodied carbon emissions in construction, and urban land take affecting biodiversity. The construction of poor quality housing today also induces ‘lock-in’ effects, passing additional decarbonisation costs into the future (Serrenho et al. 2019).

7.2.2 Rationale

In this paper, we explore whether the English government’s expansionary housing policies effectively address unmet housing need, and their compatibility with national biodiversity and decarbonisation goals. We review the political economy of England’s current policy response, and outline alternative pathways to meeting England’s housing needs without undermining national sustainability objectives. This study therefore implicitly explores solutions for simultaneously achieving infrastructure and housing-related SDGs (9, 11), and ecological SDGs (13, 15). To our knowledge we are the first to simultaneously estimate the biodiversity and carbon impacts of housing expansion in England, present the emissions of housing relative to England’s cumulative carbon budget, and investigate the sustainability implications of alternative strategies for addressing the housing affordability crisis and supply-side/demand-side debates about housing affordability. Reducing the operational emissions of existing housing is already recognised as one of the largest challenges in the UK’s decarbonisation strategy (CCC 2019; Serrenho et al. 2019; RICS 2020; EAC 2021; NEF 2021). However, emissions from new housebuilding are still a substantial contributor (Drewniok et al. 2022b), and they have received much less attention. We therefore begin to fill the gap in research around the potential impacts of reducing housebuilding and the political economic barriers and solutions.

The paper is organised as follows. Section 2 reviews the causes of the housing unaffordability crisis, reviewing evidence suggesting that simply expanding housing supply may not address key ultimate drivers of unmet housing need. Section 3 presents our novel analyses of the carbon and biodiversity impacts of alternative strategies for the English housing stock. Section 4 summarises the political economy of housing expansion in England, identifying 'growth-dependencies', unrelated to England's fundamental housing needs, that make its economy structurally dependent on housing expansion. Section 5 proposes policies for addressing unmet housing need whilst minimising conflicts with national sustainability policies. Section 6 concludes.

7.3 The causes of housing unaffordability

Understanding the true drivers of housing unaffordability is key to identifying solutions that can increase housing need satisfaction without substantially increasing the housing sector's emissions and ecological impacts (i.e. improving the ecological efficiency of the housing 'provisioning system'; Fanning et al. 2020). England's housing affordability crisis is characterised by rising average prices relative to incomes, falling rates of homeownership matched by rising levels of renting, homelessness, and general housing inequality spanning both housing space/capita and the socio-economic and demographic distribution of housing wealth (Tunstall 2015; Arundel 2017; Ryan-Collins et al. 2017; Gallent 2019). Across England and Wales, the ratio of median house price to median gross annual earnings has risen from an average of just below 4:1 in 1998 to almost 8:1 by 2020, with London reaching 12:1 (ONS 2021a). This has priced out younger and lower-income cohorts from the housing market in many of the cities where jobs are created. However, at the same time, the consumption of housing space has been rising, from 35.2m^2/capita in 1996 to 41.1m^2/capita in 2020 (Serrenho et al. 2019; EHS 2020a, Annex).

Whilst average housing rents have largely tracked incomes, the housing cost to income ratio, which incorporates all housing costs and compares these to post-tax incomes on annual basis, has risen from around 10% in the early 1980s to 35% now for private renters, with similar dynamics for housing associations (Resolution Foundation 2017). This has been driven by the liberalisation and privatisation of the rental market and declines in housing benefit, coupled with stagnating wages for renters, most of whom occupy the bottom half of the income distribution (Coulter 2017).

7.3.1 Supply-side explanations

In UK policy-circles, explanations of the affordability crisis are dominated by supply-side explanations. Multiple major reviews of the UK's housing market have concluded the reason for high prices is due to inadequate provision of new homes relative to rising demand (Lyons 2014; DCLG 2017; Wilson & Barton 2021).Both major political parties have emphasized the supply-side, with in-power Conservatives placing more emphasis on reforming an inefficient planning system (MHCLG 2021a), and Labour on building more social housing (Labour Party 2019). Other solutions include penalising developers for holding undeveloped land with planning permission secured (representing approximately 1 million unbuilt homes in the UK; Local Government Association 2021), and encouraging innovation within the sector (DCLG 2017). A substantial body of academic research also emphasizes supply-side explanations (Brown & Glanz 2018; Cheshire 2018).

However, a body of empirical evidence casts doubt on solely supply-side explanations. On planning, approximately 90% of planning applications in the UK are approved (MHCLG 2021c). Government household and housing stock data show that the UK has a surplus of dwellings relative to households (Figure 17). This surplus has grown from 660,000-1.23 million homes from 1996-2019 (Mulheirn 2019). This pattern is consistent across the country: for example, London has a higher proportion of surplus dwellings than the national average. In recent years the number of new households has been consistently outstripped by additions to the housing stock (ibid).

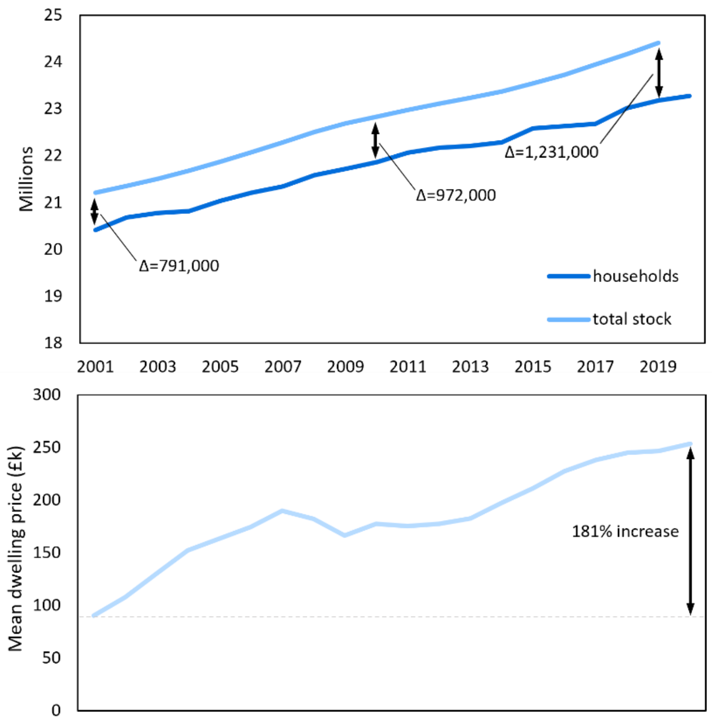

Figure 17. Comparison of the total number of dwellings and households in England, and changes in mean dwelling prices, from 2001-2020. Top panel: Numbers of households (2001-2015 data from ONS 2018, Figure 10, "household estimates"; 2015-2020 data from ONS 2021b, Table 5, England), dwellings (MHCLG 2021c, Table 104), and surplus dwellings (households - dwellings) in England. Bottom panel: annual mean dwelling price in England (HMLR 2022)

Even if there are housing supply constraints, evidence suggests that expansion of the housing stock may have a limited effect on housing affordability. Estimates of the sensitivity of UK house prices to increases in housing stock consistently show that a 1% increase in housing stock per household delivers a 1-2% reduction in house prices (Auterson 2014; Oxford Economics 2016; MHCLG 2018). This is minimal in the context of a 181% increase in mean English house prices from 2000-2020 (£84,620-£253,561; HMLR 2022).

Beyond the question of general housing shortages, it is more universally agreed that there are shortages in social housing which targets the needs of those struggling to afford market-rate homes or rents. Government-led construction of social housing was central to UK post-war social policy, with local authorities constructing the majority of housing in the 50s, peaking at 155,000 new homes in 1967, before declining in the 70s and 80s as the government ended the New Towns programme and various legal judgements increased the cost of state-led compulsory purchase of land for housebuilding (Ryan-Collins et al. 2017; Wilson 2021). Social housing stocks were sharply reduced by the Thatcher government's 'right-to-buy' policy which facilitated the discounted transfer of approximately 2 million properties from the state to private owners, 40% of which are now estimated to be on the private rental market (Inside Housing 2017; Christophers 2020). Recent estimates suggest there is currently a need for an additional 1.6 million dwellings at social rent (National Housing Federation 2020).

7.3.2 Demand-side explanations

Demand-side perspectives on house price unaffordability emphasise the interaction of multiple complex processes that cause ever-increasing capital to flow into the housing sector, competing for finite supply (Gallent et al. 2017, 2018; Ryan-Collins 2018; Kazi & MacFarlane 2022). A key observation is housing has multiple functions: it is both a consumption good and a means of accruing wealth. Evidence suggests the demand for both types of use has increased over time and would appear to provide more explanatory power in understanding rising house prices than supply.

Considering consumption demand first, UK housing and land has a high income-elasticity of demand - as incomes rise households spend more of their income on housing relative to other goods (Cheshire & Sheppard 1998). One estimate across two UK cities found that a 10% increase in incomes leads people to spend about 20% more on space in houses and gardens, with homeowners having a higher income elasticity of demand than renters (ibid). As mentioned above, high-level data shows that as incomes have risen, households in England have on average been occupying more space over the last 25 years (35.2m^2/person-41.1m^2/person from 1996-2020; Serrenho et al. 2019)). A recent long run model of UK house prices found consumption demand driven by rising incomes to be the most important single factor (Meen & Whitehead 2020).

Other studies have pointed to the effects of low real interest rates in increasing the demand for housing as a financial asset (Miles & Monro 2019; Mulheirn 2019), whilst others have pointed to weakening credit constraints as the 'elephant in the room' in explaining rising house prices (Aron et al. 2012; Ryan-Collins 2018; Bezemer et al. 2021). The liberalisation of mortgage finance in wealthy economies since the 1980s, coupled with financial innovations such as securitisation encouraging institutional investors to enter the housing market, have led to enormous increases in capital flowing in to the housing sector, competing for a finite supply of desirably located residential land, with inevitably inflationary consequences (Aalbers 2017; Gallent et al. 2017, 2018; Ryan-Collins 2018; Blakeley 2021).

In the UK, financial deregulation and liberalisation supported an increase in UK-based banks' credit creation for mortgage lending from around 15% of GDP in 1980 to 60% by 2008 whilst lending to businesses increased from 10% to just 30% (figures remain similar in 2020; Bezemer et al. 2021). Whilst with most commodities, rising prices will lead to falling demand, rising house prices relative to income create more demand for mortgage credit, whilst real estate's attractiveness as a form of collateral (being difficult to hide and increasing in value) gives banks confidence to continue to meet this demand. This creates a positive feedback loop or "housing-finance cycle" (Ryan-Collins 2021) which can be hard to break out of without repercussions for financial stability and the wider economy. These dynamics also exacerbate housing inequality as purchasers with existing housing collateral can secure additional mortgage loans at lower interest rates, out-competing first time buyers for new property that comes on to the market. In doing so, the effect is to push up prices of housing beyond that which it may have reached had only owner-occupiers been competing.

The attractiveness of land and housing as financial assets have fuelled a rise in foreign investment in the UK property sector. Between 2014-2016, 13% of all homes purchased in London were bought by overseas investors, and around half of these were of housing valued at <£0.5m (Wallace et al. 2017). Between 2009-2015 complex corporate structures mostly registered in offshore tax havens purchased nearly 28,000 London properties and land parcels at an estimated value of £100 billion (Crerar & Prynn 2015). A recent investigation found that the number of dwellings with owners registered abroad has tripled from 2010-2021, representing nearly 1% of England and Wales' entire dwelling stock (Clarence-Smith 2021).

Government policies have contributed to these dynamics, with a general shift in policy away from subsidizing the creation of the housing stock towards subsidizing the demand for homeownership and private renting. Homeownership as an asset class receives favourable tax treatment, notably with the 1963 abolishment of imputed rent and capital gains tax exemptions for primary residences (Oxley & Haffner 2010; Ryan-Collins et al. 2017). A range of mortgage subsidies have been introduced over the years, including the ability to offset taxation against interest payments on investment properties (abolished in the early 2000s) and a range of schemes supporting first time buyers. Recent evidence suggests these latter schemes had the perverse effect of increasing house prices as the increasing demand was capitalised into prices (Carozzi et al. 2019).

Additionally, government policy has created incentives for the purchase of second homes as investment properties. Most notably, the 1988 Housing Act made private renting more attractive for investors by strengthening landlords' grounds for repossession, abolishing fair rent appeals and reducing the minimum notice period of eviction from one year to six months (Leyshon & French 2009). The latent demand for second homes was realised in 1996 with the introduction of 'buy-to-let' (BTL) mortgages, which led to a flood of new credit into the housing market. By 2008, BTL made up 11% of total mortgage advances (ibid).

Rising rents have also led to huge increases in housing benefit being paid out to lower-income renters, which amounts to a significant government subsidy for landlords (housing benefit was estimated to cost the government £23.4 billion, 3% of the national budget, in 2019) (Ryan-Collins et al. 2017; Office for Budget Responsibility 2018; Christophers 2020). Since the vast majority of landlords come from the top 20% of the income distribution (Christophers 2020), these dynamics further increase housing and earnings inequality.

In summary, this exploration of the drivers of housing unaffordability suggests the problem may be less with the total supply of housing units and more with their distribution across the population and 'overconsumption' by wealthier groups, enabled by rising incomes and easy credit conditions. Policy reforms that could dampen the demand for housing beyond a basic level of need could theoretically enable the UK housing system to satisfy greater housing need without relying on rapid housing expansion. This is welcome, as a solely supply-side explanation would imply that the only way to satisfy more housing need is through housing expansion, despite the inherent environmental impacts. Next, we explore the ecological impacts of expansionary housing policies, and compare them with alternative pathways for meeting housing needs.

7.4 The environmental impacts of housing proliferation

7.4.1 Potential baseline biodiversity impacts of housing expansion without policy action

How much housing expansion in England will conflict with the 2030 species abundance target is currently unknowable, as Biodiversity Net Gain will first be introduced in late 2023 and its effectiveness is unproven, and no models yet exist for predicting changes in species abundance in response to land use changes in England. Our simple approach here is to draw on existing models estimating changes in species richness (as a proxy) from land use change, and predicted housing expansion, to generate a high-level estimate of the biodiversity impacts of predicted housing expansion without policy action, which can roughly represent how effective Biodiversity Net Gain and species mitigation policies will need to be to halt biodiversity loss from housing expansion from now-2030. This land use change model does not include the land take associated with biodiversity offsets purchased off-site to achieve developments' biodiversity net gain commitments (i.e. it implicitly assumes that all biodiversity units will be delivered on-site). This is justifiable on the grounds of zu Ermgassen et al. (2021) who identified in their sample of developments achieving net gain that the vast majority (93%) of biodiversity units were delivered on-site; although we recognise the government's own market analysis suggests up to 50% of units may be delivered off-site (eftec 2021). In Section 5 we then draw on results from recent evaluations of Biodiversity Net Gain and species mitigation measures to discuss improvements required to deliver this aim (zu Ermgassen et al. 2021b; Hunter et al. 2021).

We use the spatial projections for urban expansion in England from 2006-2031 from Eigenbrod et al. (2011), and input these into the biodiversity module of the Natural Environment Valuation modelling suite (Binner et al. 2019). The biodiversity module represents an ensemble of species distribution models which give the probability of species presence in each 2km grid cell across England for 100 species of conservation priority, given the land use in that cell (Wright et al. 2019). For each cell, the probability of species occurrence under the chosen land use is then summed for all species, and this can be compared with the baseline land use to estimate the effect of housing development on richness of species important to conservation (Supporting information).

The model estimates that 12,519ha of farmland will be lost per year from 2006-2030 to urban development in the UK under the assumption of constant housing densities over time (Eigenbrod et al. 2011; Figure 18), which is roughly the same as the mean conversion of agriculture and undeveloped land to developed land in England from 2013-2018 (13,956ha; Ministry of Housing, Communities & Local Government 2020a, Table P350). This equates to an average loss of biodiversity of approximately 0.04 species per hectare or an average 5.7% loss in species richness in the areas being developed (Figure 18). For housing expansion not to conflict with England's 2030 wildlife abundance targets, Biodiversity Net Gain and species mitigation policies will have to fully compensate for these losses, or the rate of land take for housing could be reduced to avoid these impacts on biodiversity initially.

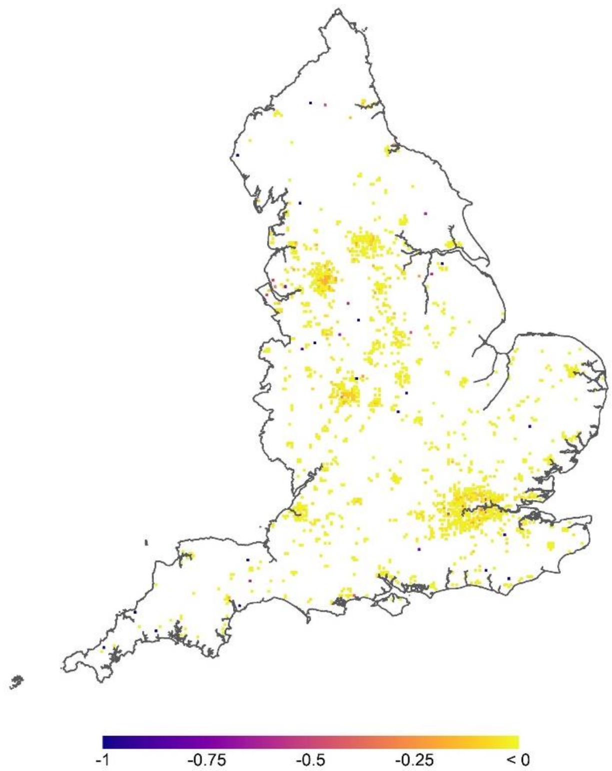

Figure 18. Estimated impact of urbanisation on biodiversity, measured as species richness per hectare for 100 species of conservation priority in England.

7.4.2 Potential carbon impacts of housing expansion

To estimate emissions, we use two recently-developed models and reparameterise them to reflect current data and alternative scenarios for housing in England: a high-resolution material flow analysis estimating the embodied carbon in housing construction developed by Drewniok et al. (2022b, 2022a), and the operational housing emissions model developed in Serrenho et al. (2019).

Drewniok et al. (2022a) estimate the amount and type of materials used in the production of new dwellings by combining information about the proportion of different dwelling types from the English Housing Survey (EHS 2020b) with case study archetypes for each dwelling type identified from letting agency or developers' websites. For each case study, information about the layout is used to estimate the dimensions for substructure, structure, roof, partitions, cladding, wall and ceiling finishes, windows and doors. The analysis excludes insulation and fixtures and fittings. For each building element, the most common technologies are estimated based on information from the English Housing Survey (EHS 2020b) and NHBC standards (nhbc-standards.co.uk), and the material intensities for the different technologies are modelled based on NHBC standards. For elements that use a mix of technologies throughout the building stock, the share of alternative technologies is estimated through discussions with industry partners. The material quantities for each dwelling typology are then normalised by gross internal floor area. Similar methods are used to determine the material quantities for the conversion of non-domestic to domestic buildings (which makes up approximately 8% of net additions to the housing stock), except it is assumed that the foundations and upper floors are reused (i.e. unassociated with embodied emissions), and it is assumed that 50% of the remaining building structure is reused. The total volume of materials required includes a small wastage rate, consistent with current building practice.

To estimate the emissions embodied in all of these materials, Drewniok et al. (2022a) use life cycle assessment methods consistent with British standards (BSI 2011). Their analyses include the emissions associated with the materials and construction process up to practical completion, which represents approximately 70% of the whole life embodied emissions for residential buildings (Gibbons & Orr 2020). Carbon coefficients for each material are taken from the Inventory of Carbon and Energy (ICE 2019), and for materials not listed in the inventory, values are taken from their Environmental Product Declarations. Transport emissions are estimated based on the average emissions of road freight. The model produces estimates of embodied emissions of housing which are consistent with other results reported in the literature, including those calculated by alternative methodologies (e.g. Steele et al. 2015).

To estimate the operational emissions of the housing stock we use the operational emissions model developed in Serrenho et al. (2019). They estimate the operational emissions of the existing stock by, firstly, identifying the Environmental Impact Rating (EIR) and floor area for all England's dwellings (with the year 2018 as a baseline) using information from the English Housing Survey (EHS 2020c). They then use the government's standard method for translating dwellings' EIR into annual emissions using the equation (DECC 2014):

$$O = \begin{cases} (A+45) * 10\left(\frac{40}{19} - \frac{EIR}{95}\right), if \ \frac{O}{A+45} \geq 28.3 \\ (A+45) * \left(\frac{100 - EIR}{1.34}\right), if \ \frac{O}{A+45} < 28.3 \end{cases}$$

where O represents the annual emissions in kg CO_2 and A represents each dwelling's floor area in m^2.

Both models then enable testing the emissions associated with various scenarios for the future of the housing stock. The Drewniok et al. (2022b) model estimates future embodied emissions associated with different housebuilding rates. The types of housing being added to the stock each year is assumed to reflect the distribution across different housing types under the baseline year. The model includes industry's own projections for the decarbonisation of production of different building materials from technological innovation (e.g. assumes 36% decarbonisation of concrete production by 2050 in line with the industry's net zero roadmap; GCCA 2021), but discounts the use of negative emissions technology as it is unproven as scale. The model allows the user to vary multiple inputs, such as the degree of material decarbonisation over time experienced by various building materials or the demolition rate, but for the sake of simplicity interpreting the results of the differences between our housing scenarios, we maintain nearly all inputs constant across all of our scenarios.

The Serrenho et al. (2019) model takes the baseline operational emissions of the existing stock and of newbuilds in 2018, and then simulates a linear rate of decarbonisation of both types of housing under varying assumptions about the time to decarbonisation, and the total degree of decarbonisation, for both housing classes (Supporting information). We update the original Serrenho et al. (2019) model by adopting the 2021 demolition rate as used in Drewniok et al. (2022b).

We simulate three scenarios for the future of the housing stock (Table 10) that correspond to alternative strategies for meeting England's housing needs from 2022-

2050 (Figure 19). Scenario 1 represents the government's current housing strategy. Scenario 2 represents a highly ambitious supply-side greening strategy where the rate of housebuilding remains aligned with government expansion targets, but new home standards are introduced so all newbuilds are zero carbon from 2035 and the existing stock is retrofitted so that it is as efficient as contemporary newbuilds (i.e. newbuilds constructed in 2018 as in Serrenho et al. 2019) by 2035. Scenario 3 implements the same ambitious roadmap for decarbonising newbuilds but coupled with extremely ambitious decarbonisation of the existing stock (so the existing stock achieves zero emissions by 2050) and more efficient use of housing space to reduce the need for new housing construction and the associated embodied emissions (to zero net additions by 2035). All scenarios are policy-relevant (i.e. derived from the government's Net Zero strategy or other policy reports; Table 10).

Key assumptions	Business as usual	Supply-side greening	Strong sustainability
Housing construction rates	300,000 net additions per year to 2050[1]	300,000 net additions per year to 2050	Linear decrease from today's level to zero net additions by 2035
Unoccupied housing	Current level	Current level	No vacant homes; fully occupied
Time to decarbonisation of new housing	2050[2]	2035[3]	2035
Retrofit rate	Halve operational emissions of the existing housing stock by 2050[4]	Retrofit all to 2018 standards by 2035[5]	Retrofit all to zero carbon by 2050[2]

Table 10. Simulated scenarios for the future of the housing stock. We hold a range of assumptions constant across all three scenarios to improve comparability, such as material decarbonisation rates, housing typology, rate of conversion of non-domestic to domestic buildings (Supporting information). The policy justifications for the assumptions we vary are: 1) Government's existing housebuilding target (Wilson & Barton 2021). 2) Consistent with Net Zero strategy goal "ensure that all homes meet a net zero minimum energy performance standard before 2050, where cost effective, practical, and affordable." (BEIS 2021a). 3) Consistent with Net Zero strategy goal "We will introduce regulations from 2025 through the Future Homes Standard to ensure all new homes in England are ready for net zero by having a high standard of energy efficiency and low carbon heating installed as standard." Note that net zero ready does not mean zero carbon, but able to be retrofitted to achieve zero carbon in the future, hence the 2035 target date. 4) Linear extrapolation of the decarbonisation rate of the emissions from homes from 1990-2019. This extrapolation considerably exceeds recent decarbonisation trends, as there has was no decarbonisation in domestic emissions from 2014-2019 (BEIS 2021, Table 1.2; Supporting information). 5) Consistent with Net Zero strategy goal of "Consulting on phasing in higher minimum performance standards to ensure all homes meet EPC Band C by 2035, where cost-effective, practical and affordable." (BEIS 2021a)

Under Scenario 1, the housing stock consumes 104% of England's cumulative carbon budget consistent with a 50% probability of remaining within 1.5°C of heating by 2050, or 52% of the cumulative carbon budget of the government's balanced net zero pathway (Figure 19). Ninety-two percent of emissions come from the existing stock, and 9% is embodied in the construction of new housing. The operational and embodied emissions of new housing consume 12% of the cumulative carbon budget for 1.5°C. Scenario 2 consumes 70% and 35%, and Scenario 3 60% and 30%, of the 1.5°C and government carbon budgets respectively.

By far the most impactful policy for reducing housing's conflict with climate targets (but not biodiversity) is rapid retrofitting of the existing stock – retrofitting all homes to emissions standards of today's newbuilds by 2035 could avoid 0.8GtCO_2e, equivalent to 32% of the cumulative carbon budget for 1.5°C. Going even further and decarbonising the existing stock entirely by 2050 could save 38% of the budget for 1.5°C.

However, slowing the rate of housebuilding and improving the standards of new construction can also play a key role, especially when we consider later government carbon budgets (the government agrees national carbon budgets for 5-year periods; e.g. the UK's 'sixth' carbon budget from 2033-2037 has been set at 965MtCO_2e, approximately 827MtCO_2e for England alone) (Figure 20). Reducing the rate of housebuilding to zero net additions by 2035 can save 6% of the cumulative budget for 1.5°C by 2050 in avoided embodied and operational emissions. As we enter later carbon budgets, concrete and construction materials consume larger proportions of the budgets– even assuming decarbonisation rates aligned with industry net zero strategies (excluding their commitments to carbon capture and storage which are currently unproven; GCCA 2021), embodied emissions in new housing construction under the government's targeted expansion rates consume 8% and 27% of the budgets for 2038-2042 and for 2043-2050 respectively (Figure 20).

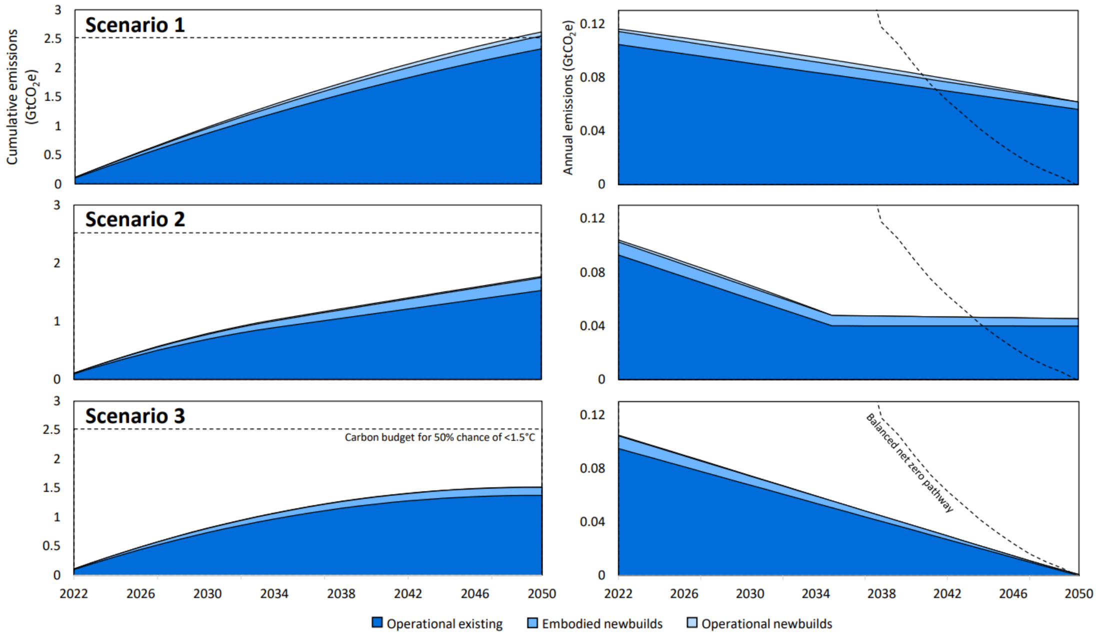

Figure 19. The impact of alternative housing policy scenarios on the emissions of the new and existing housing stock. Left hand side: cumulative emissions from housing. Dashed line represents England's cumulative carbon budget by 2050 compatible with a 50% probability of remaining within 1.5°C warming. Right hand side: annual emissions from housing. Dashed line represents England's balanced Net Zero pathway in the government's Net Zero strategy, which is consistent with <2°C warming.

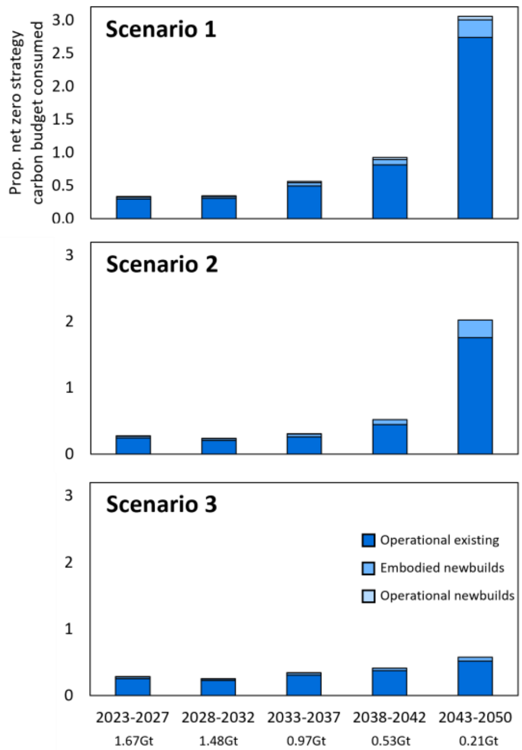

Figure 20. The proportion of each of England's future carbon budgets consumed by housing under alternative policy scenarios. The fourth, fifth and sixth carbon budgets (2023-2037) reflect those embedded in UK legislation, seventh and eighth (2038-2050) reflect the sum of the annual emissions targeted under the government's Balanced Net Zero pathway, adjusted for the population of England compared with the whole UK.

Alongside highlighting the recognised need for deep and rapid retrofitting of the existing stock, our analysis also reveals trade-offs between projected housebuilding as a mechanism for satisfying housing need and achieving national biodiversity and climate goals, empirically supporting multiple studies showing that reducing per capita demand for floor area from those with space in excess of their needs is essential to achieving sustainability goals (Serrenho et al. 2019; Pauliuk et al. 2021). So why do our policies for addressing housing need rely so heavily on housing expansion?

7.5 The political economy of housing expansion

The government's current strategy for satisfying housing need is an expansionary, high environmental resource consumption pathway, which if implemented in line with the assumptions of our Scenario 1 consumes the entire carbon budget for 1.5°C on its own. Escaping this pathway will require overcoming daunting political economy constraints. Recent theoretical work in ecological economics has uncovered major structural barriers to reducing growth rates in various sectors of the economy (Stratford 2020; Corlet Walker et al. 2021) – so called 'growth-dependencies' ("certain core aspects of human wellbeing become compromised when growth is either hard to come by or undesirable"; Corlet Walker & Jackson 2021). In this section, we explore the expansionary lock-in created by several growth-dependencies in the housing sector.

The first is created by a combination of 1) the dependence of English homeowners on property as a source of financial security (especially in the context of ongoing welfare state retrenchment; Corlet Walker et al. 2021), and 2) the political influence of those homeowners. As discussed in Section 2, a key motivation for first-time buyers in the UK is to secure housing as an investment, in the expectation that past rates of house price appreciation will continue (Gallent et al. 2017). Recent homebuyers – who may have been enabled to purchase homes because of easy credit conditions – are particularly vulnerable to the state of the housing market, finding themselves "at the top of a pyramid scheme" (Gallent et al. 2018) reliant on continued asset-price appreciation and ongoing low-interest rates to not suffer significant financial harms.

A fall in house prices induced by policies seeking to reduce the demand for housing as a financial asset would place these 'ordinary' homeowners (i.e. not institutional property investors) in an increasingly financially precarious position, potentially jeopardising their long-term financial security and even their ability to sell their house on without falling into debt. At scale, this could have a significant destabilising impact on England's entire economy.

The majority of the UK population fall into the homeowner category (63%), incentivised by half a century of government policy encouraging 'asset-based welfare' (building people's financial assets through their working life in order to compensate for relative reductions in state welfare provision, especially in old age; Doling & Ronald 2010). As a group they are significantly more likely than non-homeowners to vote in elections, vote for the Conservative party, and participate in local planning processes (Coelho et al. 2017; Christophers 2020). This political dominance has led to a competition between political parties as to who can best appeal to their preferences (Kohl 2020).

The risk of financial and social harms associated with declining house prices, combined with the political influence of homeowners, therefore translates into a lack of political will to tackle demand-side-driven house-price appreciation. Increasing supply via more housebuilding, in contrast, is much more politically feasible option (despite its inadequacy for fully addressing unmet housing need). It is also in the interests of the politically-influential UK property lobby, which made £60.8m in donations to the in-power Conservative party from 2010-2020, accounting for around 20% of the party's donations (Transparency International 2021). Of these, 10% came from just 10 specific property-connected sources (ibid). In addition, one-quarter of conservative Members of Parliament are landlords (openDemocracy 2021), presenting a potential conflict of interest against tackling house price inflation and policies supporting landlordism.

Secondly, the macroeconomic consequences of stagnating housebuilding would be profound. Sectors directly related to housebuilding (construction, housing and real estate) employ approximately one-eighth of the UK working population (ONS 2021c). Moreover, the construction sector has historically experienced considerable levels of labour productivity growth (output per job rose by 13.7% from 1990-2019 (ONS 2021d)), which theoretically means that construction must rise over time to maintain the same employment – the so-called 'productivity trap' (Jackson & Victor 2011). On the other hand, rising labour productivity may also reduce the costs of construction. The combined result of high employment in the sector and the labour productivity trap mean that even slowing the rate of growth in housebuilding (let alone halting it) could necessitate large structural changes in UK employment patterns.

Other macroeconomically important sectors are less directly dependent on housebuilding itself but would still face problems should demand-side repression policies be introduced. The financial sector is the best example, having become increasingly tied to property (see Section 2). With close to half of UK bank assets tied in to either domestic or commercial property, policies leading to a fall in house prices could materially affect the value of banks' collateral and their appetite for lending, with negative macroeconomic impacts in particular on smaller firms more dependent on bank loans (Ryan-Collins 2018).

Third, decades of government policy to reduce funding for local government have changed how affordable and social housing is financed. Under the contemporary system, local government's ability to deliver affordable and social housing to meet locals' fundamental housing needs is explicitly tied to their acceptance of new private-sector housing construction. This has come about through the rise of 'section 106 agreements' in which developers pay local councils (or promise contributions to local public services) in return for receiving planning permission for their proposed developments. Such payments financed 37-63% of all affordable housing from 2008-2014 (Brownill et al. 2015). Councils therefore have limited power to satisfy housing need without accepting expansion of market housing.

Even decarbonising the existing stock (the most important determinant of the overall emissions of the housing sector in our models) faces a challenging political economy.

Whilst aggressively upgrading the existing stock to achieving zero emissions from the existing stock by 2050 could save 38% of the carbon budget relative to business as usual (the difference in operational emissions of the existing stock between Scenarios 1 and 3), the housing sector is influenced by many vested interests who have financial stake in these high-consumption pathways. For example, research has revealed informal networks and coalition of actors from the natural gas, domestic boiler and connected industries promoting the discourse of a transition to 'green-gas' instead of the electrification of domestic heating that is favoured by the government's climate-related scientific advisory body (Lowes et al. 2020).

Combined, these growth-dependencies and political barriers not only underpin perpetual expansion of the housing stock, but also hinder the creation of a housing system that satisfies more housing need (Gallent et al. 2018). The asset poor are penalised by the ongoing inflation of house prices, but they have little political voice, disproportionately voting for out-of-power political parties (Milburn 2019; Christophers 2020). Reductions in state support for social housing has left a growing proportion of the population with no options other than be forced into the private rental sector (Ryan-Collins et al. 2017), which enables landlords to extract further rents (Stratford 2020). This then absorbs an increasing proportion of the wages of the asset-poor, reducing their opportunities to save for a deposit (Ryan-Collins et al. 2017; Mulheirn 2019), and contributing to why aspiring first-time-buyers are increasingly unable to enter the housing market.

The political economy of housing represents such a barrier to the implementation of systemic solutions to housing unaffordability that it has led housing scholars to argue that we are trapped between "the unimaginable and the unthinkable": either an unimaginable (and unsustainable) level of housebuilding, or an unthinkable definancialisation of the housing sector (Gallent et al. 2018), which runs counter to the interests of the homeowner classes and other powerful vested interests such as the construction and financial sectors.

Our analysis demonstrates that continued housing expansion with limited retrofitting of the existing stock as a mechanism for meeting housing need conflicts with England's ecological targets, and our political-economic review shows why we are locked into pathways of housing expansion regardless. Next, we explore policies for satisfying greater housing need whilst minimising ecological costs. We review three main areas: policies for satisfying greater housing need with the existing stock, definancialising housing, and improving the efficiency of the housing stock.

7.6 Policies for satisfying unmet housing need without undermining environmental policy targets

7.6.1 More efficient use of existing housing stock

The socio-economic distribution of the UK's existing housing stock and the consumption of housing services and living space is highly unequal. Tunstall (2015) shows a sustained reduction in housing space inequality from 1920-1980, counterbalanced by a significant increase from 1980 onwards and culminating in 2011 demonstrating the highest housing space inequality in over 50 years. By 2011 the most spaciously-housed decile of the population had five times the rooms/capita than the bottom decile (Dorling 2015). Therefore, one key lever for meeting greater housing need whilst minimising housing expansion could be through policies incentivising greater equity in housing space consumption and more efficient use of the existing stock (Lund 2019). We model complete utilisation of the housing stock (i.e. no vacant dwellings) as part of our Scenario 3.

There are multiple policy mechanisms for increasing the needs-satisfaction provided by the existing housing stock. There may be up to 1.2 million more homes than households in England (Mulheirn 2019); these are a mix of second homes, foreign-owned investment homes, and other classes of empty homes. In 2018-2019, there were at least 495,000 second homes in England not rented out in the private rental market (MHCLG 2020b). Rather than being treated as a public bad, second home ownership is incentivised under many current tax rules (e.g. second homes are eligible for council tax exemptions). In other jurisdictions with high house prices (e.g. Singapore, Vancouver), second homeowning is actively discouraged in order to free up stock to meet housing needs (Cheshire & Hilber 2021). Various tax reforms could be used to disincentivise the consumption of housing space for second homes: e.g. Cheshire & Hilber (2021) propose the replacement of various existing, regressive property taxes such as council tax with an Annual Proportional Property Tax, including a 25% surcharge on second homes.

Foreign homeownership similarly contributes to housing underutilisation: between 2014-2016 42% of newbuilds purchased by foreign investors in London were left unoccupied (Wallace et al. 2017). Numerous jurisdictions (e.g. Canada, New Zealand) have brought in policies to reduce housing demand from foreign investors (Minton 2021). Favilukis & Van Nieuwerburgh (2017) model the effect of taxes on out-of-town buyers on economic welfare and distributional impacts in New York, and find that transaction taxes on out-of-town purchases significantly benefit poorer residents and renters, depending on how the tax revenues are reinvested, although similar strategies in highly seasonal tourism-dependent economies could have negative economic effects (Hilber & Schöni 2020).

There are also many other forms of empty homes in England (e.g. neglected properties, properties with deceased owners), with estimates derived from council tax data (known to be underestimates) suggesting approximately 650,000 empty homes in England in 2019 (House of Commons Library 2020). Whilst local authorities do have some powers to bring empty homes back into use, additional policies have been suggested for increasing their capacity, including enhancing funding and legal powers to take control of empty homes and repurpose them for social housing (House of Commons Library 2020).

Under-occupation of existing stock could also be addressed, although equity considerations are essential, as policies such as the 'bedroom tax' (which reduced the housing benefits of people in social housing who were deemed to have one or more 'spare' bedrooms) targeting the poorest families have had demonstrable negative consequences (Shelter 2013). Consumption of inefficiently high levels of living space is implicitly subsidised through multiple mechanisms, such as a 25% council tax discount on single-occupied homes (Lund 2019). Additionally, there are barriers to families downsizing even when desired, such as stamp duty costs (a one-off tax incurred upon buying a new home) (Strutt & Parker 2015). Reducing transactions taxes might improve the efficiency of the use of the housing stock by improving occupier mobility (Hilber & Lyytikäinen 2017; Best & Kleven 2018). Some local authorities offer assistance and cash incentives to occupiers looking to downsize, to incentivise vacating underutilised stock (Lund 2019).

However, changes to taxation regimes and other approaches to incentivise more efficient use of the housing stock cannot ultimately guarantee that housing space is not overconsumed in a market with potentially insatiable demand. Baden-Baden in Germany, for example, taxes second homes up to 35% of imputed (or contract) rent - on top of a (low) property tax - yet the number of second homes has been increasing. A more ambitious approach might be the implementation of resource caps on living space, with all households occupying in excess of a given floorspace threshold (reflecting what is required to meet an individual's housing needs) participating in a 'cap-and trade' system for floorspace, capping the total amount of floorspace nationally at some level empirically estimated to be feasibly decarbonised in line with national decarbonisation targets (Horn & Ryan-Collins 2021).

All the policy proposals covered here are top-down approaches which face political barriers. Acknowledging that top-down policies are commonly implemented only if there is bottom-up support, another key dimension to increasing the efficiency of the use of housing space is cultural. From a sustainability perspective, the high income elasticity of demand for housing space presents a challenge as it implies that there will be a tendency for people to consume housing space (and therefore housing-related carbon emissions) in excess of their fundamental needs as incomes rise. However, this ultimately reflects cultural factors. There is limited empirical evidence for increases in housing space consumption improving subjective wellbeing for people who already have sufficient housing space to satisfy their needs, with evidence that people moving into larger homes quickly habituate and experience no or little long-run improvements in subjective wellbeing (Foye 2017). Enjoyment of housing space is also affected by the quality of services provided in the surrounding neighbourhood (Sirgy 2021). These suggest that voluntary reductions in the consumption of housing space by those with high levels of space could come with little adverse impact on wellbeing if embedded within high quality neighbourhood services. Culturally-transformative solutions to housing provisioning have been proposed, such as incentivising behavioural changes like increased co-living and space sharing to increase the needs-satisfaction per floor area of the existing stock (Corfe 2019).

7.6.2 Reducing demand for housing as a financial asset

Structural reforms are also possible which reduce housing's appeal as a financial asset whilst increasing its affordability to lower-income groups – thereby theoretically satisfying greater need without changes to the total stock. Multiple solutions to speculatively-driven house price inflation have been proposed (Wijburg 2020; Ryan-Collins 2021), which broadly target land rents or the unearned incremental increase in house values that is not due to the owners' own productive investment (i.e. home improvements), and reforms that slow the movement of wealth into housing assets more generally.

The tax reforms mentioned in section 5.1 in the context of increasing the efficiency of space use would also help reduce land rent extraction. The most comprehensive proposal for capturing land rents – with widespread support amongst economists - is a land-value tax, taxing the annual incremental increase in the unimproved market value of land. This tax has the benefit of capturing the increase in the price of land attributable to positive externalities of the state's and others' investments in the local area which improve public amenities and increase land value, thereby socialising the benefits that would otherwise be captured as rent (Ryan-Collins 2021). An additional positive social impact of land value taxes would be to reduce landowners' incentives to strategically hold land unproductively. Such a tax would also discourage borrowing against property for speculative gain and dampen the aforementioned housing-finance cycle.

Given the model of asset-based welfare outlined in section 4, to be politically acceptable these types of policy would need to be accompanied with public investments in the welfare state – especially pensions and social care – so that individuals are less dependent on house price for their long-term financial security.

Financial reforms could also assist in reducing house prices. The most powerful public bodies in relation to the quantity and price of mortgage lending in the economy are central banks and financial supervisors. Credit policies have been implemented by central banks historically in many high-income economies to reduce undesirable credit flows and encourage more productive and strategic lending (Bezemer et al. 2021). Historically, these favoured sectors like high value-added manufacturing and export industries and repressed lending for domestic consumption or house purchase. These became unfashionable in the 1980s with financial liberalisation but since the 2008 crisis, 'macroprudential' policies, aimed at repressing credit in particular undesirable sectors have returned (ibid). Housing-related macroprudential policy has included tighter loan-to-value and loan-to-income ratios for households on the demand side, whilst on the supply side requiring banks to hold more capital against certain types of real estate lending. The policy has proven to be effective in some cases in reducing mortgage credit flow (Muñoz 2020). Such policies have been implemented by central banks and financial supervisors due to financial stability rather than affordability/sustainability concerns, but their use could be expanded. This would probably require greater coordination between central banks and governments, which has historical precedent (Ryan-Collins & Van Lerven 2018).

In addition, currently most central banks do not include house prices (as opposed to the cost of housing) in their definition of consumer price inflation. This has allowed rapid increases in house prices to co-exist with very low or zero interest rates. Central banks could follow New Zealand's example and consider rethinking their measure of inflation to include housing costs which would create a stronger link between house prices and interest rates (Bloomberg 2021).

In England, structural reforms to help redirect lending away from property and towards productive investment may be required. The UK banking sector is dominated by large shareholder banks who have a preference for larger mortgage loans and are heavily reliant on real estate as collateral. Reforms could promote the development of local/regional community-based banks (the primary holders of bank deposits in Germany) who develop strong relationships with firms as way of de-risking their loans (Ryan-Collins 2021; Kazi & MacFarlane 2022). This would also help develop a more resilient financial sector more generally that could help mitigate the macroeconomic growth-dependencies mentioned in section 4.

Gallent et al. (2017) propose an innovative solution for reducing house prices for those seeking residence whilst maintaining opportunities for speculative investment. They discuss reforming planning law to distinguish between 'resident' and 'investment' housing, with different tax and ownership rules depending on each housing class. Households would be permitted to purchase a single resident home, which would be subject to high capital gains taxation when sold on to prevent homeowners from extracting economic rents. This 'resident' housing would be broadly designed to satisfy basic housing needs, leaving 'investment' housing as a financial asset to be consumed by investors, but without the flow of investment capital competing with ordinary homeowners for housing space and crowding out buyers looking to secure a home to meet their housing needs.

An additional set of key definancialisation solutions revolve around land ownership reforms. A simple way to ensure that the benefits of rising land values are not captured by rentiers is for land to be publicly-owned. Whilst 1.6 million hectares of publicly-owned land in the UK (8% of Britain's land area) have been privatised since the 1980s (Christophers 2020), the state still owns large tracts of land; and there is international precedent to the state playing a larger role in socialising the benefits of land value uplift. For example, in Singapore, 90% of land is owned by the state and

82% of the population lives in public housing (Ryan-Collins 2021). The state leases out land to developers for construction, and captures the land value uplift via increased lease prices on renewal.

7.6.3 Principles for newbuilds

Even implementing the above measures, there may still be unmet housing need from low-income households, and so new principles are required for newbuilds to be compatible with national sustainability targets whilst targeting unmet social needs. Directly targeting unmet needs requires primarily delivering social housing over ordinary market housing. Recent evidence from Finland demonstrates that the addition of social housing to the housing stock is initially much more likely to generate homes occupied by low-income households than market housing (for every 100 inner-city social homes added to the stock, 43 vacancies throughout the moving chain were immediately created for households in the bottom 50% of the income distribution, compared with 29 for market-rate homes, though the differences dissipate over time; Bratu et al. 2021). However, renewed construction of social housing would require a shift in government policy away from subsidising private landlords to house low-income tenants (via housing benefit) and towards direct social housing construction.

From a climate perspective, in order for new additions to the housing stock today to not require retrofitting by 2050, the government would need to implement standards to ensure that all new homes achieve net zero operational emissions and minimise embodied emissions as soon as possible. Current government policy is for all new homes from 2025 to be ‘zero carbon ready’ (i.e. energy-efficient and supplied by electrical heating so that they decarbonise over time as the grid decarbonises), although notably this same goal had previously been set in 2006 for 2016, only to be scrapped in 2015 (H.M. Treasury 2015; Oldfield 2015).

In order for new housing to unambiguously contribute to achieving the end of wildlife declines by 2030, the implementation of both Biodiversity Net Gain and species mitigation legislation should be strengthened. Biodiversity Net Gain could be improved primarily by mandating that impacts to irreplaceable habitats and protected and unprotected wildlife sites (e.g. ancient woodlands) be avoided, and by putting governance and monitoring mechanisms in place to ensure that biodiversity promises made in planning applications materialise in reality (i.e. ensuring that regulators have sufficient tools to enforce the delivery of Biodiversity Net Gain; zu Ermgassen et al. 2021, 2022). In addition, the evidence base behind the effectiveness of species mitigation measures for housing development impacts remains weak, with only 29% of species mitigation measures demonstrably successful at preventing harms to wildlife of new housing (Hunter et al. 2021). Using only evidence-based mitigation techniques would increase confidence that housing expansion does not trade-off against wildlife abundance goals. Densification can also play an important role in reducing both carbon emissions and biodiversity impacts by reducing urban land-take and reducing car-dependency (OECD 2020, 2021b).

7.6.4 Retrofitting the existing stock

The key strategies for decarbonising the existing stock revolve around electrifying heating and improving home insulation and energy efficiency (reviewed extensively in CCC 2019, 2020; RICS 2020; EAC 2021). Our models demonstrate that immediate action is required to dramatically reduce the emissions of the existing stock, as gradual decarbonisation pathways overlook that a large proportion of England's cumulative carbon budget to 2050 will be consumed in the next few years because of their high current operational emissions. For example, our models estimate the existing stock is currently consuming approximately 4% of the cumulative carbon budget (for 1.5°C) each year.

7.7 Conclusion

Our study models the effects of the English government's housing policy and estimates that it risks consuming the entire national cumulative carbon budget consistent with 1.5°C warming. It also demonstrates the urgency of retrofitting the existing stock, as retrofitting all existing homes to zero carbon by 2050 could save 38%

of the cumulative carbon budget for 1.5°C relative to a business-as-usual scenario which extrapolates current decarbonisation trends whilst achieving the government's construction targets. Meeting society's housing needs without relying on emissions-intensive housing expansion or speculative technological innovations relies on satisfying greater housing need through the existing housing stock. Accelerating retrofits, increasing the environmental standards of newbuilds so they achieve zero carbon and no net impact on wildlife populations (by strengthening Biodiversity Net Gain policy), and reductions in housing expansion rates, all play a role if the housing sector is to contribute to national sustainability objectives. However, the policy innovations that could encourage greater housing need satisfaction from the existing housing stock (e.g. tax reforms, macroprudential policy) face an intimidating political economy. Nevertheless, political and economic barriers (e.g. political power of homeowners, impacts on employment and the financial sector) cannot hide that more equitable use of housing is likely necessary to meet England's unmet housing need without transgressing national sustainability objectives. This study shows that in this case theoretical pathways to simultaneously achieving infrastructure, housing and ecological SDGs do exist, but they require a significant change from the business as usual strategy for satisfying society's housing needs.

Chapter 8 Discussion

The world is in the midst of the greatest expansion of built infrastructure in history. If projections of infrastructure growth over the coming decades are realised, under current production technologies and impact mitigation practices this infrastructure alone will most likely lead to the transgression of the planet's 'safe operating space' across multiple dimensions (Steffen et al. 2015b). Reforming biodiversity impact mitigation practices around the world so that new infrastructure projects inflict no net harm on nature or wildlife will play an important role if contradictions between the ecological and infrastructure goals within the SDGs are to be reconciled (Chapter 2). However, impact mitigation policies have a long way to come before they consistently achieve no net loss outcomes (Chapter 3). Major barriers include issues such as self-selection bias which can undermine the additionality of biodiversity offsetting (Chapter 4), or a lack of resourcing for implementing agencies so that compensation measures can go unmonitored and unenforced (Chapter 6). Governments also sometimes intervene in offsetting regulatory markets if they are perceived to act as a barrier to development objectives, suppressing their ecological outcomes (Chapter 5). As a result, improved impact mitigation practices can only be part of the solution; the other part is reducing the fundamental drivers of biodiversity loss in the first place, which can include deep-rooted economic structures which traditionally have fallen outside the domain of conservation science (Chapter 7).

8.1 How effective are biodiversity offsets at delivering No Net Loss of biodiversity?

The core question of this thesis is: does biodiversity offsetting work? Chapter 3 documents mixed ecological outcomes. Across the duration of my PhD, further evidence has been uncovered from systems all over the world showing outcomes ranging from impressive successes in challenging socio-ecological contexts (e.g. the Ambatovy offset in Madagascar; Devenish et al. 2022) to clear failures (widespread offset implementation failures found in Isère, France; Bezombes et al. 2019). This thesis has also contributed to the evidence by showing it is unlikely the two systems I have evaluated (Chapters 4 and 6) have/will achieve(d) their stated policy goals.

This thesis makes contributions towards developing a systematic understanding of when biodiversity offsets succeed or fail in ecological terms. Despite the relatively low quality of the evidence for the outcomes of biodiversity offsetting and the prevalence of unreliable study designs (Chapter 3), two consistent key themes have emerged from the last 3.5 years of offsetting research. One is that offsets are much more likely to be successful if they are compensating for impacts to degraded or fast-recovering ecosystems – better outcomes have been consistently documented in relatively ecologically simple systems. The second is that high-quality governance is vital for securing positive outcomes; and several jurisdictional offsetting policies fall short on quality governance, and least in part because the offsetting policy is trying to achieve multiple, sometimes contradictory, policy objectives.

8.1.1 Ecological complexity as a determinant of offsetting success

The former conclusion is not novel, but this thesis has helped formalise the evidence. Numerous authors have highlighted the importance of the complexity of the conservation/restoration measures for determining offsetting outcomes (Maron et al. 2012; Bull et al. 2013a; Pilgrim et al. 2013; Simmonds et al. 2022), and a key concept underpinning the mitigation hierarchy itself is that offsetting should only be permitted if the impacted biodiversity can be feasibly replaced within acceptable timescales. Simmonds et al. (2022; see Appendix 1) extend work analysing the ecological feasibility of offsetting measures to include social and governance feasibility as well. Chapter 3 documents that many of the studies that have demonstrated offsetting successes correspond to fast-recovering biotopes or species which respond rapidly if threats are minimised or appropriate habitats are provided, such as simple mitigation wetlands in North America (Quigley & Harper 2006a), birds associated with mud flats (Murata & Feest 2015), or fast-reproducing frogs under threat from the impacts of development (Pickett et al. 2013).

Despite this consistent finding, offsetting policies are still widely applied outside ecosystems where they are ecologically justifiable, with Bull & Strange (2018) demonstrating that around two-thirds of offsets globally are applied in wooded ecosystems. Ideally, strict application of the mitigation hierarchy would preclude damaging such ecosystems in the first place. However, if offsets are necessary, it is clear that the use of offsets in slow-recovering or threatened systems should be contingent on very high standards of governance and clear accountability mechanisms to ensure the risk of failure, and therefore offsets facilitating the loss of vulnerable biodiversity, is as low as possible.

8.1.2 The importance of governance, monitoring and enforcement

One such example of a successful offsetting outcome in a complex ecosystem is the Ambatovy offsets in the rainforests of Madagascar (Devenish et al. 2022), which demonstrates that ecological practicalities are not the sole determinant of offset outcomes. A second key determinant is offset governance, especially the establishment of monitoring and enforcement mechanisms which enable accountability of the offset provider. The Ambatovy offsets were an international best-practice case study, accompanied with very high levels of resourcing (a major conservation NGO was contracted by the offset proponent to implement an ambitious conservation programme as part of the offset activities) and clear and well-defined project aims.

The importance of clear objectives for offset governance cannot be understated. For individual, bespoke voluntary offsets (such as the Ambatovy offsets), the dominant objective of the offset is to achieve the stated biodiversity outcomes. Such offsets tend to be characterised by clear ecological objectives, and high levels of resourcing, and have led to arguably the clearest examples of successful ecological outcomes from offsetting (Pickett et al. 2013; Murata & Feest 2015; Devenish et al. 2022). However many of the studies documenting where offsetting policies are likely to have fallen short have been reported in jurisdictional offsetting systems (which make up >99% of known implemented offsets globally; Bull & Strange 2018), such as New South Wales

(Gibbons et al. 2018), Isère (France; Bezombes et al. 2019) or England (Chapter 6). In most jurisdictional systems, biodiversity compensation has emerged as part of the development planning process as an uneasy policy compromise between nature conservation and development objectives (Miller et al. 2015; Sullivan & Hannis 2015; Evans 2017). In England, for example, the government explicitly states in the relevant policy documentation that the policy simultaneously aims to achieve net gain in biodiversity, simplify the planning process for developers, and improve people's access to greenspace – without mentioning that all three of these objectives may occasionally contradict each other (Defra 2019b).

One impact of the contradictory policy goals facing jurisdictional offsetting policies is that it is often in the offset procurer's and regulator's interest to minimise the costs and administrative burden of implementing the offset system (Walker et al. 2009). This can manifest in attempts to reduce monitoring costs, a lack of enforcement, or regulators intervening in offsetting regulatory markets to bring down transaction costs to the detriment of ecological outcomes (Chapter 5). In Australia, offsetting policies such as environmental offsets under the Environmental Protection and Biodiversity Conservation Act nominally aim to achieve no net loss of biodiversity, but their governance and policy design has been consistently demonstrated to be inadequate for securing these ecological outcomes (Maron et al. 2015; Maseyk et al. 2020; Samuel 2020), which could be interpreted as strategic under-resourcing to ensure offsetting is not perceived to hinder Australia's mining- and agriculture-focused economic development model. Similarly, under-resourcing which leads to weak offset governance has been identified in the French and English systems (Quétier et al. 2014; Chapter 6).

8.1.3 The challenging political economy of doing offsetting properly

One major unresolved question that has emerged from this thesis is whether these reported shortcomings in several jurisdictional offsetting systems are a feature, or a bug. Damiens et al. (2020) review how the discourses around biodiversity offsetting have changed over time and across offsetting systems around the world since the

1950s. They argue that a consistent pattern observable across the history of offsetting discourses is that, when an influential stakeholder group (such as the IUCN) plans to operationalise a radical interpretation of offsetting where the true biodiversity costs of development are internalised into the development process and argue that limits to development are necessary to reconcile ecological and development goals, sectors potentially impacted by this more economically-burdensome interpretation politically organise to counter this narrative. They argue that this concerted effort from those aiming to shape and operationalise a less restrictive interpretation of offsetting has led to them becoming "'symbolic instruments', discursively acting to divert attention away from more transformative action limiting and reversing the depletion of biological diversity". Some of the trends towards more flexible biodiversity trading systems over time I observed in Chapter 5 could be seen to reflect this narrative.

On the other hand, there have been enough offsetting successes (e.g. Devenish et al. 2022) that it remains possible that it is politically feasible to construct an offsetting system which genuinely internalises the biodiversity impacts of development and achieves overall no net loss or net gain outcomes. Damiens et al. (2020) document recent shifts in offsetting discourses, towards operationalising more ambitious biodiversity targets consistent with nature recovery rather than merely offsetting the impacts of individual developments (e.g. Bull et al. 2020). The choice of a 'net gain' policy goal in England's new system over no net loss could be seen as evidence for this increased ambition (although we also document the governance shortcomings that mean it is unlikely to fulfil this ambition, Chapter 6, and ongoing UK cuts in statutory biodiversity funding; zu Ermgassen et al. 2021a, Appendix 1). Damiens et al. (2020) argue that the historical evidence suggests there will be a reactionary response from affected industries to these renewed attempts to operationalise more radical interpretations of offsetting, and time will tell whether their predictions hold true.

8.2 Opportunities for integration between conservation science and postgrowth economics

Given ongoing uncertainty about the effectiveness of biodiversity offsetting, there is a need to address the fundamental drivers of biodiversity loss and focus on the demand-side of reducing the production of new infrastructure that does not directly contribute to improvements in social welfare. Chapter 7 shows, in our chosen case study, infrastructure expansion is on track to cause the transgression of environmental limits whilst simultaneously being unlikely to directly satisfy unmet infrastructure need. Working with colleagues from multiple disciplines, I make a start at analysing the political-economic drivers of this unsustainable infrastructure proliferation. Chapter 7 is amongst the first research projects to have explored the growth-dependencies in the infrastructure sector – but only for a single infrastructure class in a single country (zu Ermgassen et al. 2022a). There is an obvious need to expand this work to other contexts.

Williams et al. (2020) have recently discussed the role of conservation science in halting biodiversity loss. They review the conservation science literature and identify that the vast majority of conservation science research documents trends in the changing state of nature and biodiversity. One of their main conclusions is that, for conservation science to become more effective, it must change to focus more on understanding the proximate and ultimate drivers of biodiversity loss, as well as an increased focus on identifying and evaluating solutions addressing these threats (Figure 21).

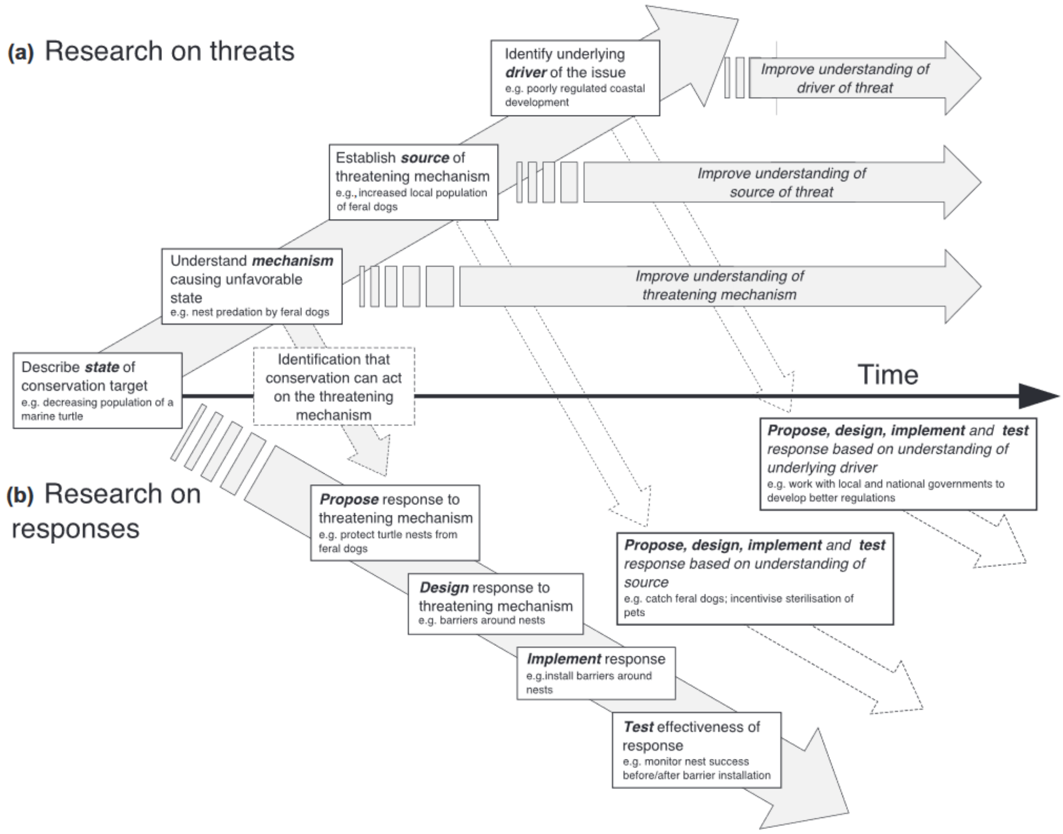

Figure 21. Schematic of the role of conservation science in tackling biodiversity loss. One of the authors' key arguments is that conservation science research requires greater focus on the underlying drivers of biodiversity loss (top box in the figure), as well as uncovering and eventually testing mechanisms for addressing these threats (bottom box). Figure adopted from Williams et al. (2020) with permission from David Williams

The main focus of this thesis was on testing and evaluating the outcomes of conservation measures implemented to address the threats caused by infrastructure expansion and land use change, which falls comfortably within the wider discipline of conservation science and within the 'research into responses' component of the conservation science framework of Williams et al. (2020) (Figure 21). However, in Chapter 7, we focus more on the political economy and economic structural drivers of infrastructure production, one of the ultimate drivers of biodiversity loss. This also falls within the Williams et al. (2020) framework. However, very little of the literature we used to explore the political economy of this driver of biodiversity loss came from the conservation science literature, with most coming from the disciplines of political economy, industrial ecology and postgrowth economics.

I would argue that this thesis shows that there is considerable promise in better integration between conservation science and postgrowth economics, especially in helping identify the structural drivers of biodiversity loss and identify leverage points to induce "transformative change" to economic systems so they stop generating poor biodiversity outcomes in aggregate (Meadows 2008; Díaz et al. 2019). Analysing how to solve the ecological impacts of housing production in England from a conservation science perspective would conventionally focus on improving the quality of the conservation measures used to compensate for the ecological damage, which was the focus of Chapter 6 analysing the impacts of Biodiversity Net Gain. However Chapter 7 demonstrates that, even with the perfect implementation of Biodiversity Net Gain, housing would still have ecological impacts (via carbon emissions) that would lead to the transgression of ecological policy goals. Working with political economists and postgrowth economists helped identify the structural drivers and solutions to this fundamental driver of biodiversity loss, which ended up being related to both monetary policy (credit-creation) and 'growth-dependencies' which would traditionally have fallen outside the domain of conservation science.

These results suggest there may be many other major biodiversity threats driven by structural economic factors that traditionally fall outside the domain of conservation science, but which should be more intensively researched. For example, research has shown connections between individuals' extreme wealth (i.e. economic inequality) and agricultural commodity-driven deforestation (Ceddia 2020), some evidence for linkages between exchange rates and deforestation (Arcand et al. 2008), and evidence that the European Central Bank's quantitative easing programme has inadvertently harmed ecosystems by purchasing corporate bonds from companies with large deforestation and water use footprints (Kedward et al. 2021). Greater collaboration between the domains of postgrowth economics, sustainable finance and conservation science could be a powerful mechanism for understanding and addressing these ultimate drivers of biodiversity loss, and 'bending the curve' of biodiversity loss over the coming decades (Mace et al. 2018).

8.3 Conclusion

This thesis has investigated mechanisms for mitigating the ecological impacts of the global infrastructure boom. Primarily focused on the effectiveness of biodiversity offsetting as a tool for compensating for the residual impacts of infrastructure, the thesis demonstrates that outcomes of offsets are variable, and big investments in governance and improvements in policy design are necessary if offsetting is to play a key role in mitigating the impacts of infrastructure into the future. The inconsistent outcomes of offsets indicate that improved mitigation practices must be coupled with demand-reduction initiatives which reduce the need for additional infrastructure proliferation and its associated ecological impacts. Only with a deep commitment to improving mitigation policies and reducing these underlying pressures, will we be able to reconcile the SDGs and meet the infrastructure needs of all whilst halting nature's decline.

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Appendix I Co-authored publications

I published 13 peer-reviewed articles and two preprints during my PhD, of which 5 are included as thesis chapters. I made a substantial contribution to five articles directly related to this thesis which are not included as chapters, listed below. I was the primary supervisor for Hunter et al. (2021), which went on to win the Georgina Mace award for best early-career (<5 years post-PhD) research paper in the journal *Ecological Solutions and Evidence*. I then list the remaining papers I co-authored during my thesis.

Papers relating to my thesis where I made a key contribution (with Abstracts)

Torres, A., **zu Ermgassen, S.O.S.E.**, Ferri-Yanez, F., Navarro, L.M., Rosa, I.M., Teixeira, F.Z., Wittkopp, C. and Liu, J., 2022. Unearthing the global impact of mining construction minerals on biodiversity. *bioRxiv*.

Construction minerals – sand, gravel, limestone – are the most extracted solid raw materials and account for most of the world's anthropogenic mass, which as of 2020 outweighed all of Earth's living biomass. However, knowledge about the magnitude, geography, and profile of this widespread threat to biodiversity remains scarce and scattered. Combining long-term data from the IUCN Red List and new species descriptions we provide the first systematic evaluation of species threatened by mining of construction minerals globally. We found 1,047 species in the Red List impacted by this type of mining, of which 58.5% are threatened with extinction and four species already went extinct. We also identified 234 new species descriptions in 20 biodiversity hotspots reporting impacts from mining. Temporal trends in the assessments highlight the increased saliency of this threat to biodiversity, whose full extent may well reach over 24,000 animal and plant species. While rock quarrying mostly threatens karst biodiversity and narrow-ranged species, sand and gravel extraction is a more prominent threat to freshwater and coastal systems. This study provides the first evidence base to support a global strategy to limit the biodiversity impacts of construction mineral extraction.

Simmonds, J.S., von Hase, A., Quétier, F., Brownlie, S., Maron, M., Possingham, H.P., Souquet, M., **zu Ermgassen, S.O.S.E.**, ten Kate, K., Costa, H.M. and Sonter, L.J., 2022. Aligning ecological compensation policies with the Post-2020 Global Biodiversity Framework to achieve real net gain in biodiversity. *Conservation Science and Practice*, *4*(3), p.e12634.

Increasingly, government and corporate policies on ecological compensation (e.g., offsetting) are requiring "net gain" outcomes for biodiversity. This presents an opportunity to align development with the United Nations Convention on Biological Diversity Post-2020 Global Biodiversity Framework's (GBF) proposed ambition for overall biodiversity recovery. In this perspective, we describe three conditions that should be accounted for in net gain policy to align outcomes with biodiversity recovery goals: namely, a requirement for residual losses from development to be compensated for by (1) absolute gains, which are (2) scaled to the achievement of explicit biodiversity targets, where (3) gains are demonstrably feasible. We show that few current policies meet these conditions, which risks undermining efforts to achieve the proposed Post-2020 GBF milestones and goals, as well as other jurisdictional policy imperatives to halt and reverse biodiversity decline. To guide future decision-making, we provide a supporting decision tree outlining net gain compensation feasibility.

Hunter, S.B., **zu Ermgassen, S.O.S.E.**, Downey, H., Griffiths, R.A. and Howe, C., 2021. Evidence shortfalls in the recommendations and guidance underpinning ecological mitigation for infrastructure developments. *Ecological Solutions and Evidence*, *2*(3), p.e12089.

In the United Kingdom and European Union, legal protection of species from the impacts of infrastructure development depends upon a number of ecological mitigation and compensation (EMC) measures to moderate the conflict between development and conservation. However, the scientific evidence supporting their effectiveness has not yet been comprehensively assessed.

This study compiled the measures used in practice, identified and explored the guidance that informed them and, using the Conservation Evidence database, evaluated the empirical evidence for their effectiveness.

In a sample of 50 U.K. housing applications, we identified the recommendation of 446 measures in total, comprising 65 different mitigation measures relating to eight taxa. Although most (56%) measures were justified by citing published guidance, exploration of the literature underpinning this guidance revealed that empirical evaluations of EMC measure effectiveness accounted for less than 10% of referenced texts. Citation network analysis also identified circular referencing across bat, amphibian and reptile EMC guidance. Comparison with Conservation Evidence synopses showed that over half of measures recommended in ecological reports had not been empirically evaluated, with only 13 measures assessed as beneficial.

As such, most EMC measures recommended in practice are not evidence based. The limited reference to empirical evidence in published guidance, as well as the circular referencing, suggests potential ‘evidence complacency’, in which evidence is not sought to inform recommendations. In addition, limited evidence availability indicates a thematic gap between conservation research and mitigation practice. More broadly, absence of evidence on the effectiveness of EMC measures calls into question the ability of current practice to compensate for the impact of development on protected species, thus highlighting the need to strengthen requirements for impact avoidance. Given the recent political drive to invest in infrastructure expansion, high-quality, context-specific evidence is urgently needed to inform decision-making in infrastructure development.

Torres, A., Simoni, M.U., Keiding, J.K., Müller, D.B., **zu Ermgassen, S.O.S.E.**, Liu, J., Jaeger, J.A., Winter, M. and Lambin, E.F., 2021. Sustainability of the global sand system in the Anthropocene. One Earth, 4(5), pp.639-650.

Sand, gravel, and crushed rock, together referred to as construction aggregates, are the most extracted solid materials. Growing demand is damaging ecosystems, triggering social conflicts, and fueling concerns over sand scarcity. Balancing protection efforts and extraction to meet society's needs requires designing sustainable pathways at a system level. Here, we present a perspective on global sand sustainability that shifts the focus from the mining site to the entire sand-supply network (SSN) of a region understood as a coupled human-natural system whose backbone is the physical system of construction aggregates. We introduce the idea of transitions in sand production from subsistence mining toward larger-scale regional supply systems that include mega-quarries for crushed rock, marine dredging, and recycled secondary materials. We discuss claims of an imminent global sand scarcity, evaluate whether new mining frontiers such as Greenland could alleviate it, and highlight three action fields to foster a sustainable global sand system.

zu Ermgassen, S.O.S.E., Bull, J.W. and Groom, B., 2021. UK biodiversity: close gap between reality and rhetoric. *Nature*, *595*(7866), pp.172-172.

In a bid to position the United Kingdom as a global environmental leader before this year's United Nations biodiversity conference (COP15) and climate-change conference (COP26), the UK government has announced biodiversity initiatives to halt species declines by 2030 and to protect 30% of its land area (see, for example, go.nature.com/3x4yk1k). These plans are at odds with its current spending on conservation.

The government's conservation funding fell by 42% in real terms between 2008 and 2018 to just 0.02% of gross domestic product (GDP; see go.nature.com/2udg3od). It missed 14 of its 20 international biodiversity commitments (Aichi targets) in 2020 (see go.nature.com/3dor8ra). This year it commissioned the Dasgupta Review, which calls for economic changes to stop biodiversity loss (see go.nature.com/3jozldl).

However, even taking into account the May announcement of a 47% increase in Natural England's funding (see go.nature.com/2t96qjn), the country still spends less than other nations with comparable GDP (see A. Seidl et al. Nature Ecol. Evol. 5, 530–539; 2021 and go.nature.com/2udg3od). The United Kingdom needs to reconsider its public expenditure priorities if it is to close the gap between rhetoric and reality.

Additional peer-reviewed publications published during my thesis

zu Ermgassen, P.S.E., Baker, R., Beck, M.W., Dodds, K., **zu Ermgassen, S.O.S.E.**, Mallick, D., Taylor, M.D. and Turner, R.E., 2021. Ecosystem services: Delivering decision-making for salt marshes. *Estuaries and Coasts*, 44(6), pp.1691-1698.

zu Ermgassen, P.S.E., DeAngelis, B., Gair, J.R., **zu Ermgassen, S.O.S.E.**, Baker, R., Daniels, A., MacDonald, T.C., Meckley, K., Powers, S., Ribera, M. and Rozas, L.P., 2021. Estimating and applying fish and invertebrate density and production enhancement from seagrass, salt marsh edge, and oyster reef nursery habitats in the Gulf of Mexico. *Estuaries and Coasts*, 44(6), pp.1588-1603.

Everard, M., Kass, G., Longhurst, J., **zu Ermgassen, S.O.S.E.**, Girardet, H., Stewart-Evans, J., Wentworth, J., Austin, K., Dwyer, C., Fish, R. and Johnston, P., 2021. Reconnecting society with its ecological roots. *Environmental Science & Policy*, 116, pp.8-19.

Milner-Gulland, E.J., Addison, P., Arlidge, W.N., Baker, J., Booth, H., Brooks, T., Bull, J.W., Burgass, M.J., Ekstrom, J., **zu Ermgassen, S.O.S.E.**, Fleming, L.V. et al., 2021. Four steps for the Earth: mainstreaming the post-2020 global biodiversity framework. *One Earth*, 4(1), pp.75-87.

Bull, J.W., Milner-Gulland, E.J., Addison, P.F., Arlidge, W.N., Baker, J., Brooks, T.M., Burgass, M.J., Hinsley, A., Maron, M., Robinson, J.G., Sekhran, N. **et al.**, 2020. Net positive outcomes for nature. Nature Ecology & Evolution, 4(1), pp.4-7.

Appendix II Chapter 3 Supporting Information

Review search terms

WoS search

((Biodiversity OR ecological OR environmental OR wetland OR conservation OR species) AND (mitigation OR offset* OR compensat* OR bank* OR "no net loss" OR credit*) AND (effective* OR impact* OR success OR evaluat* OR failure* OR compliance OR outcome* OR benefit* OR evidence) NOT ("CO2 mitigation" OR "climate mitigation" OR "GHG emissions" OR “CO2 emission” OR “CO2 emissions” OR "ecological compensation depth" OR "river bank" OR "seed bank" OR "spore bank" OR "egg bank" OR "saba bank" OR "bank vole" OR "gas mitigation" OR "tissue bank" OR "DNA bank" OR "gene bank" OR "Doppler radar" OR digital OR psychotherapy OR alcohol OR wireless OR "brood size" OR runoff OR "Banks peninsula" OR CMOS OR "inflation bias" OR "beetle bank" OR arsenic OR selenium OR boron OR cadmium OR zinc OR gypsum OR “nitrous oxide” OR “lead concentration” OR “lead concentrations” OR catalyst* OR wind OR “disaster mitigation” OR REDD OR REDD+ OR “renewable energy” OR “paternal care” OR ocean* OR deep-sea OR “animal welfare” OR atmospher* OR ester OR photovoltaic OR agrivoltaic OR roadkill OR N20 OR denitrification OR fuzzy OR “air quality” OR PM2.5 OR PM10 OR physiolog* OR nitrogen OR membrane OR fishmeal OR tetracycline OR thermoregulat* OR “CO2 enrichment” OR gender OR dimorph* OR polymorph* OR “life stage” OR “life stages” OR airline OR “energy savings” OR “energy saving” OR “integrated circuits” OR ammonia OR liquefied OR “food waste” OR enzyme OR “propagule banks” OR turbine* OR microb* OR metro OR “sex change” OR “thermal stress” OR “vehicle strike” OR “tax reform” OR speciat* OR chromosome* OR earthquake OR “seed size” OR seafood OR phthalate OR CPUE OR “vehicle collision” OR “green roof” OR “green roofs” OR “drought impact” OR Dodd-Frank OR deepwater OR xylem OR head-start* OR Beverton-Holt OR “agricultural intensification” OR “sustainable intensification” OR pesticide* OR herbicide* OR biochar OR rodenticide* OR pharma* OR "experimental drought" OR "social ratings" OR "social bonds" OR "thermal regimes" OR "rodent eradication" OR seaweed OR phenolog* OR tropospher* OR phospholipid* OR nano* OR metal* OR disaster OR mRNA OR "biological invasions" OR smoking OR nicotine OR "competitive release" OR phytoscreening OR brewery OR mutualis* OR nonlethal OR germplasm OR "hazardous waste" OR ciliate* OR cultivar OR diabetes OR benz* OR "climate-smart agriculture" OR electroc*))

Refined by: WEB OF SCIENCE CATEGORIES: (ENVIRONMENTAL SCIENCES OR ECOLOGY OR ENVIRONMENTAL STUDIES OR BIODIVERSITY CONSERVATION OR MARINE FRESHWATER BIOLOGY OR WATER RESOURCES OR ECONOMICS OR ENGINEERING ENVIRONMENTAL OR ZOOLOGY OR FORESTRY OR PLANT SCIENCES OR PUBLIC ENVIRONMENTAL OCCUPATIONAL HEALTH OR PLANNING DEVELOPMENT)

Timespan: 2003-2018. Indexes: SCI-EXPANDED, SSCI, A&HCI, CPCI-S, CPCI-SSH, ESCI.

Restricted review language to English only.

Scopus search

TITLE-ABS-KEY ((biodiversity OR ecological OR environmental OR wetland OR conservation OR species) AND (mitigation OR offset* OR compensat* OR bank* OR "no net loss" OR credit*) AND (effective* OR impact* OR success OR evaluat* OR failure* OR compliance OR outcome* OR benefit* OR evidence) AND NOT ("CO2 mitigation" OR "climate mitigation" OR "GHG emissions" OR "CO2 emission" OR "CO2 emissions" OR "ecological compensation depth" OR "river bank" OR "seed bank" OR "spore bank" OR "egg bank" OR "saba bank" OR "bank vole" OR "gas mitigation" OR "tissue bank" OR "DNA bank" OR "gene bank" OR "Doppler radar" OR digital OR psychotherapy OR alcohol OR wireless OR "brood size" OR runoff OR "Banks peninsula" OR cmos OR "inflation bias"

OR "beetle bank" OR arsenic OR selenium OR boron OR cadmium OR zinc OR gypsum OR "nitrous oxide" OR "lead concentration" OR "lead concentrations" OR catalyst* OR wind OR "disaster mitigation" OR redd OR redd+ OR "renewable energy" OR "paternal care" OR ocean* OR deep-sea OR "animal welfare" OR atmospher* OR ester OR photovoltaic OR agrivoltaic OR roadkill OR n20 OR denitrification OR fuzzy OR "air quality" OR pm2.5 OR pm10 OR physiolog* OR nitrogen OR membrane OR fishmeal OR tetracycline OR thermoregulat* OR "CO2 enrichment" OR gender OR dimorph* OR polymorph* OR "life stage" OR "life stages" OR airline OR "energy savings" OR "energy saving" OR "integrated circuits" OR ammonia OR liquefied OR "food waste" OR enzyme OR "propagule banks" OR turbine* OR microb* OR metro OR "sex change" OR "thermal stress" OR "vehicle strike" OR "tax reform" OR speciat* OR chromosome* OR earthquake OR "seed size" OR seafood OR phthalate OR cpue OR "vehicle collision" OR "green roof" OR "green roofs" OR "drought impact" OR dodd-frank OR deepwater OR xylem OR head-start* OR beverton-holt OR "agricultural intensification" OR "sustainable intensification" OR pesticide* OR herbicide* OR biochar OR rodenticide* OR pharma* OR "experimental drought" OR "social ratings" OR "social bonds" OR "thermal regimes" OR "rodent eradication" OR seaweed OR phenolog* OR tropospher* OR phospholipid* OR nano* OR metal* OR disaster OR mrna OR "biological invasions" OR smoking OR nicotine OR "competitive release" OR phytoscreening OR brewery OR mutualis* OR nonlethal OR germplasm OR "hazardous waste" OR ciliate* OR cultivar OR diabetes OR benz* OR "climate-smart agriculture" OR electroc*)) AND PUBYEAR > 2002 AND (LIMIT-TO (SUBJAREA , "ENVI")) AND (LIMIT-TO (LANGUAGE , "English"))

As indicated in the methods of the main paper, nuisance terms were refined by identifying unrelated papers in the first 200 hits of our WoS review.

Breakdown of studies captured by review

Search date	Search period	Databases	Number of hits	Number of additional unique hits	Number of potentially relevant papers assessed in full
29/10/18	2003-2018	Web of Science	9286	9286	269
16/11/18	2003-2018	Scopus	9691	5643	117
13/3/19	2018-2019	Web of Science and Scopus	3062	786	32
			Total unique hits=15715; total assessed in full =418		

Table 11. Summary of all studies captured in our review across both databases and all time periods.

Studies excluded from review

Exclusion reason	Studies
Studies reporting on a set of land use changes under a NNL policy but not disaggregating their data by land cover type	(Swenson & Ambrose 2007)
Studies reporting results of compensation / mitigation that are not or were not at the time associated with a NNL policy	(Edgar et al. 2005; Tischew et al. 2010; Brown et al. 2013; Alonso et al. 2014; Clare & Creed 2014; Lewis et al. 2017; Brower et al. 2018; Loder et al. 2018; McAdam et al. 2018; Power et al. 2018; Jarvis et al. 2019)
Studies reporting results of compensation / mitigation efforts targeting improving landscape connectivity, rather than biodiversity outcomes per se	(Claireau et al. 2018)
Comparisons between mitigation sites and unimpacted references without explicitly defining the size / characteristics of the impact site	e.g. (Gingerich & Anderson 2011; Pöll et al. 2016; Strain et al. 2017; Price et al. 2019; Sueltenfuss & Cooper 2019)
Studies reporting on trial offsets with areas significantly smaller than the impacted areas	(Valdez et al. 2017)
Studies reporting results from datasets already included in the review	(Harper & Quigley 2005b)
Studies reporting land use changes on either a sub-sample of habitats under a NNL policy, or a sample of habitats including but greater than those under a NNL policy, so that it is unknown to what degree outcomes are attributable to the NNL policy in question	(Kentula et al. 2004; Griffin & Dahl 2016; Guehlstorf & Martinez 2019; Reside et al. 2019)

Reviews of the effectiveness of no net loss policies that contain no original data	(Mbobi 2004; Kihslinger 2008)
Studies comparing impacted sites with offset sites before the implementation of offset actions	(Regnery et al. 2013)
Studies using matching to assess causal impact of species conservation banks (most of which do not have explicit NNL objectives (Gamarra & Toombs 2017))	(Sonter et al. 2019)
Studies that quantify all habitat losses, but only a subset of habitat gains (those attributable to mitigation banks, and not those attributable to permittee-responsible mitigation)	(Julian & Weaver 2019)
Studies reporting the outcomes of offsets but without comparing with impact / reference sites	(Baecher et al. 2018; Rueegger et al. 2018)

Table 12. Summary of all papers carefully considered but ultimately rejected from our review with associated justification

Appendix III Chapter 4 Supporting Information

Supplementary Introduction

Victoria's Native Vegetation Framework

The Native Vegetation Framework ran in Victoria from 2002-2013, when it was superseded by new native vegetation regulation associated with a slightly-altered offset policy. Under the Framework, applications to remove native vegetation were sent to local councils and processed through the planning system, with larger impacts and those to ecologically significant biodiversity conventionally referred to the state authorities for approval (this pathway comprised approximately one third of applications in 2010/2011; DSE 2012). Offsets required to compensate for clearance events that were referred to the State government were then registered.

Entering into an offset agreement committed landholders to both protect the registered native vegetation in perpetuity, and implement management actions (most commonly grazing exclusion and invasive plant or weed removal) under a 10-year management plan to deliver enhancements in biodiversity across that time period. Biodiversity gains were calculated using the 'habitat hectares' (HH) currency (Parkes et al. 2003), which allowed an estimate of the predicted gains in biodiversity over the 10-year management lifetime. These biodiversity gains translated into biodiversity credits which could be used directly to compensate for native vegetation clearance conducted by the same entity as that creating the offset ('first party offset'), or sold to other land clearers to offset their liabilities ('third party'). The State government implemented the Bushbroker programme, an initiative to create a regulatory market in offsets whereby land clearers could purchase offset credits to offset their native vegetation liabilities, which has since developed into a fully-fledged state offsetting sector brokered predominantly by private firms.

Habitat hectares

The HH approach is one of the original and most influential area*condition biodiversity metrics implemented in biodiversity offsetting systems around the world, which has served to underpin numerous derivative metrics such as England's Biodiversity Metric (Crosher et al. 2019a). To calculate a site's HH score, a qualified consultant conducts a site-based assessment, and scores the ecological quality of each ecological vegetation class (EVC) on the impacted site according to a number of ecological criteria (Table 13; Parkes et al. 2003). Each criteria is scaled so an EVC scores the maximum number of points if the ecological criteria are equivalent to those found at an intact reference patch of that EVC. Different ecological criteria contribute variably to the overall habitat score for the site, with the most ecologically important criteria contributing more to the overall habitat score. The total habitat score for the site adds up to a maximum of 100. This habitat score is then multiplied by the total area of the site in ha to yield the HH score. For example, 10ha of an intact reference EVC would score 10HH, and 10ha of a moderate condition EVC which achieved an overall habitat score of 50 would yield 5HH.

Ecological criteria	**Maximum value (sums to 100)**
Coverage of large trees	10
Canopy cover	5
Richness and degree of modifications of understory strata	25
Invasiveness and coverage of weeds	15
Plant recruitment	10
Coverage of organic litter	5
Total length of logs on site	5
EVC patch size	10
Neighbourhood / connectivity with surrounding vegetation patches	10
Distance to core area (vegetation patch >50ha)	5

Table 13. Components of the Habitat Hectares score. Adopted from Parkes et al. (2003)

Supplementary Methodology

Criteria for including offsets in evaluation

To decide which EVCs to include in our analysis, we used the information from the EVC benchmarks (https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks). We include all EVCs which, when in good condition, would be expected exceed the threshold of >2m vegetation height and >20% canopy cover based on the information provided in the EVC benchmarks (Table 14).

IBRA bioregion	EVC	Include in analysis
Central Victorian Uplands	Lowland Forest	yes
Central Victorian Uplands	Heathy Dry Forest	yes
Central Victorian Uplands	Grassy Dry Forest	yes
Central Victorian Uplands	Herb-rich Foothill Forest	yes
Central Victorian Uplands	Valley Grassy Forest	yes
Central Victorian Uplands	Heathy Woodland	yes
Central Victorian Uplands	Plains Grassy Woodland	yes
Central Victorian Uplands	Box Ironbark Forest	yes
Central Victorian Uplands	Rocky Chenopod Woodland	yes
Central Victorian Uplands	Hills Herb-rich Woodland	yes
Central Victorian Uplands	Grassy Woodland	yes
Dundas Tablelands	Damp Sands Herb-rich Woodland	yes
Dundas Tablelands	Plains Grassy Woodland	yes
East Gippsland Lowlands	Coast Banksia Woodland	yes
East Gippsland Lowlands	Damp Sands Herb-rich Woodland	yes
East Gippsland Lowlands	Banksia Woodland	yes
East Gippsland Lowlands	Lowland Forest	yes
East Gippsland Lowlands	Estuarine scrub	yes
East Gippsland Uplands	Dry Valley Forest	yes
East Gippsland Uplands	Grassy Woodland	yes
East Gippsland Uplands	Lowland Herb-rich Forest	yes
Greater Grampians	Grassy Dry Forest	yes
Greater Grampians	Valley Grassy Forest	yes
Greater Grampians	Heathy Woodland	yes
Gippsland Plain	Coast Banksia Woodland	yes
Gippsland Plain	Damp Sands Herb-rich Woodland	yes
Gippsland Plain	Sand Heathland	no
Gippsland Plain	Wet Heathland	no
Gippsland Plain	Coastal Saltmarsh	no
Gippsland Plain	Estuarine Wetland	no
Gippsland Plain	Lowland Forest	yes
Gippsland Plain	Riparian Forest	yes
Gippsland Plain	Herb-rich Foothill Forest	yes
Gippsland Plain	Damp Forest	yes
Gippsland Plain	Valley Grassy Forest	yes
Gippsland Plain	Heathy Woodland	yes
Gippsland Plain	Swamp Scrub	no

Gippsland Plain	Plains Grassy Woodland	yes
Gippsland Plain	Swampy Riparian Woodland	yes
Gippsland Plain	Plains Grassy Wetland	no
Gippsland Plain	Sand Forest	yes
Gippsland Plain	Sedge Wetland	no
Gippsland Plain	Mangrove Shrubland	no
Gippsland Plain	Plains Grassy Forest	yes
Gippsland Plain	Creekline Herb-rich Woodland	yes
Gippsland Plain	Grassy Woodland	yes
Gippsland Plain	Sedgy Swamp Woodland	yes
Gippsland Plain	Damp Heathy Woodland	yes
Gippsland Plain	Coastal Alkaline Scrub	yes
Gippsland Plain	Estuarine Flats Grassland	no
Gippsland Plain	Swampy Woodland	yes
Gippsland Plain	Estuarine scrub	yes
Glenelg Plain	Coastal Headland Scrub	no
Glenelg Plain	Coastal Mallee Scrub	yes
Goldfields	Grassy Dry Forest	yes
Goldfields	Herb-rich Foothill Forest	yes
Goldfields	Heathy Dry Forest	yes
Goldfields	Valley Grassy Forest	yes
Goldfields	Heathy Woodland	yes
Goldfields	Plains Grassy Woodland	yes
Goldfields	Box Ironbark Forest	yes
Goldfields	Alluvial Terraces Herb-rich Woodland	yes
Goldfields	Creekline Grassy Woodland	yes
Goldfields	Hillcrest Herb-rich Woodland	yes
Goldfields	Hills Herb-rich Woodland	yes
Goldfields	Sandstone Ridge Shrubland	yes
Goldfields	Plains Grassy Wetland	no
Goldfields	Grassy Woodland	yes
Goldfields	Plains Woodland	yes
Highlands - Northern Fall	Heathy Dry Forest	yes
Highlands - Northern Fall	Shrubby Dry Forest	yes
Highlands - Northern Fall	Grassy Dry Forest	yes
Highlands - Northern Fall	Herb-rich Foothill Forest	yes
Highlands - Northern Fall	Rocky Outcrop Shrubland	no
Highlands - Northern Fall	Damp Forest	yes
Highlands - Northern Fall	Montane Grassy Woodland	yes
Highlands - Northern Fall	Montane Riparian Woodland	yes
Highlands - Northern Fall	Swampy Riparian Woodland	yes
Highlands - Southern Fall	Lowland Forest	yes
Highlands - Southern Fall	Riparian Forest	yes
Highlands - Southern Fall	Heathy Dry Forest	yes
Highlands - Southern Fall	Shrubby Dry Forest	yes
Highlands - Southern Fall	Grassy Dry Forest	yes
Highlands - Southern Fall	Herb-rich Foothill Forest	yes
Highlands - Southern Fall	Damp Forest	yes
Highlands - Southern Fall	Wet Forest	yes
Highlands - Southern Fall	Shrubby Foothill Forest	yes
Highlands - Southern Fall	Valley Grassy Forest	yes
Highlands - Southern Fall	Riparian Thicket	yes
Highlands - Southern Fall	Box Ironbark Forest	yes
Highlands - Southern Fall	Creekline Grassy Woodland	yes
Highlands - Southern Fall	Swampy Riparian Woodland	yes
Highlands - Southern Fall	Valley Heathy Forest	yes

Highlands - Southern Fall	Grassy Forest	yes
Highlands - Southern Fall	Plains Grassy Forest	yes
Highlands - Southern Fall	Creekline Herb-rich Woodland	yes
Highlands - Southern Fall	Shrubby Damp Forest	yes
Highlands - Southern Fall	Damp Heathy Woodland	yes
Highlands - Southern Fall	Gully Woodland	yes
Lowan Mallee	Lowan Sands Mallee	yes
Lowan Mallee	Heathy Mallee	yes
Lowan Mallee	Treed Sandstone Ridge Shrubland	yes
Murray Fans	Riverine Chenopod Woodland	yes
Murray Fans	Grassy Riverine Forest	yes
Murray Fans	Riverine Grassy Woodland	yes
Murray Fans	Plains Woodland	yes
Murray Fans	Riverine Swamp Forest	yes
Murray Fans	Riverine Swampy Woodland	yes
Murray Fans	Sedgy Riverine Forest	yes
Murray Fans	Lignum Swampy Woodland	yes
Murray Mallee	Woorinen Sands Mallee	yes
Murray Mallee	Loamy Sands Mallee	yes
Murray Mallee	Chenopod Mallee	yes
Murray Mallee	Plains Woodland	yes
Murray Mallee	Woorinen Mallee	yes
Northern Inland Slopes	Heathy Dry Forest	yes
Northern Inland Slopes	Grassy Dry Forest	yes
Northern Inland Slopes	Herb-rich Foothill Forest	yes
Northern Inland Slopes	Valley Grassy Forest	yes
Northern Inland Slopes	Plains Grassy Woodland	yes
Northern Inland Slopes	Box Ironbark Forest	yes
Northern Inland Slopes	Alluvial Terraces Herb-rich Woodland	yes
Northern Inland Slopes	Creekline Grassy Woodland	yes
Northern Inland Slopes	Granitic Hills Woodland	yes
Northern Inland Slopes	Spring Soak Woodland	yes
Northern Inland Slopes	Low Rises Grassy Woodland	yes
Northern Inland Slopes	Shrubby Granitic-outwash Grassy Woodland	yes
Otway Plain	Coastal Saltmarsh	no
Otway Plain	Lowland Forest	yes
Otway Plain	Shrubby Dry Forest	yes
Otway Plain	Herb-rich Foothill Forest	yes
Otway Plain	Heathy Woodland	yes
Otway Plain	Swamp Scrub	yes
Otway Plain	Swampy Riparian Woodland	yes
Otway Plain	Plains Sedgy Wetland	yes
Otway Plain	Tall Marsh	no
Otway Plain	Plains Brackish Sedge Wetland	no
Otway Ranges	Wet Forest	yes
Otway Ranges	Cool Temperate Rainforest	yes
Otway Ranges	Shrubby Foothill Forest	yes
Otway Ranges	Shrubby Wet Forest	yes
Robinvale Plains	Riverine Chenopod Woodland	yes
Strzelecki Ranges	Lowland Forest	yes
Strzelecki Ranges	Damp Forest	yes
Strzelecki Ranges	Wet Forest	yes
Strzelecki Ranges	Cool Temperate Rainforest	yes
Strzelecki Ranges	Warm Temperate Rainforest	yes
Strzelecki Ranges	Shrubby Foothill Forest	yes
Strzelecki Ranges	Heathy Woodland	yes

Strzelecki Ranges	Swamp Scrub	yes
Strzelecki Ranges	Swampy Riparian Woodland	yes
Strzelecki Ranges	Riparian Scrub	yes
Victorian Alps	Montane Damp Forest	yes
Victorian Alps	Montane Riparian Thicket	yes
Victorian Alps	Sub-alpine Shrubland	no
Victorian Alps	Sub-alpine Woodland	yes
Victorian Alps	Sub-alpine Wet Heathland	no
Victorian Riverina	Plains Grassy Woodland	yes
Victorian Riverina	Floodplain Riparian Woodland	yes
Victorian Riverina	Ridged Plains Mallee	yes
Victorian Riverina	Riverine Chenopod Woodland	yes
Victorian Riverina	Plains Grassland	no
Victorian Riverina	Red Gum Swamp	yes
Victorian Riverina	Riverine Grassy Woodland	yes
Victorian Riverina	Plains Woodland	yes
Victorian Riverina	Riverine Swamp Forest	yes
Victorian Riverina	Riverine Swampy Woodland	yes
Victorian Riverina	Sedgy Riverine Forest	yes
Victorian Riverina	Lignum Swampy Woodland	yes
Victorian Riverina	Chenopod Grassland	no
Victorian Volcanic Plain	Lowland Forest	yes
Victorian Volcanic Plain	Herb-rich Foothill Forest	yes
Victorian Volcanic Plain	Swamp Scrub	yes
Victorian Volcanic Plain	Plains Grassy Woodland	yes
Victorian Volcanic Plain	Higher Rainfall Plains Grassy Woodland	yes
Victorian Volcanic Plain	Floodplain Riparian Woodland	yes
Victorian Volcanic Plain	Creekline Grassy Woodland	yes
Victorian Volcanic Plain	Lignum Swamp	yes
Victorian Volcanic Plain	Plains Grassy Wetland	no
Victorian Volcanic Plain	Plains Grassland	no
Victorian Volcanic Plain	Heavier-soils Plains Grassland	no
Victorian Volcanic Plain	Low-rainfall Plains Grassland	no
Victorian Volcanic Plain	Grassy Woodland	yes
Victorian Volcanic Plain	Stony Rises Woodland	yes
Victorian Volcanic Plain	Cane Grass Wetland	no
Victorian Volcanic Plain	Riparian Woodland	yes
Victorian Volcanic Plain	Plains Sedgy Wetland	no
Victorian Volcanic Plain	Stony Knoll Shrubland	yes
Victorian Volcanic Plain	Escarpment Shrubland	yes
Warrnambool Plain	Riparian Forest	yes
Warrnambool Plain	Herb-rich Foothill Forest	yes
Warrnambool Plain	Swamp Scrub	yes
Warrnambool Plain	Coastal Dune Scrub	no
Warrnambool Plain	Coastal Headland Scrub	no
Warrnambool Plain	Aquatic Herbland	no
Wimmera	Low Rises Woodland	yes
Wimmera	Lower Rainfall Shallow Sands Woodland	yes

Table 14. Summary of all of the EVCs included in offsets in the Victorian offset database, noting which would be expected to be classified as complete woody vegetation cover in our outcome dataset and therefore which are included in our evaluation

Data sources

Dataset	Details	Source
Outcome variables		
Woody vegetation cover	Landsat satellite imagery is used to estimate woody vegetation extent annually from 1998-2018. Each $25m^2$ pixel can take on a value of 0 (no woody vegetation), 1 (sparse woody vegetation, canopy cover between 5-19%), or 2 (minimum 20% canopy cover, with vegetation >2 metres high and a minimum area of 0.2 hectares. Sparse woody is defined as woody vegetation with a canopy cover between 5-19 per cent.	https://data.gov.au/data/dataset/d734c65e-0e7b-4190-9aa5-ddbb5844e86d/resource/bf7420cc-2ec7-470d-87ba-f0a2c0ea1b60/download/woody-vegetation-extent-v3_0-metadata_2018.pdf
Agricultural opportunity cost / ecological variables		
Rainfall	Mean annual precipitation from 1981-2010, 5km resolution.	http://www.bom.gov.au/climate/data-services/maps.shtml
Elevation	Digital terrain model, 20m resolution.	https://www.land.vic.gov.au/maps-and-spatial/spatial-data/vicmap-catalogue/vicmap-elevation
Slope	Slope, 20m resolution. Obtained using the 'Slope' command in QGIS using 20m digital terrain model as input.	https://www.land.vic.gov.au/maps-and-spatial/spatial-data/vicmap-catalogue/vicmap-elevation
Temperature	Mean annual temperature from 1961-1990, 2.5km resolution.	http://www.bom.gov.au/climate/data-services/maps.shtml
Soil carbon	Soil carbon in top 5cm of soil, 3 arc second (~30m) resolution.	https://www.clw.csiro.au/aclep/soillandlandscapegrid/GetData-GIS.html
Soil water capacity	Soil water capacity in top 5cm of soil, 3 arc second (~30m) resolution.	https://www.clw.csiro.au/aclep/soillandlandscapegrid/GetData-GIS.html
Remoteness / human pressure variables		
Distance from roads	Distance to major roads in 2016. 100m resolution raster. Values represent the distance (in kilometres) from the cell centre to the road recorded in Open Street Map.	https://www.worldpop.org/geodata/summary?id=17302
Remoteness	1km resolution remoteness raster. "ARIA+ measures remoteness in terms of access along the road network from populated localities to each of five categories of Service Centre based on population size. If one thinks of ARIA as based on the distances people have to travel to obtain services, then populated localities are where they are coming from, and Service Centres are where they are going to." Remoteness ranges from 1-15.	https://arts.adelaide.edu.au/hugo-centre/services/aria
Distance from conservation areas	100m resolution raster of distance to nearest conservation area in 2006. All conservation areas in 2006 are marked with a specific numerical code (9*) in the Victorian 2006 land use dataset (below). These conservation areas were rasterised,	See 'land use' below.

	and distance from areas was obtained using the proximity (raster distance) tool in QGIS.	
Other geographical variables		
IBRA subregions	IBRA 5.1 regions, the ecological regions used under the native vegetation framework framework (established in 2000). Polygons.	http://www.environment.gov.au/fed/catalog/search/resource/details.page?uuid=%7BA98C1395-42E9-43AE-9EE4-0083B0414658%7D
Land use	Spatial boundaries and land use for every cadastral land parcel in Victoria 2006, derived from the government's land use information system (polygon). Each parcel is marked with a specific numerical land use code, allowing the identification of all land parcels used for farming, nature conservation, forestry, and other land uses across the state.	https://discover.data.vic.gov.au/dataset/victorian-land-use-information-system-2006-2007 http://data.daff.gov.au/brs/data/warehouse/pe_abares99001806/GuidelinesLandUseMappingLowRes2011.pdf
Local government authorities (LGAs)	Spatial boundaries of LGAs and unincorporated Alpine resorts in Victoria, polygons.	https://www.land.vic.gov.au/maps-and-spatial/spatial-data/vicmap-catalogue/vicmap-admin
Burn scars from 2008 onwards	Collated data representing fire locations from Ward et al. (2019).	https://conbio.onlinelibrary.wiley.com/doi/full/10.1111/csp2.117

Table 15. Summary of the data layers used as covariates in the regressions and statistical matching, and justifications

Supplementary Results

Dataset description

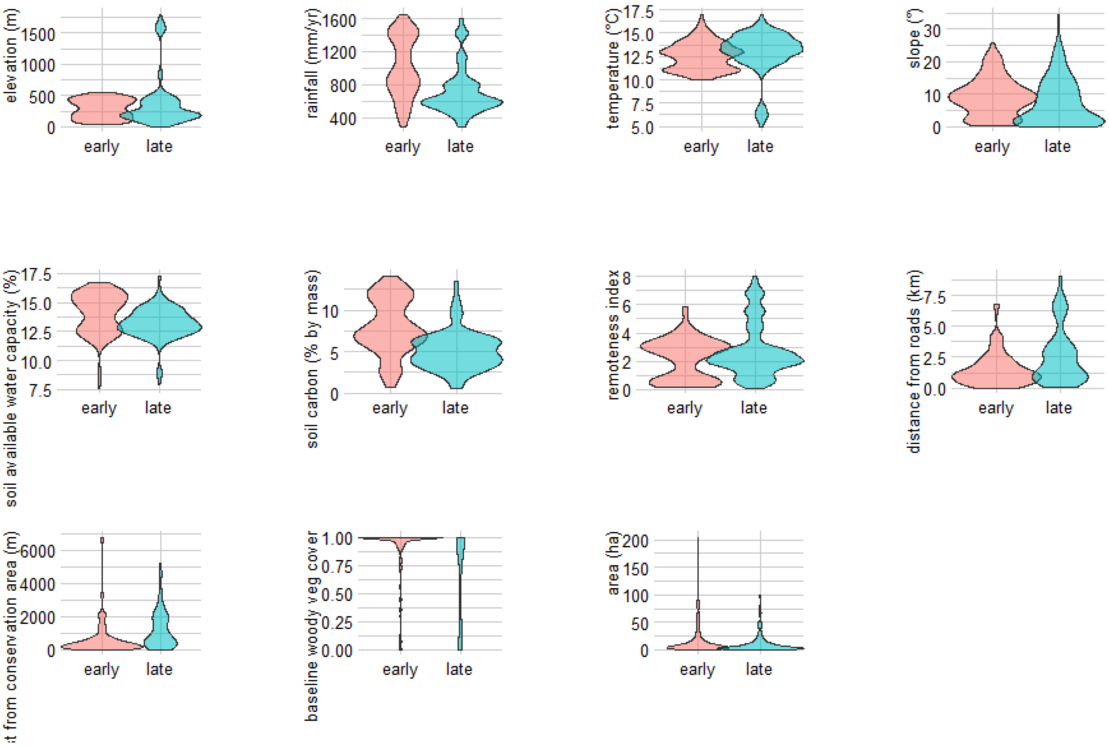

Figure 22. Comparison of the distribution of covariate values between the early (pink) and late offsets (blue) for each covariate.

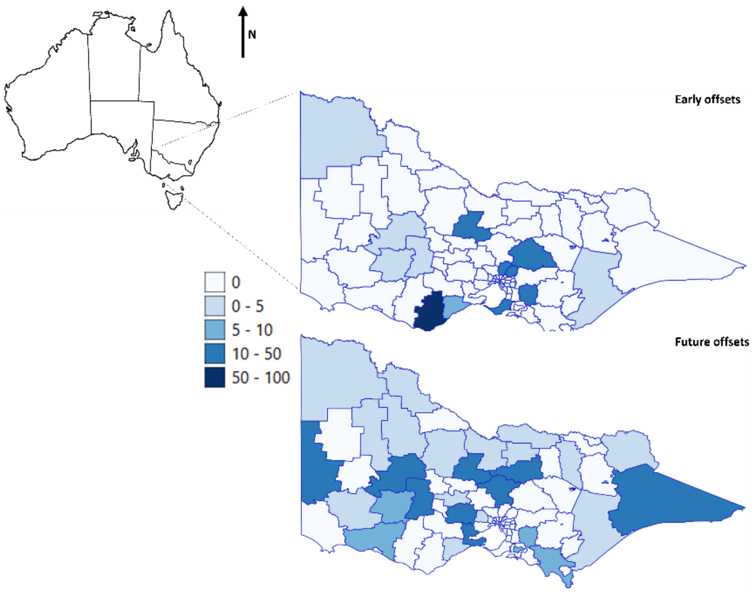

Figure 23. Spatial distribution of early and late offsets by Local Government Area. Darker blue indicates a higher number of offsets and lighter blue indicates a lower number of offsets.

Statistical matching

Smaller standardised mean differences in covariate values between offsets and matched controls indicate better matches, with standardised mean differences <0.1 considered high-quality matches (Greifer 2022). For all specifications, we match 1:1 and without replacement, as our pool of potential controls is vastly larger than treated observations. As a robustness check we conduct two commonly-used matching methods (Sonter et al. 2019; Devenish et al. 2022), and progressively reduce the caliper until there are no further gains in balance or until large numbers of observations are dropped. We use: a) nearest neighbour matching on propensity scores derived using logistic regression; b) Mahalanobis distance matching with exact matching on land use and a caliper of 1 standard deviation; c) and d) the same as b) but with 0.5 and 0.25 standard deviation calipers respectively.

The performance of our alternative matching specifications is detailed in Figure 24 and Figure 25.

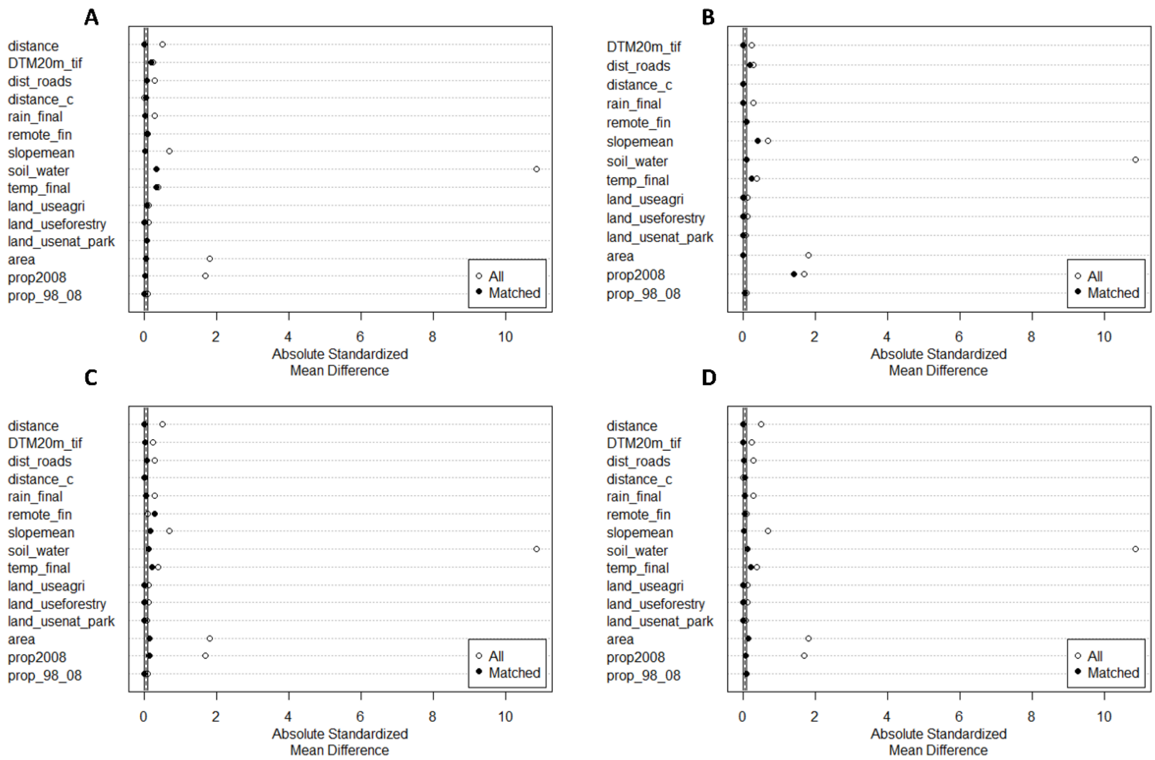

Figure 24. Loveplots showing the standardised mean difference between full and matched datasets and treated observations (regeneration offsets) under various matching specifications. All specifications achieve full matching

of treated and control observations: A) 1:1 propensity score matching without replacement; B) 1:1 Mahalanobis distance matching with 1 standard deviation calipers and exact matching for the land use for each land parcel; C) As B, with 0.5 standard deviation calipers; D) As B, with 0.25 standard deviation calipers

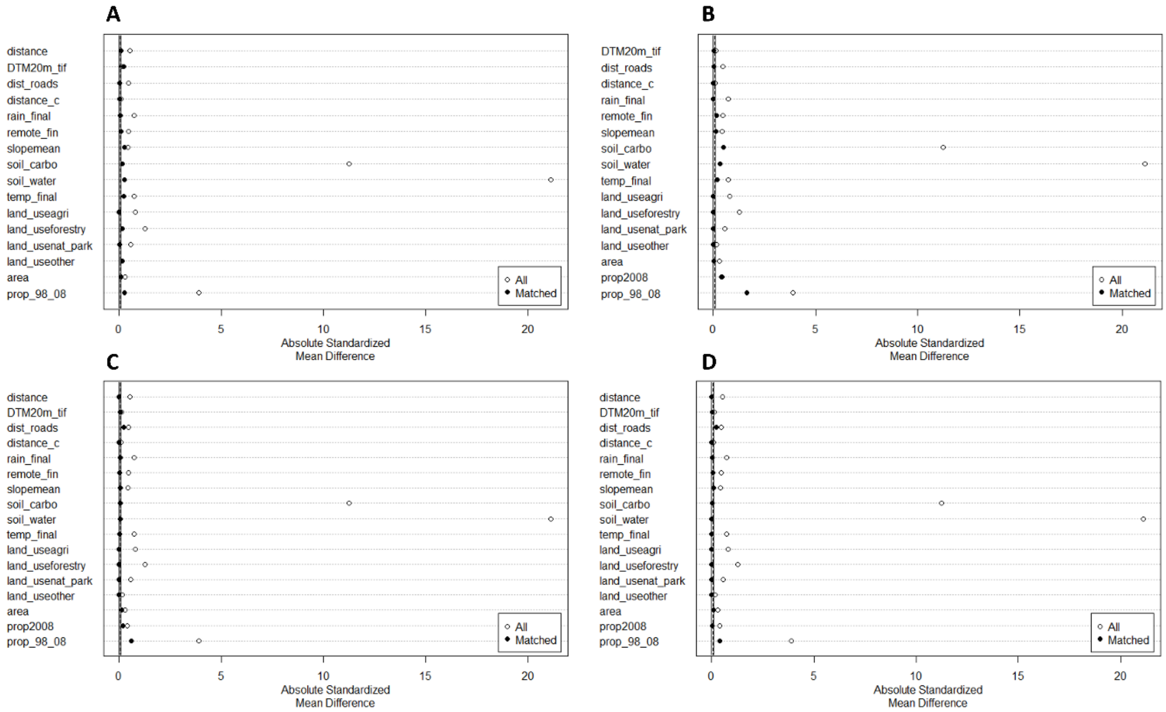

Figure 25. Loveplots showing the standardised mean difference between full and matched datasets and treated observations (avoided loss offsets) under various matching specifications: A) 1:1 propensity score matching without replacement (complete matching of treated and controls); B) 1:1 Mahalanobis distance matching with 1 standard deviation calipers and exact matching for the land use for each land parcel (4 treated observations unmatched and dropped); C) As B, with 0.5 standard deviation calipers (4 treated observations unmatched and dropped); D) As B, with 0.25 standard deviation calipers (4 treated observations unmatched and dropped)

Background trend analysis

To test for parallel trends before implementing the difference-in-difference analysis, we followed the methods of Devenish et al. (2022). We regressed the pre-intervention woody vegetation cover data against the interaction between whether the site is from the control or intervention sample, and year. If the interaction is significant, it implies that there is a significantly different time trend between the offsets and controls. Regression outputs are given in Table 16.

Parameter	Early offsets vs future offsets	Early offsets vs matched non-adopters
Intercept	7.05 (8.48)	4.92
Year	-0.00 (0.00)	-0.00 (0.01)
Treatment	-0.31 (15.27)	0.74 (16.57)
Year:Treatment	0.00 (0.01)	-0.00 (0.01)

*Table 16. Regression outputs for the regressions testing the parallel trends assumptions. Values represent regression coefficients, standard errors in brackets. Significance (p<0.05) is indicated by **

Regression outputs

Here we present full outputs of our regression comparing changes in woody vegetation cover between early regeneration offsets and matched controls (Table 17), and early regeneration offsets and late regeneration offsets (Table 18).

Parameter	**Coefficient (std errors)**
(Intercept)	0.11 (0.02) ***
Time since policy	-0.00 (0.00)
Before/after intervention dummy	-0.00 (0.02)
Treatment/control dummy	0.06 (0.04)
Baseline woody vegetation cover	0.80 (0.02) ***
Distance from roads	0.02 (0.01)
Elevation	-0.03 (0.02)
Rainfall	0.01 (0.02)
Remoteness	-0.01 (0.02)
Slope	0.01 (0.01)
Temperature	-0.01 (0.03)
Soil water	0.01 (0.01)
Distance from conservation area	-0.04 (0.01) *
Area	-0.01 (0.01)
Land use (conservation area)	0.03 (0.06)
X	-0.02 (0.02)
Y	-0.00 (0.03)
Time since policy: before/after dummy	0.01 (0.00) ***
Time since policy: treatment/control dummy	-0.00 (0.00)
before/after dummy: treatment/control dummy	-0.11 (0.03) ***
Time since policy: bef/aft: treat/control	0.03 (0.00) ***
AIC	-1535.41
Num. obs.	1944

Num. groups: HH_PAI	68
Var: HH_PAI (Intercept)	0.01
Var: Residual	0.02
R^2	0.73

*Table 17. Regression outputs for our linear mixed effects model estimating the impact of offset management on woody vegetation cover, comparing regeneration offsets with matched counterfactual land parcels. Coefficient estimates and associated standard errors are presented. For the categorical Land use variable, the baseline land use against which alternatives are compared is agriculture. P-values are denoted by stars: *= p<0.05, ***=p<0.001*

Parameter	**Coefficient (std errors)**
(Intercept)	0.06 (0.02) **
Time since policy	-0.00 (0.00)
Before/after intervention dummy	-0.04 (0.01) **
Treatment/control dummy	0.09 (0.03) **
Baseline woody vegetation cover	0.84 (0.01) ***
Distance from roads	0.01 (0.01)
Elevation	-0.04 (0.02)
Rainfall	-0.01 (0.03)
Remoteness	0.02 (0.02)
Slope	0.00 (0.01)
Temperature	-0.03 (0.02)
Soil water	0.01 (0.01)
Distance from conservation area	-0.03 (0.01) **
Area	-0.01 (0.00) **
Land use (forestry)	0.14 (0.06) *
Land use (conservation area)	-0.08 (0.07)
Land use (other)	0.02 (0.03)
X	-0.00 (0.01)
Y	0.06 (0.02) *
Time since policy: before/after dummy	0.03 (0.00) ***
Time since policy: treatment/control dummy	0.00 (0.00)
before/after dummy: treatment/control dummy	-0.09 (0.02) ***
Time since policy: bef/aft: treat/control	0.01 (0.00) **
AIC	-2456.12
Num. obs.	3150
Num. groups: HH_PAI	52
Var: HH_PAI (Intercept)	0.01
Var: Residual	0.02
R^2	0.80

*Table 18. Regression outputs for our linear mixed effects model estimating the impact of offset management on woody vegetation cover, comparing early regeneration offsets with late regeneration offsets. Coefficient estimates and associated standard errors are presented. For the categorical Land use variable, the baseline land use against which alternatives are compared is agriculture. P-values are denoted by stars: *= p<0.05, ***=p<0.001*

Sensitivity analyses

Effects of varying the threshold for assigning offsets to 'regeneration' or 'avoided loss'

In our main analysis, we chose the threshold of a proportion native vegetation cover of 0.95 to assign offsets to the regeneration or avoided loss category, because this retains an effective sample size for the regeneration offsets. As we lower the threshold, the sample size declines (0.9, N early offsets=37; 0.8, N early offsets=29), and the mean woody vegetation cover in our offsets declines. This means that there is greater potential for woody vegetation cover to increase over the 10-year evaluation period. Therefore, as we reduce the sample size and lower the threshold, the effect size of the impacts of offsets on native vegetation increases slightly (these offsets that start with lower baseline woody vegetation cover experience larger increases in woody vegetation cover than offsets starting with a higher baseline woody vegetation cover).

The regression outputs for the diff-in-diff regression comparing the change in woody vegetation cover in early regeneration offsets and matched non-adopter parcels (using our core model) at varying baseline woody vegetation thresholds is presented in Table 19. When the threshold for regeneration offsets is set at a baseline proportion woody vegetation cover <0.9, early offsets are associated with an increase in woody vegetation cover of 3.09% per year relative to controls. When the threshold for regeneration offsets is set at a baseline proportion woody vegetation cover <0.8, early offsets are associated with an increase in woody vegetation cover of 4.04% per year relative to controls.

Parameter	**Coefficient (std errors), threshold = 0.9**	**Coefficient (std errors), threshold = 0.8**
(Intercept)	0.08 (0.03) **	0.03 (0.03)
Time since policy	-0.00 (0.00)	-0.00 (0.00)
Before/after intervention dummy	-0.01 (0.02)	0.02 (0.03)
Treatment/control dummy	0.05 (0.04)	0.09 (0.05)
Baseline woody vegetation cover	0.82 (0.03) ***	0.88 (0.03) ***

Distance from roads	0.08 (0.02) ***	0.01 (0.02)
Elevation	-0.01 (0.02)	0.00 (0.04)
Rainfall	0.01 (0.03)	0.02 (0.03)
Remoteness	-0.08 (0.03) **	-0.03 (0.02)
Slope	-0.01 (0.02)	-0.03 (0.02)
Temperature	-0.04 (0.04)	-0.05 (0.05)
Soil water	0.02 (0.02)	0.01 (0.04)
Distance from conservation area	-0.03 (0.02)	-0.03 (0.02)
Area	-0.01 (0.01)	-0.03 (0.01) **
X	-0.02 (0.02)	-0.01 (0.02)
Y	0.02 (0.05)	0.07 (0.06)
Time since policy: before/after dummy	0.02 (0.00) ***	0.01 (0.01) **
Time since policy: treatment/control dummy	0.00 (0.00)	0.00 (0.01)
before/after dummy: treatment/control dummy	-0.12 (0.03) ***	-0.13 (0.04) **
Time since policy: bef/aft: treat/control	0.03 (0.01) ***	0.04 (0.01) ***
AIC	-910.79	-455.64
Num. obs.	1332	972
Num. groups: HH_PAI	50	39
Var: HH_PAI (Intercept)	0.01	0.01
Var: Residual	0.02	0.03
R^2	0.73	0.72

*Table 19. Regression outputs for our linear mixed effects model estimating the impact of offset management on woody vegetation cover, comparing early regeneration offsets with matched non-adopters, and assuming different thresholds for categorising regeneration offsets. Coefficient estimates and associated standard errors are presented. P-values are denoted by stars: *= p<0.05, ***=p<0.001*

The regression outputs for the diff-in-diff regression comparing the change in woody vegetation cover in early regeneration offsets and future regeneration offsets at varying baseline woody vegetation thresholds is presented in Table 20. When the threshold for regeneration offsets is set at a baseline proportion woody vegetation cover <0.9, early offsets are associated with an increase in woody vegetation cover of 1.85% per year relative to controls. When the threshold for regeneration offsets is set at a baseline proportion woody vegetation cover <0.8, early offsets are associated with an increase in woody vegetation cover of 2.09% per year relative to controls.

Parameter	Coefficient (std errors), threshold = 0.9	Coefficient (std errors), threshold = 0.8
(Intercept)	0.05 (0.02) *	0.01 (0.02)
Time since policy	-0.00 (0.00) *	-0.01 (0.00) **
Before/after intervention dummy	-0.04 (0.01) **	-0.04 (0.02) *
Treatment/control dummy	0.07 (0.04)	0.09 (0.04) *
Baseline woody vegetation cover	0.88 (0.02) ***	0.94 (0.02) ***
Distance from roads	0.01 (0.01)	0.01 (0.01)
Elevation	-0.03 (0.03)	0.04 (0.03)
Rainfall	-0.02 (0.03)	-0.02 (0.03)
Remoteness	0.02 (0.02)	0.03 (0.03)
Slope	-0.00 (0.01)	-0.01 (0.01)
Temperature	-0.03 (0.03)	0.02 (0.03)
Soil water	0.01 (0.01)	-0.00 (0.02)
Distance from conservation area	-0.03 (0.01) **	-0.03 (0.01) *
Area	-0.02 (0.01) ***	-0.02 (0.01) **
Land use (forestry)	0.14 (0.07) *	0.20 (0.08) *
Land use (conservation area)	-0.21 (0.12)	
Land use (other)	0.00 (0.04)	0.01 (0.04)
X	-0.00 (0.02)	0.00 (0.02)
Y	0.07 (0.03) *	-0.01 (0.04)
Time since policy: before/after dummy	0.03 (0.00) ***	0.04 (0.00) ***
Time since policy: treatment/control dummy	0.00 (0.00)	0.00 (0.00)
before/after dummy: treatment/control dummy	-0.10 (0.03) ***	-0.10 (0.03) **
Time since policy: bef/aft: treat/control	0.02 (0.00) ***	0.02 (0.01) ***
AIC	-1721.09	-1252.34
Num. obs.	2538	2070
Num. groups: HH_PAI	50	45
Var: HH_PAI (Intercept)	0.01	0.01
Var: Residual	0.03	0.03
R^2	0.77	0.72

*Table 20. Regression outputs for our linear mixed effects model estimating the impact of offset management on woody vegetation cover, comparing early regeneration offsets with future offsets, and assuming different thresholds for categorising regeneration offsets. Coefficient estimates and associated standard errors are presented. For the categorical Land use variable, the baseline land use against which alternatives are compared is agriculture. P-values are denoted by stars: *= p<0.05, ***=p<0.001*

Effects of removing sites burned by wildfires during the analysis period

Removing the landholding containing offsets which experienced catastrophic loss of woody vegetation cover in the 2009 Black Saturday fires altered the outputs of the regression models, as the rapid vegetation regrowth in these offsets caused by the fire but coincident with the onset of offset management in 2008 contributed in increasing the effect size of offset management. Full results of the core regression analyses excluding these offsets is in Table 21.

Parameter	Coefficient (std errors), early offsets versus matched non-adopters	Coefficient (std errors), early offsets versus future offsets
(Intercept)	0.14 (0.03) ***	0.06 (0.02) ***
Time since policy	0.00 (0.00)	-0.00 (0.00)
Before/after intervention dummy	-0.03 (0.02)	-0.04 (0.01) **
Treatment/control dummy	0.02 (0.04)	0.09 (0.03) **
Baseline woody vegetation cover	0.79 (0.02) ***	0.84 (0.01) ***
Distance from roads	0.05 (0.02) **	0.01 (0.01)
Elevation	-0.01 (0.02)	-0.03 (0.03)
Rainfall	-0.01 (0.03)	-0.02 (0.03)
Remoteness	-0.01 (0.02)	0.01 (0.02)
Slope	-0.01 (0.01)	0.00 (0.01)
Temperature	-0.03 (0.03)	-0.03 (0.02)
Soil water	-0.00 (0.02)	0.00 (0.01)
Distance from conservation area	-0.06 (0.02) ***	-0.03 (0.01) ***
Area	-0.01 (0.01)	-0.01 (0.00) **
X	-0.01 (0.02)	-0.00 (0.01)
Y	-0.01 (0.04)	0.05 (0.02) *
Time since policy: before/after dummy	0.01 (0.00) **	0.03 (0.00) ***
Time since policy: treatment/control dummy	-0.00 (0.00)	0.00 (0.00)
before/after dummy: treatment/control dummy	-0.04 (0.03)	-0.04 (0.02)
Time since policy: bef/aft: treat/control	0.02 (0.00) ***	0.01 (0.00)
AIC	-1664.57	-2603.80
Num. obs.	1692	3060
Num. groups: HH_PAI	61	51
Var: HH_PAI (Intercept)	0.01	0.01
Var: Residual	0.02	0.02
R^2	0.77	0.72

*Table 21. Regression outputs for our linear mixed effects models estimating the impact of offset management on woody vegetation cover, excluding sites burned by wildfires. Coefficient estimates and associated standard errors are presented. P-values are denoted by stars: *= p<0.05, ***=p<0.001*

Appendix IV Chapter 6 Supporting Information

Introduction to the "Biodiversity metric 2.0"

The Metric (Crosher et al., 2019a, 2019b, originally described in Treweek et al., 2010) estimates biodiversity value through using the area, condition and other attributes of the underlying habitats as proxies. It represents the sum of all of the biodiversity scores for each of the underlying habitat types within the development site. The formula used to estimate the baseline biodiversity value of the site is:

$$\sum area * habitat\ distinctiveness * condition * strategic\ location * connectivity$$

where *area* is the absolute area of each constituent habitat in ha, *distinctiveness* is automatically determined by the Metric with each potential habitat type in the UK Habitat classification given a distinctiveness score a priori, *condition* is determined by the ecological assessor for each habitat using a categorical scale from 'poor' to 'good' (this judgement is supported by a condition guidance document advising on the distinctions between condition categories for each different habitat (Crosher et al. 2019b)), *strategic location* is determined using a categorical scale from 'low' to 'high' representing the degree to which the site is spatially located within an area identified as contributing to the local biodiversity strategy, and *connectivity* represents the ecological connectivity of the habitat (at the point of this evaluation, the tool used to determine this parameter was not yet fully functioning and was therefore not used in planning applications).

The formula for predicting post-development biodiversity value is:

$$\sum area * habitat\ distinctiveness * condition * strategic\ location * connectivity \\ * restoration\ difficulty * time\ to\ target\ condition * offsite\ risk$$

As with the baseline formula, the overall value is the sum of the units across all habitat types. The first five parameters are determined in the same way as for the baseline, albeit based on the predicted future state of the habitats after the development is complete (i.e. not based on real data at the time the calculation is made). *Restoration difficulty* is automatically determined for each desired habitat type, with each being given a difficulty score a priori, *time to target condition* is a temporal discounting term which imposes a penalty for the longer into the future the predicted biodiversity gains will be delivered (this is bounded at a maximum of '≥32 years' and pre-set for each habitat type and dependent on whether that habitat results from creation from scratch or from enhancing existing habitats), and *offsite risk* represents a penalty if the units are being delivered outside the local authority of the development. For a complete guide to the Metric see Crosher et al. (2019a, 2019b).

In addition to the biodiversity units delivered by habitats, the Metric generates 'hedgerow' and 'river' units, which are calculated using similar principles in separate sections of the Metric spreadsheet tool (Crosher et al. 2019a).

Characteristics of councils in the database

The LPAs which we included in the database and the properties of their mandatory BNG-type requirements are outlined in Table 22. We identified appropriate councils by speaking with representatives from Defra and industry associations who were able to direct us towards relevant councils. Construction of the database is ongoing, and more councils will be added as they are identified. In total, 16 councils were identified with the potential to be implementing relevant policies; of these, only the six councils included in our database turned out to have appropriate policies. By appropriate policies, we mean that all six councils implement a mandatory BNG or No Net Loss (NNL) requirement for new developments for planning permission to be granted, and the method used the assess whether the criteria has been met is through a BNG assessment and Metric calculation using the Biodiversity Metric 2.0.

Council	Local authority type	Biodiversity metric	Required uplift in units	Development types included	Local offsetting fund
Tunbridge Wells Borough Council	Non-metropolitan district	Defra Metric 2.0	10%	All newbuilds	No
Leeds City Council	Metropolitan district	Defra Metric 2.0	0% (i.e. NNL) if biodiversity units delivered on-site, 10% if developer chooses to purchase biodiversity units from the council	All newbuilds	Yes
South Oxfordshire District Council	Non-metropolitan district	Defra Metric 2.0	0% (i.e. NNL)	'Major developments' (i.e. >10 dwellings and other infrastructure)	Yes
Vale of White Horse District Council	Non-metropolitan district	Defra Metric 2.0	0% (i.e. NNL)	'Major developments' (i.e. >10 dwellings and other infrastructure)	Yes
West Oxfordshire District Council	Non-metropolitan district	Defra Metric 2.0	10% for large-scale developments; 0-10% negotiated on a discretionary basis for other types of developments	All newbuilds	No; accept third-party offsetting
Cornwall Council	County	Defra Metric 2.0	10%	'Major developments' (i.e. >10 dwellings and other infrastructure)	Yes

Table 22. Summary of the characteristics of the local councils and BNG-equivalent requirements included in our dataset

These councils implement mandatory BNG/NNL in a way that is very similar to that expected nationally under the Environment Bill. The largest difference between their implementation and the government's expected nationwide implementation is that the market for biodiversity credits is still immature, and it is possible that a more mature market for biodiversity units offering accredited biodiversity units to developers may alter developers' behaviour relative that observed in our dataset. However, five of the councils in the dataset are already facilitating developers to pass on their biodiversity liabilities to the councils or third-parties via offsetting funds or habitat banks, so the dataset represents the most comprehensive and empirically-grounded evaluation of the market size to date.

Database of Biodiversity Net Gain assessments

Our authorship team contains biodiversity officers for three of the councils in our database, and the co-Director of an AONB Partnership spanning a fourth council. After submitting the first draft of the manuscript, we also developed equivalent relationships with biodiversity officers for the remaining two councils. As a result, our project team receives all of the Net Gain applications within our councils as a routine part of their work. We assembled all of the BNG applications for these six councils from January 2020-February 2021. In order to validate that our authorship team had access to all relevant BNG applications, we conducted a set of manual data validation checks. We manually searched all of the relevant (i.e. new builds, not retrofits or other planning application types) planning applications on the Leeds City Council planning portal for the month of July, and found no additional relevant applications. We manually searched all of the relevant (i.e. 'major developments') planning applications on the Vale of White Horse and South Oxfordshire District Council planning portal for the period January-August 11, and found no additional applications. Lastly, we went through all of the relevant planning applications for all of our councils for the month of September, and identified one additional application.

For each project for which a BNG assessment was identified, we extracted all of the ecology-related documentation from the relevant online planning portal (e.g. ecological impact assessments, preliminary ecological appraisals, biodiversity net gain reports). We extracted the following information from each application:

- Complete biodiversity metric information for the baseline and post-development scenarios

- Information about the application, including client name, ecological consultancy name, the type of development and spatial location

Our search identified 90 potential developments referencing BNG assessments. We excluded projects that provided insufficient information about their BNG assessments (e.g. provided the baseline assessment but not the post-development), projects unrelated to built infrastructure, and projects using the earlier August 2019 version of the biodiversity Metric. Some projects provided BNG reports which did not contain complete information. In these cases, we took several courses of action. If the diagrams of the landscape plans before and after the development were sufficiently clear (projects 10 and 12), we digitalised the landscape plans and measured out the area of habitats in the landscape plans using image manipulation software ImageJ (https://imagej.net). If the habitats were presented as a block colour, we used thresholding to measure the area within those habitats, otherwise we measured habitat patches manually. Using these manually-derived areas and combining it with information provided in the BNG reports, we were able to estimate the inputs into the Metric and add the estimated Metric information to our database. If this method yielded biodiversity unit estimates that were far off those reported in the BNG report (project 11), we submitted a Freedom of Information request to the relevant council to obtain the raw Metric spreadsheet submitted with the application. In response to the Freedom of Information request, the council reported that it did not possess the required Metric spreadsheet. Lastly, we excluded six rejected projects and one outlier (described in main text) from the calculations of aggregate changes in habitat area under the policy.

Three of our councils (Leeds, Vale of White Horse and South Oxfordshire) have mandatory NNL policies rather than the 10% Net Gain expected to be made mandatory nationally under the Environment Bill. For biodiversity units delivered on-site (i.e. within the Metric calculations submitted by developers), we made no adjustment to account for the lower biodiversity threshold. This means that for developments that achieved NNL within their development footprint in these councils, the total number of units delivered by these projects in our database falls up to 10% short of what would be required by national mandatory BNG. This applies to a single project in our database, which would therefore be expected under the national mandatory policy to deliver an additional 4.5 biodiversity units. Within these councils, if developers fall short of the mandatory NNL requirement, they have the option of making up the shortfall through payments to the councils to pass on their biodiversity liability. In these cases, in our database, we assumed that a 10% uplift in biodiversity units would be required from the offset payments, aligning our database with what would be expected under the mandatory national policy.

At the time of writing this manuscript, one major offset (project 401) is currently under negotiation. In our dataset, it is assumed that the offset delivers units up to a 10% net gain, but it is possible that a higher value is later negotiated. Current documentation suggests an offset delivering units up to a 25% net gain may be secured. If this were the case, the total units delivered by offsets in our dataset would rise from 171 to 235; bringing the total percentage of units delivered by offsets from 4.5-6%.

Grassland expert survey

In order to explore the role of professional judgements on the outcomes of the policy, we surveyed a sample of specialised grassland experts to identify whether they would recreate the grassland habitat type and condition reported in the applications when assessing the base information provided accompanying those applications. For every application in our database, we extracted information relating to the quality of their grassland assessments, including whether they:

- presented disaggregated grassland results for each grassland patch;
- reported the dominant grassland species in the sward;
- reported the % cover of dominant grass species;
- reported the full list of species found in the assessed grasslands;
- reported the % herb cover;
- reported the number of quadrats used if sampling priority grassland;

- used the initial Phase 1 habitat classification scheme and then subsequently converted it into the UKHab classification scheme, or whether they assessed the grassland in UKhab;
- reported the month of the grassland survey (best practice grassland surveys are to be conducted from April-October (JNCC 2003)).

The projects we chose to include in our survey were chosen because they reflected a broad range in the quality of the grassland assessments included in our database (with quality judged by the number of criteria from the top five in the above list they met). Our experts were selected through snowball sampling, with the criteria that they be experienced grassland experts (>10 years professional experience, or PhDs in grassland ecology) with competencies with Phase 1 habitat surveys, the UK Habitat classification system, and the new Defra Metric condition guidance. The highly specialised nature of the expertise required to respond to our survey justifies our small sample size.

For our survey, we took all of the grassland information from our five selected applications and stripped out all of the information relating to the final condition score, or the final UKHab classification type allocated to each grassland patch. For each application, we then asked our participants the following: 1) what (UK Hab) grassland type would you assign to the grasslands? 2) what condition score would you assign the grasslands? 3) what is your degree of confidence in your judgement, given the available information? 4) what additional information would you like to see in order to be more confident in your judgement? We used open-ended rather than closed questions to simulate the situation faced by planners reviewing BNG assessments, in which there are no constraints on which habitat type each patch could be classified as. Our survey document can be found in the supporting information for this study.

Our results reveal mixed levels of agreement both between grassland experts and the judgements presenting in the BNG assessments, and between grassland experts (Table 23, Table 24).

Site and grassland number	Quality criteria met (/5)	Survey month	modified grassland	other neutral grassland	amenity grassland	cropland	should be two habitats	Insufficient info to make judgement
1, G1	5	July, September	6	1	0	0	0	0
1, G2	5	July, September	2	5	0	0	0	0
2, G1	2	March	5	0	2	0	0	0
2, G2	2	March	0	4	0	0	3	0
3, G1	3	March	3	2	0	0	0	2
3, G2	3	March	5	0	0	0	0	2
3, G3	3	March	4	1	0	0	0	2
4, G1	1	April, October, February	3	0	0	3	0	1
5, G1	1	March	6	0	1	0	0	0
5, G2	1	March	4	2	0	0	0	1
5, G3	1	March	3	3	0	0	0	1
5, G4	1	March	4	2	0	0	0	1
5, G5	1	March	4	2	0	0	0	1

Table 23. Count of the number of experts judging each grassland in the survey to be a given grassland type. Green boxes represent the judgement that was made in the actual grassland survey provided alongside the BNG assessment. The different categories of answer represent the responses of experts to the open-ended questions in our survey.

Site and grassland number	Quality criteria met (/5)	Survey month	poor	fairly poor	moderate	n/a - Agricultural	should be two habitats	insufficient info to make judgement
1, G1	5	July, September	7	0	0	0	0	0
1, G2	5	July, September	3	0	4	0	0	0
2, G1	2	March	7	0	0	0	0	0
2, G2	2	March	2	0	2	0	3	0
3, G1	3	March	4	0	2	0	0	1
3, G2	3	March	6	0	0	0	0	1
3, G3	3	March	5	0	1	0	0	1
4, G1	1	April, October, February	4	0	0	2	0	1
5, G1	1	March	7	0	0	0	0	0
5, G2	1	March	4	0	3	0	0	0
5, G3	1	March	3	0	3	0	0	1
5, G4	1	March	6	0	1	0	0	0
5, G5	1	March	6	0	1	0	0	0

Table 24. Count of the number of experts judging each grassland in the survey to be a given condition level. Green boxes represent the judgement that was made in the actual grassland survey provided alongside the BNG assessment. The different categories of answer represent the responses of experts to the open-ended questions in our survey.

Review of long-term governance measures

To explore the long-term governance mechanisms proposed under BNG, we surveyed the following documents in consultation with the Net Gain team at Defra:

- Biodiversity Net Gain Economic Impact Assessment (Defra 2019b)
- UK Government Net Gain consultation document (Defra 2018c)
- UK Government response to consultation (Defra 2019a)
- Biodiversity Net Gain – good practice principles for development (Baker et al. 2019)
- Biodiversity metric 2.0 Technical Supplement (Crosher et al. 2019b)
- Biodiversity metric 2.0 User guide (Crosher et al. 2019a)
- British Standard: draft process for designing and implementing Biodiversity Net Gain – specification (BSI 2020)
- RPC opinion: Biodiversity Net Gain (Regulatory Policy Committee 2020)
- Env Bill clauses themselves and the explanatory notes for the Environment Bill (Defra 2020b, 2020c)

We extracted all references to achieving compliance, monitoring, long-term capacity of councils or other governance-related issues arising in the documents, and thereby summarised the current publicly-available information on the long-term governance of BNG. To identify the relevant sections in these documents, we searched them for the words: capacity, skills, knowledge, shortfall*, monitor*, compliance, comply*, enforce*, governance, fund*, financ*, contingen*. The purpose of our review was to collate the publicly-available information on BNG, and as such we took all the information included in the document list at face value. Our methodology was not a critical review.

Appendix V Chapter 7 Supporting Information

England's cumulative carbon budget

To determine England's cumulative carbon budget for remaining within 1.5°C, we follow the simplest methodology outlined in Jackson (2021). We assume a global population of 10 billion (https://interactives.prb.org/2021-wpds/) and an English population of 60 million by 2050 (ONS 2019). We then multiply the total remaining global carbon budget for a 50% probability of remaining below 1.5°C (420GtCO_2e) outlined in the 2021 Global Carbon Budget (Friedlingstein et al. 2021) by the proportion of the global population in 2050 represented by England (420*0.006=2.52Gt). We note that there is a strong case for a lower cumulative carbon budget on equity grounds, given England's historical contribution to global emissions - Jackson (2021) argues for an adjustment accounting for England's current emissions being considerably higher than the global per capita average that leads to a 'fair carbon budget' of 2.4Gt for the UK, or approximately 2.1Gt for England.

To determine the cumulative carbon budget for England implicit in the UK government's net zero strategy from 2020, we take the values of the 3rd-6th carbon budgets (multiplying the 3rd carbon budget by 3/5 as it runs from 2018-2022 and our analysis starting year is 2020), and from 2037 onwards we sum the annual emissions reported under the Balanced Net Zero decarbonisation scenario (CCC 2020b). We then multiply all carbon budgets by the expected proportion of the UK population represented by England in 2050 based on ONS population projections, approximately 60/70 million (ONS 2019).

Modelling the biodiversity impacts of housing expansion

To explore the potential biodiversity impacts of housing expansion without policy action (and therefore to estimate the required effectiveness of BNG and species mitigation legislation in order for housing expansion not to undermine the 2030 national species abundance target), we used the projections of urban expansion from

2006-2031 developed in Eigenbrod et al. (2011), and combine these land use projections with an ensemble of species distribution models representing the ranges of 100 species of conservation priority in the UK, to generate an estimate of the overall and annual impact of predicted urban expansion on English biodiversity to 2031.

Housing expansion model

To model England's housing requirements we follow the methodology proposed by Eigenbrod et al. (2011), who estimate urban and suburban sprawl from 2006-2031 on a 1x1 km square basis for Great Britain from population growth projections at a district level and land cover data derived from the 25 m resolution raster Land Cover 2000 dataset (Fuller et al. 2002). Locations of urbanisation are restricted to be realistic, i.e. they exclude existing urbanization, water, wetland, coastal rock, submerged rock and montane areas, as well as all areas covered by statutory protection, National Parks, listed landscapes, parks, gardens and monuments. In this work, we adopt the scenario presented in the Eigenbrod et al. (2011) paper in which current housing densities are maintained throughout time. The resulting 1km resolution urbanisation raster is then remapped to a 2x2km spatial resolution, in order to match the spatial resolution of the biodiversity module of this study. As high biodiversity value land such as forests, heaths, moors tend to be prioritised for conservation, we assume that urbanisation can only occur on farmland, which is derived from the 2007 Land Cover Map (Morton et al. 2014) and remapped on a 2x2km spatial grid. This simplifying assumption can be justified by government data showing that on average 88% of land use change from undeveloped to developed land from 2013-2018 in England occurred on land classified as agriculture, undeveloped land or vacant land (MHCLG 2020 Table P350).

We compute land uses after urbanisation in 2031, subtracting the amount of land required from housing derived from the urban sprawl model from the farmland available in 2007 in each of the 2km cells in England. Note that, due to data availability, the urbanisation model uses the 2000 Land Cover Map but computes urban land changes from population projections in the interval 2006 to 2031; we thus compute loss of farmland from 2007 (the closest available data on land cover to the year 2006) to 2031. The increase in urban land at the expense of farmland is the determinant of the biodiversity changes computed with the biodiversity model.

Biodiversity model

To model the impact of housing expansion on biodiversity we use the biodiversity module of the NEVO modelling suite (Binner et al. 2019; Wright et al. 2019). The biodiversity model uses an ensemble of species distribution models produced by the UK Joint Nature Conservation Committee (JNCC; Trippier & Hutchinson 2018), which generates probabilistic maps of presence of a series of 100 species of conservation priority in the UK. This modelling framework is designed to support data from as wide a range of taxa as possible, and allows and improvement of representation beyond only highly recorded species such as birds, thus providing a better indication of biodiversity changes in comparison to analyses focused on a single taxonomic group. The 100 species considered are selected from mammals, birds, vascular plants, invertebrates, lichens and herptiles. Plants represent a significant proportion of the species selected as they can be closely associated with habitat types and are generally good indicators for biodiversity (Brunbjerg et al. 2018). The species selection criteria were: to capture a wide range of taxonomic diversity, to capture species that had reasonably strong habitat associations so that they could be used to robustly estimate changes in biodiversity resulting from land use changes, to be species of UK conservation priority (all species were selected from the UK's 2007 Biodiversity Action Plan list), and to be charismatic to the general public. The full list of species is presented in Table 1 in Wright et al. (2019).

The output of the biodiversity model in each cell is the summed probability of each of the 100 species occurring in that cell, given the land use in the cell, and therefore provides a proxy for the overall species richness occurring in that cell. The main species distribution model covariates are environmental variables such as climate, elevation and land use and the results are the output of the best performing among seven different species distribution models run in multiple iterations, as in Croft, Chauvenet, and Smith (2017). Data on species presence used as inputs into the species distribution models was retrieved from the UK NBN Atlas (NBN, 2020) and ranged in spatial resolution from 100m to 10km; for the records at a lower spatial resolution than the 2km grid resolution used in this study, the modelling framework randomly allocates in different iterations the location of the records to all of the intersecting 2km grid cells. While this repeat process improves the species distribution maps, particularly for under-represented species, it also increases model uncertainty. Therefore, if species were deemed to have sufficient high precision records (more than 5000 recorded at a 2 km or higher precision), low precision records were excluded from the dataset.

Most species distribution modelling methods require both species presence and absence data. 5000 pseudo-absence points were generated within a mask of cells where priority species from the same taxonomic group had been recorded but the target species had not, in order to reduce the bias of false-absence due to a lack of recorder effort (Mateo, Croat, Felicísimo, & Munoz, 2010).

For the environmental covariates, monthly minimum temperature, maximum temperature and precipitation for 1961-1990 were produced by the Met Office and downloaded from the CEDA archive (Met Office 2019); the data was remapped to match the 2km resolution of the other modules in this study using bilinear interpolation (ignoring topography). From the climate data, 19 climatic variables were generated using the biovars function in the R package dismo (Hijmans et al. 2017). Agricultural land use data was modelled following Fezzi & Bateman (2011) and Fezzi & Bateman (2015), whose spatially explicit econometric model estimates the allocation of farmland between a set of 8 different crops and 3 types of grassland driven by agricultural input prices, existing spatial constraints (e.g. environmentally sensitive areas, national parks etc.), climate and other environmental variables (e.g. soil types, land gradient etc.). The allocation of farmland between crops and grassland types constitutes the agricultural land uses inputted for the biodiversity model, together with six other main land use categories (coastal margins, freshwater, marine, woodland, urban, semi-natural grassland) derived from 2007 Land Cover

Map (Morton et al., 2014). More details on the data sources, data preparation and model estimation are available in Fezzi and Bateman (2011) and Fezzi and Bateman (2015). The correlation coefficients between all variable pairs were tested. Where correlation coefficient was greater than ±0.7, only one variable was retained.

The biodiversity model used has a number of limitations. Firstly, it models species richness, as opposed to the policy goal of halting declines in wildlife abundance by 2030. We cannot rule out significant changes in species abundance, even in cases in which the richness of species in a given area does not change. Secondly, the observed changes in species richness for a certain area resulting from urban development do not allow us to derive the risk of extinction of threatened species over the whole country, highlighting the careful consideration that needs to be put into the ecological structure of the green landscapes and the impacts on ecological connectivity arising from housing expansion. Thirdly, with the species richness measure in each cell representing the sum of the occurrence probabilities for all 100 species, the outcome measure does cannot distinguish between an outcome where a set of species are present with near certainty, and one where more species are potentially present but with much lower certainty. Lastly, as the 100 species chosen were partially chosen for their habitat associations, generalist species were excluded from the analysis and the model therefore does not capture changes in generalists from land use change. For these reasons, we use the results as a proxy to demonstrate how effective species mitigation and BNG policies will need to be to prevent wildlife loss associated with new housing construction, rather than a precise quantification of the biodiversity impacts. The main take-away message is that lower rates of land take associated with housing expansion will have smaller biodiversity impacts than high rates of expansion, and therefore are less likely to contradict overarching national biodiversity goals.

Embodied and operational emissions models

Baseline embodied emissions model

To estimate the embodied emissions model associated with producing new housing, we use the embodied emissions model developed in and described in detail in Drewniok et al. (2022). Drewniok et al. (2022) estimate the amount and type of materials used in the production of new dwellings by combining information about the proportion of different dwelling types from the English Housing Survey (EHS 2020b) with case study archetypes for each dwelling type identified from letting agency or developers' websites. For each case study, information about the layout is used to estimate the dimensions for substructure, structure, roof, partitions, cladding, walls ceiling finishes, windows and doors. The analysis excludes insulation and fixtures and fittings. For each building element, the most common technologies are estimated based on information from the English Housing Survey (EHS 2020b) and NHBC standards (nhbc-standards.co.uk), and the material intensities for the different technologies are modelled based on NHBC standards. For elements that use a mix of technologies throughout the building stock, the share of alternative technologies is estimated through discussions with industry partners. The material quantities for each dwelling typology are then normalised by gross internal floor area. Similar methods are used to determine the material quantities for the conversion of non-domestic to domestic buildings, except it is assumed that the foundations and upper floors are reused (i.e. unassociated with embodied emissions), and it is assumed that 70% of the remaining building structure is reused. The total volume of materials required includes a small wastage rate, consistent with current building practice.

To estimate the emissions embodied in all of these materials, Drewniok et al. (2022) use life cycle assessment methods consistent with British standards (BSI 2011). Their analyses include the emissions associated with the materials and construction process up to practical completion, which represents approximately 70% of the whole life embodied emissions for residential buildings (Gibbons & Orr 2020). Carbon coefficients for each material are taken from the Inventory of Carbon and Energy (ICE

2019), and for materials not listed in the inventory, values are taken from their Environmental Product Declarations. Transport emissions are estimated based on the average emissions of road freight.

In our analysis we are concerned with changes in housebuilding and retrofit rates on the cumulative emissions of the entire housing stock. Detailed results about the breakdown of emissions between various structural components and housing types can be found in Drewniok et al. (2022). The model produces estimates of embodied emissions of housing which are consistent with other results reported in the literature, including those calculated by alternative methodologies (e.g. Steele et al. 2015).

Baseline operational emissions model

To estimate the operational emissions of the housing stock we use the operational emissions model developed in Serrenho et al. (2019). They estimate the operational emissions of the existing stock by, firstly, identifying the Environmental Impact Rating (EIR) and floor area for all England's dwellings (with the year 2018 as a baseline) using information from the English Housing Survey (EHS 2020c). They then use the government's standard method for translating dwellings' EIR into annual emissions using the equation (DECC 2014):

$$O = \begin{cases} (A+45) * 10\left(\frac{40}{19} - \frac{EIR}{95}\right), if \ \frac{O}{A+45} \geq 28.3 \\ (A+45) * \left(\frac{100-EIR}{1.34}\right), if \ \frac{O}{A+45} < 28.3 \end{cases}$$

where O represents the annual emissions in kg CO_2 and A represents each dwelling's floor area in m^2.

Simulating future housing scenarios

The Drewniok et al. (2022) model allows the user to vary the annual net additions to the housing stock, allowing us to simulate various future scenarios by altering the input for the annual net additions. The types of housing being added to the stock each year is assumed to reflect the distribution across different housing types under the baseline year. The model allows the user to vary multiple inputs, such as the degree of material decarbonisation over time experienced by various building materials or the demolition rate, but for the sake of simplicity interpreting the results of the differences between our housing scenarios, we maintain nearly all inputs constant across all of our scenarios. The key inputs we vary across scenarios are the annual net additions and the proportion of unoccupied homes in the housing stock.

To simulate the future operational emissions of the housing stock, the Serrenho et al. (2019) model calculates the operational emissions for the existing stock (i.e. the stock already present during the baseline year of 2018) and new stock separately. For the existing stock, the model takes the operational emissions of the stock during the baseline year derived using the method in Section 3.2 and uses this as a baseline annual operational emissions value (this is estimated at 54.2kg CO_2/m^2 as in Serrenho et al. (2019)). The user then sets a time to decarbonisation and a degree of decarbonisation for the existing stock, and the model then linearly reduces the operational emissions of the existing stock each subsequent year in line with the chosen input parameters. This allows us to simply model the effect of different decarbonisation rates on the existing housing, with that decarbonisation rate reflecting the combined effect of decarbonisation processes such as retrofitting, electrification and decarbonisation of the national grid.

For the operational emissions model, the floor area of new housing added to the housing stock each year is a function of the extent of new housebuilding (as with the embodied emissions model, we vary this input across our various scenarios), which is determined by the simulated change in floor area and the demolition rate (as demolished buildings are rebuilt to new standards). We update the demolition rate used in Serrenho et al. (2019) by altering the parameters of the Weibull distribution used to determine the building failure rate so that the extent of demolished buildings aligns with the up-to-date estimates used in Drewniok et al. (2022), approximately 12,000 homes/year. We then alter the change in floor area over time in the model and align it with the net additions input into the embodied emissions model, accounting for demolitions.

Once the scenarios for the change in the floor area of newbuilds over time are parameterised, the model then estimates the operational emissions of newbuilds over time in a mechanistically identical way to the existing stock. The operational emissions of newbuilds during the baseline year (2018) are estimated from English Housing Survey data (estimated at 19.1kg CO_2/m^2). The user then sets a time to decarbonisation and a degree of decarbonisation for newbuilds (this reflects the improvement in the sustainability standards of newbuilds over time), and the operational emissions of newbuilds are assumed to decline linearly over time to reach the desired level of emissions in the target year. Operational emissions are estimated annually by multiplying the floor area of newbuilds by their projected emissions for that year. Newbuilds constructed are assumed to decarbonise over time at the same rate as the improvement in building standards, again reflecting the decarbonisation caused by retrofitting and grid decarbonisation over time.

We sum the embodied and operational emissions of the housing stock annually from 2020 to 2050 to estimate the cumulative emissions under the alternative pathways.

Scenarios for the future of the housing stock

To estimate the emissions associated with alternative strategies for meeting England's housing need, we run three policy-relevant scenarios for the future of the housing stock, aligned with a) business as usual government housing policy, b) highly ambitious supply-side greening and the most optimistic retrofitting rates in the government's decarbonisation strategy, and c) highly ambitious supply-side greening coupled with reduced new construction and retrofit rates that exceed those targeted by government decarbonisation strategy. For the sake of the clarity of comparison between scenarios, and because there is no obvious justification for altering them based on government strategy or current trends, we maintain several of the parameters that can be altered in our models constant between the three scenarios. These are the demolition rate, the proportion of the housing stock represented by different forms of housing, the rate of improvements in material efficiency of construction (i.e. through better building design and more efficient use of materials), and the rate of converting non-domestic to domestic buildings.

Across all scenarios we also assume material decarbonisation rates across all of the key materials included in the analysis consistent with the net zero pathways highlighted by relevant industry associations. These decarbonisation rates include all improvements in efficiency and adoption of zero carbon electricity in their production, but Drewniok et al. (2022) exclude the components of different industry associations' decarbonisaton pathways that rely on negative emissions technologies or hydrogen as both are unproven at scale. For example, they assume a 36% decarbonisation of a unit of concrete in line with the industry's net zero strategy (GCCA 2021).

Under scenario 1, we assume that business as usual policy delivers a halving of the operational emissions of the housing stock by 2050. We derive this assumption by extrapolating the greenhouse gas emissions of UK housing from 1990-2019 to 2050 using national emissions data disaggregated by sector (BEIS 2021, Table 1.2; Figure 26). We note this could be considered an optimistic baseline decarbonisation scenario, as this trend is driven by decarbonisation from 1990-2014 – from 2014-2019, the carbon emissions of the housing stock in reality rose.

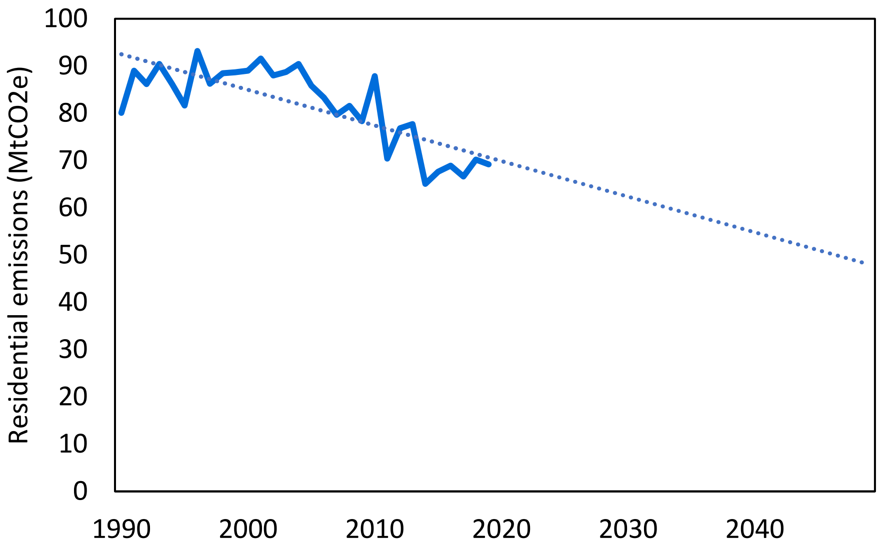

Figure 26. Decarbonisation rate of the UK housing stock from 1990-2050. Data from (BEIS 2021, Table 1.2)