A comparative analysis of ecological compensation programs:
The effect of program design on the social and ecological outcomes

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Abstract

An increasing interest is emerging in ecological compensation or biodiversity offsets as an instrument for slowing down the rate of biodiversity losses, with programs ongoing in more than 40 countries. Adhering to the polluter-pays principle, the instrument requires the intervening party to compensate for environmental degradation that has occurred as a result of development. However, its use is contested as the instrument may facilitate a ‘license to trash’ if more development permits are allowed under the assumption that the degradation can simply be compensated. This paper increases the understanding of whether existing programs for ecological compensation effectively addresses the loss of biodiversity and ecosystem services. A literature review is conducted with five case studies of ecological compensation programs representing middle- and high- income countries. We begin by describing the different approaches of designing compensation programs in Australia, England, Germany, South Africa and the U.S. Key issues in the program design are reviewed: policy goals, adherence to the mitigation hierarchy, weighting of losses and gains, and monitoring activities. We then evaluate the programs’ outcomes, in terms of the ecological and social benefits provided. From these case studies, we find three design aspects that may contribute towards improving the compensation programs’ outcomes: (1) integration of compensation programs with conservation landscape planning, (2) adequate commensurability of ecosystem functions and (3) an open access centralised reporting system. We also identify four safeguards to contribute towards protecting the ecological and social benefits: (1) allocate co-responsibilities of equivalence weighting and monitoring to an independent, external organisation, (2) consider local livelihoods at both the impact and proposed compensation site, (3) ensure access to recreation and (4) stakeholder participation from the general society and consultation of the directly affected community. We highlight that the use of this controversial instrument requires considerable regulation and capacity, with democratically decided performance criteria.
Table of Contents
1. Introduction .............................................................................................................................................. 4
2. Theoretical Framework .......................................................................................................................... 5
3. Review of Ecological Compensation Programs Design and Implementation ........................................ 7
   3.1 Description of Design and Policy Goals .......................................................................................... 7
   3.2 Adherence to the Mitigation Hierarchy ......................................................................................... 10
   3.3 Weighting Criteria for Measurement of Losses and Gains ............................................................. 12
   3.4 Monitoring Activities .................................................................................................................. 17
4. Review of Ecological Compensation Program Outcomes ....................................................................... 21
   4.1 Evaluation of Ecological Benefits ............................................................................................... 21
   4.2 Evaluation of Social Benefits ....................................................................................................... 22
5. Discussion ............................................................................................................................................... 26
6. Conclusion ............................................................................................................................................... 29
References .................................................................................................................................................. 30
1. Introduction

Biodiversity loss represents a critical environmental issue faced on a global scale. The Millennium Ecosystem Assessment reported that in the past 50 years, land use change from human activities has had significant influence on accelerating biodiversity losses. Species extinction rates were found to be increased by as much as 1,000 times background rates that were perceived natural by Earth’s history (MEA, 2005). With increasing urbanisation and infrastructure development, such biodiversity losses could continue to intensify dramatically (Quétier & Lavorel, 2011).

At the tenth meeting of the Conference of the Parties (COP) to the Convention on Biological Diversity (CBD), a Strategic Plan for Biodiversity was adopted to address this concern (CBD, 2010a). The Aichi Biodiversity Targets were adopted, with the Parties agreeing to incorporate them into their national biodiversity strategies and action plans. Under this Strategic Plan, Target 20 aims to mobilise financial resources towards reducing rates of biodiversity losses. A variety of resource mobilisation mechanisms has since been identified: ecological compensation, environmental fiscal reforms, markets for green products, and payment for ecosystem services.

The focus of this paper is upon ecological compensation, as a growing interest has been observed in such programs globally. More than 40 countries, including the 27 Member States of the EU, have regulations that call for some form of compensation for development impacts (OECD, 2013, p. 68). As a result of these programs, an estimated 187,000 hectares of land is placed under a protection scheme each year (Madsen et al., 2011). Throughout the literature, the terms ‘ecological compensation’ and ‘biodiversity offsets’ are often used synonymously. Conway et al. (2013) distinguishes ecological compensation as a general recompense for biodiversity loss, which can be done through various methods such as payments or conservation actions. Conversely, biodiversity offsets are understood as one method of compensation involving conservation actions with the goal of achieving no net loss or a net gain of biodiversity (BBOP, 2009a, p. 6). This paper favours the use of ecological compensation, as the term is regarded here as a broader concept that encompasses a range of compensation methods for the adverse environmental impacts of development.

The appropriateness of ecological compensation as a strategy for halting biodiversity loss has been widely contested at consecutive CBD-COP meetings (CBD, 2010b; CBD, 2012; CBD, 2014). Advocates of the instrument concur that ecological compensation, at the very least, attempts to offset losses that would occur anyway. Opponents raise the concern that compensation may undermine unique components of biodiversity as it facilitates habitat exchange, assuming equivalence in biological sites. A deeper understanding of key issues within ecological compensation is needed to contribute towards effective institutions, adapted to national and local contexts.

The aim of this paper is to increase the understanding of whether ecological compensation effectively addresses the loss of biodiversity and ecosystem services. This will be determined through three research objectives:

i) To assess the different approaches of designing ecological compensation programs
ii) To examine how ecological compensation programs adhere to the avoidance step of the mitigation hierarchy in its design and implementation
iii) To evaluate the outcomes of ecological compensation programs, in terms of the ecological and social benefits provided
With these questions in mind, five case studies of ecological compensation programs are examined in Australia, England, Germany, South Africa and U.S. These case studies were selected to reflect ecological compensation experiences from high and middle-income countries, as well as due to the availability of data. U.S. and Germany’s compensation programs are well established with over two decades of experience; Australia’s program is recently established with approximately a decade, whilst England and South Africa’s programs are newly emerging in their pilot stages. The key issues in compensation design that influences its social and ecological outcomes are then explored. Recommendations are provided to support ongoing improvements in the design of both existing and emerging ecological compensation programs.

2. Theoretical Framework
Ecological compensation has been described as ‘the substitution of ecological functions or qualities that are impaired by development’ (Cuperus et al., 1999). Similarly, biodiversity offset are defined as ‘measurable conservation outcomes resulting from actions designed to compensate for biodiversity impacts arising from project development. The goal is to achieve no net loss and preferably a net gain of biodiversity’ (BBOP, 2009a, p. 6). In either case, the instrument is intended to be used as a final option after all reasonable measures have been taken to adhere to the steps in the mitigation hierarchy. The hierarchy first prioritises the avoidance of negative impacts by identifying no-go areas and steering development to land with less biological value. Secondly, it refers to reduction of those impacts which cannot be avoided; and finally, to offset or compensate for the residual adverse impacts. However, certain interpretations of policy may cause less emphasis on the avoidance step as it involves making value judgements to determine what the threshold should be before resorting to the appropriate use of ecological compensation (Bull et al., 2012, p. 6). Regulators must weigh the interests of economic development, environmental conservation and benefits to society when contemplating the instrument’s use.

Advocates note that compensation is in line with the polluter-pays principle, which states that the polluter should bear the costs of pollution prevention and control measures towards encouraging rational use of scarce environmental resources (OECD, 1975, p. 12). The negative externalities of habitat degradation and biodiversity loss are internalised into the monetary costs of development projects, thereby holding the developers accountable for their residual impacts on land and water. Ecological compensation represents a potential instrument to balance meeting development goals whilst protecting biodiversity objectives by implementing conservation actions.

Opponents of the instrument highlight that permits may be more easily obtained by developers if all negative ecological effects are assumed to be readily compensated. Regulators may be partial to the ‘restoration myth’, which assumes the ability to mitigate any ecosystem damage by simply restoring degraded land or creating new habitats (Gibbons & Lindenmayer, 2007). Ecologists Hilderbrand et al. (2005) note that within restoration activities, complex ecosystems are broken down into simplified concepts and often recreated through minimalist guiding principles. Furthermore, studies have shown that restorations are not necessarily successful in replacing ecosystem structure or function due to weaknesses in the compensation program design and implementation (Turner et al., 2001; Kihslinger, 2008).

Hence, without strict guidance from regulatory and planning authorities, ecological compensation may result in a ‘license to trash’ (McKenney & Kiesecker, 2010, p. 173). This occurs when regulators focus on the compensation part at the expense of avoidance and
minimisation, leading to regulators approving developments that would have been rejected in the absence of a compensation program. Offsets may incentivise development by allowing irreplaceable ecosystems to be degraded, under the pretext that it can be recreated somewhere else. Hence, the debate on the use of ecological compensation demonstrates a controversial instrument that has potential to protect or accelerate biodiversity loss. The avoidance step of the mitigation hierarchy is thus crucial since it relates to no-go areas, the restoration myth and the license to trash.

The following issues in Figure 1 has been identified throughout the literature as key determinants in the design that influences the effectiveness of ecological compensation outcomes: policy goals, mitigation hierarchy, loss-gain measurements, and monitoring activities (McKenney & Kiesecker, 2010; Bull et al., 2012; Conway et al., 2013). These issues describe the overarching framework of the paper, as seen in Figure 1. We start by describing the policy goals and overall design of the programs. In the following sections, we analyse the adherence to the mitigation hierarchy, weighting of losses and gains, and the monitoring activities. Then we review the outcomes in terms of the social and ecological benefits of each program, followed by a discussion and conclusion. This paper represents a synthesis of data gathered from scientific papers, grey literature, government policy papers and industry publications.

Figure 1: Key issues influencing the design and outcomes of ecological compensation
3. Review of Ecological Compensation Programs Design and Implementation

3.1 Description of Design and Policy Goals

The U.S. and Germany have compensation programs on a national scale, whilst Australia and South Africa have state-wide programs. England is undergoing discussions for a national policy while trialling biodiversity offsets on a county basis. These programs have different institutional design schemes with varying degrees of involvement by market actors. Understanding policy goals provides an overview on how the programs approach compensation, as the goals set the tone on the biodiversity conservation.

**U.S.**

The U.S. Environmental Protection Agency’s mitigation policy goal applies to the federal level and is specific to preserving wetlands. This is observed in 40 CFR Part 230, which requires ‘No net loss of wetland acreage and function’ (EPA, 2008). The Clean Water Act (1977) states a preference for compensation to provide, at a minimum, a 1:1 loss to gain ratio of wetland acreage and functional value for protection in perpetuity. Developers are obligated to compensate for their wetland degradation activities, and can do so through the recommended mechanism of purchasing credits from mitigation banks.

To establish a mitigation bank, the initiator must first obtain certification by submitting a mitigation plan to the U.S. Army Corps of Engineers (Corps). This consists of the bank’s objectives, initial site status, methodology used to determine credits, ecological performance standards, monitoring requirements, long-term management plan and financial guarantees (EPA, 2008). With this, the bank can then begin to sell its credits to developers for creating, restoring, enhancing or protecting wetland functions. The price of credits is determined by the mitigation banks. Compensation by banks is established in advance of the authorised impacts, thereby accounting for a time lag (ELI, 2002, p. 8).

**Australia**

Australia has no policy for compensation at a federal level yet; however certain states have taken the initiative for their own compensation requirements. 12 programs are currently active in Australia, and the focus of the review is on the state of Victoria’s BushBroker program as this represents one of the larger established compensation markets in terms of hectares (Madsen et al., 2010, p. 50). In 2002, the state issued a policy requiring ‘a reversal, across the entire landscape, of the long term decline in the extent and quality of native vegetation, leading to a net gain’. The policy is specific to native vegetation, which includes various landscapes such as rainforest, wetland, grassland and forest. Any clearing of such requires compensation by the developer if on-site compensation is found unsuitable by regulators. The local council considers the role of the site in protecting water quality and its conservation status before determining the number of credits needed to be compensated.

BushBroker has a market-based approach that enables the trading of native vegetation credits. The government sets regional conservation goals for native vegetation and invites landowners to propose conservation actions, which contribute towards these goals (Dempsey & Robertson, 2012, p. 771). The government then selects the landowners that meet the defined standards, allowing the landowners to register their land for credits in exchange for site protection in perpetuity. A potential new income stream is created for landowners from managing their land. The local council facilitates the trading by matching landowners’ offsets and developers’ planning permits needs. This overcomes the issue of regulators having to manage several scattered, small areas of native vegetation (EFTEC, 2010, p. 23).
England

Within the EU, compensation is a legal requirement only for habitats protected by the EU Birds and Habitats Directives, and the Environment Liability Directive in the case of damage to a Natura 2000 site (European Commission, 2014). However, compensation is determined on an ad hoc basis without any specific criteria that determines a baseline for biodiversity. There exists a lack of guidelines, methods and metrics to ensure compensation is consistently implemented, hence the responsibility falls upon the Member States to interpret and uphold accordingly (Conway et al., 2013, p. i).

Outside of Natura 2000, England emphasises a ‘No net loss of priority habitat and an increase in the overall extent of priority habitats by at least 200,000 hectares’ by 2020 (DEFRA, 2011, p. 4). The offset scheme focuses upon protecting ‘priority habitats’, which have been identified by the government as those that are most threatened and require conservation action. Additionally, the National Planning Policy Framework (NPPF) imposes the responsibility upon local planning authorities to consider whether harm to biodiversity is ‘significant’ when evaluating development proposals, and seek compensation where appropriate (DCLG, 2012, p. 27). The local authorities are then to determine the amount of harm inflicted in terms of ‘biodiversity units’, subjecting the developer to secure an offset worth the same amount of units. Although the NPPF requires offsets to be conducted when development impacts are ‘significant’, lack of resources and trained ecologists in planning authorities have resulted in its limited application (Tucker et al., 2013, p. 227).

Offset projects are currently being piloted in six counties, with the outcomes determining how the instrument can be implemented more widespread across England. Each pilot area has an offsetting strategy, in which the local authorities have defined the type of habitats sought after and the locations that contribute towards habitat connectivity (DEFRA, 2012a, p. 3). An offset provider should either have ownership rights or a long-term agreement for the land as the offsets needs to last as long as the development project’s impacts. By measuring the amount of biodiversity units that the land encloses and the cost of maintaining those units, the offset providers then propose a price to the developers.

Germany

Within Germany, the Federal Nature Conservation Act introduced the Impact Mitigation Regulation (IMR) ‘Eingriffsregelung’ in 1976 to address compensation for biodiversity loss from development impacts. Regulations call for impacts on nature and landscape to be avoided, defined as ‘changes affecting the shape or use of areas, or changes in the groundwater level associated with the active soil layer, which may significantly impair the performance and functioning of the natural balance or landscape appearance’ (BMUB, 2010). This addresses ecosystem function and services in general, with an underlying theme of ‘no net loss’ (EFTEC, 2010, p. 60). Under these regulations, each municipality determines the measure of losses and the intervening party is responsible for ensuring adequate compensation is carried out. Compensation should be maintained as long as the impact persists, however this is usually decided by the municipality on a case by case basis. An average of 25-30 years for length of commitment by the developer was noted as compensation sites are only deemed ecologically suitable if they can be restored within that period (Albrecht, Schumacher, & Wende, 2014, p. 319).

The IMR requirements have resulted in the emergence of compensation pools (Flächenpools), which are linked to eco-accounts (Ökokontos), as an instrument for provision and bundling of usable sites for mitigation impacts. Compensation agencies, administered by the municipality
or the private companies, secure land and make it available for project developers to implement compensation. Plyusnin and Müller (2014, p. 70) report that 75% of project developers opt for the pools as their preferred form of compensation measure. The IMR applies to areas outside of Natura 2000 requirements, suggesting that the regulations in Germany are comprehensive as they go beyond the safeguarding measures implemented by the EU (Wende, Herberg, & Herzberg, 2005, p. 110).

**South Africa**

Environmental conservation is supported nationally through the National Environmental Management Act 107 of 1998 and the Biodiversity Act 10 of 2004. As offsets are an emerging practice in South Africa, a national policy is under discussion. The Western Cape Biodiversity Offset program is examined as it is the pioneering initiative in South Africa. In 2007, the local government released a draft policy requiring ‘Residual impacts on biodiversity and ecosystem services that are of moderate to high significance are compensated by developers in such a way that ecological integrity is maintained and development is sustainable’ (DEA&DP, 2007). Acknowledging that no net loss of biodiversity is ‘unlikely to be realistic in a developing country such as South Africa’, the policy aims to ensure that offset requirements are attached to certain amount of ‘acceptable loss’ of threatened vegetation types and ecosystem services as deemed by decision-makers (DEA&DP, 2007, p. 11).

For a development project to receive environmental authorisation, it is mandatory for an Environmental Impact Assessment (EIA) to be conducted by an independent environmental assessment practitioner. If harm is found to be moderate or high, the planning authorities will then propose a type of offset prior to the development project. A biodiversity specialist can also be appointed to advise the environmental assessment practitioner on the amount of losses and gains. The developers then have the responsibility to secure an offset in perpetuity through two methods: a stewardship agreement with a landowner or through donation of land to a conservation agency. The offset design process is integrated with regional planning as they are encouraged to be located within the ‘offset receiving areas’ that are identified by landscape planners (EFTEC, 2010, p. 51).
3.2 Adherence to the Mitigation Hierarchy

Adherence to the mitigation hierarchy is usually first defined within the program’s design in order to determine whether implementing compensation activities are appropriate. As per the precautionary principle\(^1\), compensation should be considered in regards to the mitigation hierarchy of avoidance, then impact minimisation and finally compensation. Appropriate interpretation of the first step of avoidance is detrimental towards ensuring the mitigation hierarchy’s implementation. Although all the reviewed programs do acknowledge the hierarchy, their policy design and actual implementation varies throughout. Table 1 examines the mitigation hierarchy in policy design by decision-makers and evidence of avoidance in its implementation by scholars. All the programs adhere to the hierarchy in principle; however its implementation was found to vary.

Table 1: The role of the mitigation hierarchy in the policy design and implementation of ecological compensation programs

<table>
<thead>
<tr>
<th>Mitigation Program</th>
<th>Policy Design of Mitigation Hierarchy</th>
<th>Implementation of Mitigation Hierarchy</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetland Banking, U.S.</td>
<td>Insufficient design is noted as although the legislation formally prioritises avoidance over impact minimisation and compensation, there is no guideline across districts for evaluating avoidance practices (ELI, 2008, p. 4). Evaluation guidance is needed in practical terms given the type and size of the project, and the type, function or quality of the wetland.</td>
<td>Limited implementation is observed as permittees can argue that there are no other alternatives by citing limitations to the proposed development, as there is substantial flexibility in the application of the alternatives (Clare et al., 2011, p. 167). Avoidance is overlooked as typically, less than 1% of permits applied for are denied whilst a majority of applications are granted under an expedited process (Murphy et al., 2009, p. 315). Hough and Robertson (2009, p. 30) highlight that the program has focused on improving the success rate of compensation but neglected the effectiveness of avoidance and minimisation.</td>
</tr>
<tr>
<td>BushBroker, Australia</td>
<td>Sufficient design is noted. The policy utilises a risk-based pathway to process applications based on the effect of the native vegetation’s clearing on the surrounding landscape, instead of applicants demonstrating that the</td>
<td>Implementation of the hierarchy is observed as Treewek (2009, p. 45) notes that Victoria’s strong policy for ‘net gain’ combined with the local government’s working compensation system has been effective in reducing applications for clearing of native vegetation. Avoidance is observed as the number of</td>
</tr>
</tbody>
</table>

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\(^1\) Defined as “In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation.” Rio Declaration, United Nations (1992)
clearing was avoided and minimised (DSE, 2012, p. 25). The removal site is assessed in relation to the landscape’s environmental significance.

**Habitat Hectares (HHa)** of native vegetation that was permitted to be cleared by the Victoria state department had reduced from 163 HHa in 2008 to 76 HHa in 2011 (DSE, 2012, p. 13).

**Biodiversity Offsetting Pilots, England**

Sufficient design is noted. The local planning authorities examine development decisions in line with planning guidance; the NPPF notes the mitigation hierarchy with the assessment of applications. Paragraph 118 states that planning permission should be refused for ‘development resulting in the loss or deterioration of irreplaceable habitats’ (DCLG, 2012, p. 28).

Limited implementation as biodiversity impacts are not examined appropriately or consistently due to a shortage of trained ecologists within local authorities (Tucker, et al., 2013, p. 227).

Avoidance tends to be overlooked, as Evans (2013) highlights that planning authorities are balancing the options of refusing permission on biodiversity grounds against the pressure for new housing and business development. There is a tendency for economic priorities to override the planning authorities’ responsibilities for biodiversity conservation.

**Compensation Pools, Germany**

Sufficient design is noted with the definition of a decision-making sequence in the Federal Nature Conservation Act. Clear residual effects on biodiversity are stated, such as if the project will cause a change in the form or use of land surface or groundwater level with the active soil layer (EFTEC, 2010, p. 62). If impairment of nature cannot be avoided or minimised, a clarification is needed on the extent taken.

Implementation of the hierarchy is observed as Tucker et al. (2013, p. 197) notes that within the evaluation of development permits, a strong emphasis is placed on the location of projects according to regional spatial plans. Sites with development restrictions or ‘no-go’ areas have been identified by the landscape planning system (Rajvanshi, 2008, p. 171).

Avoidance is observed, as German respondents noted that the consensus needed on the development project’s location by local planners and developers has limited the actual demand for compensation (Conway et al., 2013, p. 88).

**Western Cape Biodiversity Offsets, South Africa**

Sufficient design is noted. Decision-making sequences are provided for the district and local municipalities to determine the residual impacts on both biodiversity and ecosystem services (DEA&DP, 2007, p. 81). Habitat mapping advises where developments should

Implementation of the hierarchy is observed as regional planning reports highlight the use of comprehensive spatial biodiversity mapping and establish offset requirements for developers through identifying ‘no-go’ areas (Purves & Holmes, 2014). Todd (2012, p. 12) demonstrates that EIAs consider these critical biodiversity areas and actively avoid them in the site selections of
be avoided, emphasising a landscape ‘planning with nature’ approach before resorting to biodiversity offsets (DEA&DP, 2007, p. 37).

Avoidance is noted as 32% of applications from 2009-2012 were returned to developers by the independent commenting authority (Turner A., 2012, p. 13).

### 3.3 Weighting Criteria for Measurement of Losses and Gains

The ability of the instrument to compensate for degradation impacts depends on the measurements used to determine its losses at the impact site and gains at the compensation site. Quantifiable, conservation outcomes are inherent to measuring the program’s success on compensating losses. Policy-makers must decide if the program aims to compensate species, ecosystem function or ecosystem services and subsequently ensure this is reflected in the choice of metrics.

**U.S.**

The wetland credits and debits traded by banks are standard units that relate to a measure of quantifying wetland acreage or function (ELI, 2002, p. 23). The banks have flexibility in deciding how to measure losses and gains, where usually a combination of acreage-based measurement and functional assessment methodology is used. Depending on the complexity of the wetland, the number of acres needed to compensate is multiplied with an index of functional value. Various methods are available to quantify wetland function such as the Wetland Evaluation Technique, Habitat Evaluation Procedures and Hydrogeomorphic Approach. This plurality of methods may allow for experimentation and learning amongst mitigation sites.

One commonly used method is the Wetland Rapid Assessment Procedure (WRAP), where a number of criteria are measured across ten categories, such as hydrology, vegetation, soil and wildlife utilisation. These categories address the main factors that create, maintain and degrade wetlands (Fennessy, Jacobs, & Kentula, 2007, p. 546). This score is then considered against a reference benchmark of wetlands that have been least impacted by human activities. The assessment method allows for comparisons among compensation sites, relative to the reference condition. ‘Like-for-like’ exchanges are required amongst sites. The variables measured by WRAP represents a qualitative assessment of a wetland’s functional capacity, thereby determining whether the overarching policy goal is achieved (Brown & Benjamin Vivas, 2005, p. 299).

**Australia**

Government representatives assess the losses and gains to determine the site’s Habitat Hectares (Conway et al., 2013, p. xiv). The method has been developed for native vegetation and measures the condition of the site against its vegetation class. Each class has a characteristic structural variation of plant species that represent the benchmark of a ‘natural condition’, of which the site is measured against. After identifying the site’s vegetation class, its Habitat Score is calculated through the following pre-defined indicators as seen in Table 2.
Table 2: Ecological performance criteria to assess the Habitat Score (%). Modified from Morandeau & Vilaysack (2012)

<table>
<thead>
<tr>
<th>Indicators</th>
<th>Maximum value (equivalent to the class benchmark in %)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Site</strong></td>
<td></td>
</tr>
<tr>
<td>Number of large trees</td>
<td>10</td>
</tr>
<tr>
<td>Tree canopy cover</td>
<td>5</td>
</tr>
<tr>
<td>Number of understory life forms</td>
<td>25</td>
</tr>
<tr>
<td>Cover of weeds</td>
<td>15</td>
</tr>
<tr>
<td>Number of new recruits established through revegetation</td>
<td>10</td>
</tr>
<tr>
<td>Cover of organic litter</td>
<td>5</td>
</tr>
<tr>
<td>Abundance of logs</td>
<td>5</td>
</tr>
<tr>
<td><strong>Landscape</strong></td>
<td></td>
</tr>
<tr>
<td>Patch size</td>
<td>10</td>
</tr>
<tr>
<td>Proximity of remnant vegetation</td>
<td>10</td>
</tr>
<tr>
<td>Distance to core area</td>
<td>5</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td>100</td>
</tr>
</tbody>
</table>

The Habitat Hectares are then determined by a multiplication of the habitat score and proposed site area. The emphasis of the methodology on the compensation area size aids to measure the policy goal of achieving net gain in the quality and extent of native vegetation. ‘Like-for-like’ exchanges are required based on the habitat type and quality (EDO, 2012, p. 15). The method is appropriately aligned with state policy objectives as the metric is used consistently across Victoria.

**England**

To quantify the impacts on biodiversity, DEFRA utilises a ‘biodiversity units per hectare’ metric which compares the Habitat Quality against Habitat Distinctiveness (DEFRA, 2012b, p. 7). As seen in Table 3 below, Habitat Quality is assessed on a scale of poor, moderate or good condition. The criteria for classifying each scale is set out in the national Farm Environment Plan Manual (Natural England, 2010). Subsequently, Habitat Distinctiveness is distinguished from a low to high scale by identifying its habitat type. Habitats with high distinctiveness are determined by the classification of priority habitats, medium distinctiveness is indicated by a semi-natural state and low distinctiveness consists of land that may have been used for agricultural purposes. Hence, high distinctiveness and good habitat quality of the impact site would require a higher number of biodiversity units per hectare for compensation. As an emphasis is placed upon specifying habitat types, the policy goal of ‘no net loss of priority habitats’ is reflected.

Table 3: Biodiversity units per hectare as determined by the Habitat Condition and Habitat Distinctiveness. Modified from: DEFRA (2013)

<table>
<thead>
<tr>
<th>Biodiversity units per hectare</th>
<th>Habitat distinctiveness</th>
</tr>
</thead>
<tbody>
<tr>
<td>Habitat quality</td>
<td>Low (2)</td>
</tr>
<tr>
<td>Good (3)</td>
<td>6</td>
</tr>
<tr>
<td>Moderate (2)</td>
<td>4</td>
</tr>
<tr>
<td>Poor (1)</td>
<td>2</td>
</tr>
</tbody>
</table>
Critics note that the measurement of biodiversity units is ‘overly simplistic’ as it does not consider habitat connectivity or ecosystem function (EAC, 2013, p. 3). Furthermore, it assumes that all biodiversity values are commensurable. The importance of species diversity living within those habitats is also omitted. An ecologist notes that the grassland sites in the Somerset pilot may have low scores on the metric but they have high biological value as a feeding ground for species, which the habitat-based metric does not take into account (Evans, 2013). Unlike the Australian Habitat Hectares, the metric’s simplicity leaves room for subjective scoring. Local authorities may resort to making value judgements when differentiating between ‘poor’ and ‘moderate’ quality grassland. This is an issue in the event where local authorities face a lack of capacity in properly characterising ecological impacts. Tewes (2009, p. 66) reports that only an estimated 35% of local planning authorities employ an ecologist or biodiversity specialist to advise upon the need for an offset.

Germany

Biotope mapping has been established as a standard practice within Germany and its valuation procedures are widely used in the compensation pools as an evaluation metric for losses and gains. A biotope is a habitat of organisms living in the same area, characterised by similar ecological conditions. Biotopes are further categorised into biotopes types based on its function, and then assigned values according to their ecological priorities (Quétier & Lavorel, 2011, p. 2994). Table 4 presents an extract of biotope type values from Saxon-Anhalt, in which the values are scored on a scale of 0 to 30 per square meter. Each state develops their own biotope valuation list, leading to site-specific allocated values that are not comparable across states.

Table 4: Example of a biotope list from the state of Saxon-Anhalt. Modified from: Morandeau & Vilaysack (2012, p. 60).

<table>
<thead>
<tr>
<th>Biotope examples</th>
<th>Biotope examples</th>
<th>Biotope impact type</th>
<th>Biotope value before impact (per square meter)</th>
<th>Biotope value after impact (per square meter)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>Beech forest</td>
<td>Forest</td>
<td>27</td>
<td>20</td>
</tr>
<tr>
<td>Meadow</td>
<td>Intensive grassland</td>
<td>Meadow</td>
<td>10</td>
<td>9</td>
</tr>
<tr>
<td>Inland waters, streams</td>
<td>River</td>
<td>Inland waters, standing bodies of water</td>
<td>30</td>
<td>23</td>
</tr>
<tr>
<td>Inland waters, standing bodies of water</td>
<td>Marsh</td>
<td>Inland waters, standing bodies of water</td>
<td>18</td>
<td>15</td>
</tr>
</tbody>
</table>

With the biotope type identified according to the state list, the difference in values is measured before and after at the impact and compensation site respectively. The biotope types are then multiplied by the area to establish a ‘quality x quantity’ indicator, determining the overall value lost and gained. Figure 2 demonstrates a calculation example where the impact loss of 10 hectares of grassland resulted in a value loss of 250, which was subsequently balanced with 12.5 hectares of shrubbery. Within a state list, commensurability is assumed between biotopes of the same habitat type. This is observed in the following example of compensating grassland with shrubbery, as ‘like-for-similar’ exchanges are

---

2 This example displays just one biotope type per biotope category. For instance, several species of beech are listed in the state list.
conducted between biotopes with similar geographic location and ecosystem function (Wende et al., 2005).

Table 5: Biotope Valuation Procedure Loss-Gain calculation in the state of Thuringia. Modified from: Darbi & Tausch (2010)

<table>
<thead>
<tr>
<th>Impact</th>
<th>Biotope type before impact</th>
<th>Biotope value before impact</th>
<th>Biotope type after impact</th>
<th>Biotope value after impact</th>
<th>Difference between biotope values</th>
<th>New extensive grassland</th>
<th>100% sealed road</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>(vb_1)</td>
<td></td>
<td>(va_1)</td>
<td>(vd_1 = vb_1 - va_1)</td>
<td>25</td>
<td></td>
</tr>
<tr>
<td>Area size</td>
<td>(a_1)</td>
<td>(v_1 = vd_1 + a_1)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Resulting value loss</td>
<td>10 ha</td>
<td>250</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Compensation</th>
<th>Biotope type before compensation</th>
<th>Biotope value before compensation</th>
<th>Biotope type after compensation</th>
<th>Biotope value after compensation</th>
<th>Difference between biotope values</th>
<th>Fallow field</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(vb_2)</td>
<td>(va_2)</td>
<td>(vd_2 = vb_2 - va_2)</td>
<td></td>
<td></td>
<td>20</td>
</tr>
<tr>
<td>Area size</td>
<td>(a_2)</td>
<td>(v_2 = vd_2 + a_2)</td>
<td></td>
<td></td>
<td></td>
<td>12.5 ha</td>
</tr>
<tr>
<td>Resulting value gain</td>
<td>250</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Balance</th>
<th>Value loss impact = Value gain offset</th>
</tr>
</thead>
</table>

The selection of biotope types to be compensated is identified on a case by case basis, determined by the compensation agencies and experts (Darbi & Tausch, 2010, p. 2). However, the metric does not consider abiotic functions, as soil or climate conditions within the ecosystems are not measured. This neglects vital components that influence the habitat’s natural balance (Tucker, et al., 2013, p. 199). Despite its criticisms, biotope valuation is relatively comprehensive, simple to apply and widely accepted by authorities in Germany. Ideally, compensation activities would be within the same biotope type. The transparency of the method creates a standardised basis of assessment for monitoring activities. Additionally, the flexibility of site-specific biotope type selection addresses the broad policy goal of compensating for ‘interventions that affect the shape or use of areas or changes in groundwater level’.

South Africa

An offset ratio is used to assess the baseline losses and determine the number of hectares needed to compensate biodiversity. This is influenced by the threatened ecosystem’s status, considering the degree of losses associated with the affected biodiversity. If the ecosystem impacted is found to be ‘critically endangered’, the development should be refused and the area determined as a ‘no-go’. Certain exceptional cases are allowed if the proposed development has substantial benefits to society and there are no alternative ways for those benefits to be obtained.

If the status were ‘endangered’ or ‘vulnerable’, reasonable compensation would need to be investigated. The Western Cape program applies multipliers in the form of offset ratios, as seen in Table 5. The area is then adjusted by a range of context-specific considerations;

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3 The listing of threatened ecosystems is defined by the National Environmental Management Biodiversity Act (2004), [http://faolex.fao.org/docs/pdf/saf45083.pdf](http://faolex.fao.org/docs/pdf/saf45083.pdf)
influenced by criteria such as the habitat condition, threatened species, special habitats, ecological functions and socio-economic value of ecosystem services (EFTEC, 2010, p. 39). Revision of the ratio is advised upon by biodiversity specialists in the regulatory body.

Table 5: Offset ratio as determined by the conservation status of the affected ecosystem. Modified from DEA&DP (2007, p. 28)

<table>
<thead>
<tr>
<th>Impact on Biodiversity</th>
<th>Ecosystem Status</th>
<th>Basic Offset Ratio (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Irreversible and irreplaceable loss of ecosystem or species, with very high impact on international and national biodiversity</td>
<td>Critically endangered</td>
<td>30:1</td>
</tr>
<tr>
<td>Irreversible impact leading to substantial change in ecosystem or species, with high impact on national or provincial biodiversity</td>
<td>Endangered</td>
<td>20:1</td>
</tr>
<tr>
<td>Irreversible impact leading to change in ecosystem status, with low impacts on provincial but medium impacts on local biodiversity</td>
<td>Vulnerable</td>
<td>10:1</td>
</tr>
<tr>
<td>Negligible or low impact on local biodiversity or valued ecosystem services</td>
<td>Least threatened</td>
<td>No offset required</td>
</tr>
</tbody>
</table>

No offset is required for impacts on least threatened ecosystems as the policy defines a level of acceptable biodiversity loss due to South Africa’s developing status (Brownlie & Botha, 2009, p. 228). With the limited amount of resources, the regulators choose to focus their efforts on prioritising ‘irreversible impacts’. This addresses the constraints of land availability in the balance between achieving development and nature-conservation goals (Quétier & Lavorel, 2011, p. 2997). As the offset ratios are explicitly related to the significance of biodiversity impacts, measurement of compensation activities are in line with state policy.
3.4 Monitoring Activities

The successful implementation of ecological compensation is dependent on effective institutional arrangements and clear responsibilities in monitoring and enforcement (Conway et al., 2013, p. 103). Transparency in monitoring activities is needed to ensure the design requirements in Figure 1 are being adhered to and implemented appropriately. These monitoring activities should track the outcomes of ecological compensation, which provides the basis for evaluation of social and ecological benefits. Table 6 describes the time span and frequency of monitoring activities.

Table 6: Frequency and role of monitoring activities of ecological compensation programs

<table>
<thead>
<tr>
<th>Monitoring Activities</th>
<th>Wetland Banking, U.S.</th>
<th>BushBroker, Australia</th>
<th>Biodiversity Offsetting Pilots, England</th>
<th>Compensation Pools, Germany</th>
<th>Western Cape Biodiversity Offsets, South Africa</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Annual reporting for a minimum of 5 years, by the mitigation bank to the district engineer (EPA, 2008, p. 19607)</td>
<td>Annual reporting for a minimum of 10 years, by the landowner to the local council compliance officer (EDO, 2012, p. 25)</td>
<td>Ad hoc basis, roles under discussion (DEFRA, 2013, p. 29)</td>
<td>Ad hoc basis, by the compensation pools to the municipal authorities (Plyusnin &amp; Müller, 2014, p. 79)</td>
<td>Ad hoc basis, by appointed party to the provincial authority (DEA&amp;DP, 2007, p. 89)</td>
</tr>
</tbody>
</table>

**United States**

National policy states that the monitoring period for most compensation sites is a minimum of five years, although the period should be extended sufficiently if needed to show that the project has met its targeted ecological performance standards (EPA, 2008, p. 19645). The mitigation banks are responsible for monitoring site performance against the requirements outlined in their mitigation plan. These monitoring reports should then be submitted to the Corps’s district engineer annually (EPA, 2008, p. 19607). Where mitigation banks are found by the district engineer to not be meeting performance standards, measures can be implemented such as decreasing available credits, suspending credit sales or requiring alternative mitigation (EPA, 2008, p. 19610).

However, weak government capacity has been identified as a barrier for effective monitoring as the allocated funds for contract enforcement by the federal government was found to be insufficient (GAO, 2005). Brown and Veneman (2001) surveyed 114 wetland projects in Massachusetts and found that 54% were not in compliance with state regulations. Penalties for non-compliance with banking regulations are rarely enforced by the federal government, which leaves room for shirking.
**Australia**

Annual monitoring reports are required from the landowner in addition to government representatives inspecting the site bi-annually. The Landowner Agreement is a minimum ten-year management contract between the compensation provider and the local council, which states how the native vegetation will be protected and maintained in perpetuity (DSE, 2012). Responsibility for habitat management and a majority of the monitoring activities belong to the landowner, with thorough reporting guidelines provided on the use of photo monitoring by the state Department of Environment and Primary Industries. In the event that compliance by the landowner is found to be inadequate, the local council withholds the annual scheduled payments until the necessary works are completed. This indicates an incentive for landowners to fulfil their management requirements.

In practice, however, the EDO (2012, p. 22) notes that there has been limited or no monitoring of compliance in most local councils and limited enforcement action for breaches in contract. A lack of resources was noted, as certain councils may have just one enforcement officer who is entasked with managing compliance across the entire landscape planning scheme.

**England**

Monitoring is conducted on an ad hoc basis, as it is currently not a legal requirement of the Environmental Assessment Regulations. This leads to a lack of resources for planning authorities to enforce them (Treweek J., 2009, p. 73). Without being able to recoup the costs of the process, an administrative burden is placed on the local authorities to audit and validate the applications. This has resulted in a slow adoption of offsets despite it being clearly stated in the planning policy framework. A dialogue is currently in progress to determine if individual planning authorities, a national body or accredited assessors should undertake the assessments of monitoring activities (DEFRA, 2013, p. 29).

**Germany**

Insufficient monitoring was noted in the past, due to ambiguous regulations and a lack of clear obligations (Plyusnin & Müller, 2014, p. 24). Monitoring and reporting requirements were then strengthened in 2010, under Article 17.7 of the Federal Nature Conservation Act. The planning authorities are responsible for ensuring that clear goals are established and then verifying the subsequent outcomes with the compensation pools, which in turn are responsible for the long-term maintenance and monitoring of the sites. Site monitoring is conducted on an ad hoc basis according to the project schedule, with compensation pools reporting to the municipal authorities (Plyusnin & Müller, 2014, p. 79). A state report is then submitted to the Federal State every six years. There has yet to be empirical evidence of the effects of these legislation changes, although stakeholders anticipate that the monitoring situation will improve (Tucker, et al., 2013, p. 200)

**South Africa**

The developer is responsible for monitoring the offset site and submitting reports at required intervals to the provincial authority, but is able to do this through an appointed party. Biodiversity specialists or consultants can be appointed to audit the monitoring report and ensure compliance according to the respective offset management plan (DEA&DP, 2007). Heavy reliance exists upon the professional judgement of consultants and conservation agencies. In the event of a large-scale project, an external body consisting of independent
technical experts, community representatives and the provincial conservation agency can be established to monitor the offset site. In practice, monitoring is seldom conducted due to the authorities’ lack of capacity to interpret and evaluate results (Brownlie & Wynberg, 2001, p. 34; Turner, 2012, p. 15).

Overall, diversity in monitoring responsibilities is observed throughout the compensation programs. The responsibilities are born by the mitigation banks and compensation pools for U.S. and Germany, whereas Australia and England delegate it to the landowner in addition to monitoring by the regulator. In South Africa, the developer is mainly responsible for ensuring how the site is monitored and managed in perpetuity. This diversity could be attributed to the financial constraints and institutional resource capacity dedicated to each program. U.S. and Germany are nation-wide programs whilst England, Australia and South Africa are regional programs.
Table 7: Summary of key issues in the design and implementation of ecological compensation programs, corresponding to research objectives 1 and 2

<table>
<thead>
<tr>
<th>Research Objective (1)</th>
<th>Policy Goals</th>
<th>Wetland Banking, U.S.</th>
<th>Biodiversity Offsetting Pilots, England</th>
<th>Compensation Pools, Germany</th>
<th>Western Cape Biodiversity Offsets, South Africa</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Time frame of protection</strong></td>
<td>Perpetuity</td>
<td>Perpetuity</td>
<td>As long as development impact</td>
<td>Average 25 years</td>
<td>Perpetuity</td>
</tr>
<tr>
<td><strong>Measurement of losses and gains</strong></td>
<td>Wetland Functional score x Area</td>
<td>Habitat Hectares</td>
<td>Distinctiveness of habitat type x Habitat condition x Area</td>
<td>Biotope value x Area</td>
<td>Offset ratio x Area</td>
</tr>
<tr>
<td><strong>Monitoring Activities</strong></td>
<td>Annual reporting for a minimum of 5 years, by the mitigation bank to the district engineer (EPA, 2008)</td>
<td>Annual reporting for a minimum of 10 years, by the landowner to the local council compliance officer (EDO, 2012)</td>
<td>Ad hoc basis, roles under discussion (DEFRA, 2013)</td>
<td>Ad hoc basis, by the compensation pools to the municipal authorities (Plyusnin &amp; Müller, 2014)</td>
<td>Ad hoc basis, by appointed party to the provincial authority (DEA&amp;DP, 2007)</td>
</tr>
</tbody>
</table>

| Research Objective (2) | Mitigation Hierarchy Design & Implementation<sup>6</sup> | Insufficient design Limited implementation Avoidance overlooked | Sufficient design Implementation noted Avoidance noted | Sufficient design Limited implementation Avoidance noted | Sufficient design Implementation noted Avoidance noted |

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<sup>4</sup> such as changes affecting the shape or use of areas or changes in the groundwater level associated with the active soil layer, which may significantly impair the performance and functioning of the natural balance or landscape appearance.

<sup>5</sup> By developers in such a way that ecological integrity is maintained and development is sustainable.

<sup>6</sup> Justification of terms in Table 1.
4. Review of Ecological Compensation Program Outcomes

4.1 Evaluation of Ecological Benefits

Monitoring is a key element in evaluating whether ecological compensation programs fulfil management requirements, as well as whether their objectives are achieved (Quintero & Mathur, 2011). The ecological benefits are evaluated with available studies, based on the monitoring data or external scientific analyses, to determine whether the program’s implementation meets its ecological objectives and policy goals.

In the U.S., the program’s success in achieving policy goals of ‘No net loss of wetland acreage and function’ has been questioned by studies such as Turner, Redmond, & Zedler (2001) and Kihslinger (2008). A net loss of ecosystem services was reported to be occurring on ground as numerous banks face difficulty in replacing wetland function. Based on a primary data analysis, Ambrose and Lee (2004) found that 46% of the 250 sites surveyed in California failed to replace key wetland ecosystem services that had been determined by the state. Another primary data study of 60 sites in the New England district determined that only 17% were considered adequate functional replacements in comparison to the original impacted sites (Minkin & Ladd, 2003). Nevertheless, functional replacement has been slowly improving. Recent studies by Hill et al. (2013) reported that 74% of wetland mitigation projects in North Carolina had fulfilled the ecological criteria specified in their respective mitigation plan. To coordinate data on a national level, a web-based system was established: the ‘Regulatory In lieu fee and Bank Information Tracking System’ (RIBITS). The system allows public access to information related to compensatory mitigation banks, however its use is limited as the data is not yet extensive for all projects and updated periodically (Kett, 2010).

In Germany, Tischew et al. (2010, p. 472) studied compensation pools throughout different states and found that 26 of 57 pools had poorly defined goals, unrecognisable implementation or lacked an actual site. Primary data was obtained from previous research projects for the analysis. From the 31 remaining pools, an analysis of the 326 clearly defined compensation goals revealed that only 33% of goals were fully achieved. The study attributed these poor results to a deficiency in the planning of objectives, program implementation or follow-up management of the sites. Hence, it is unlikely that the overall policy goal of ‘no net loss’ is fulfilled. Additionally, Tucker et al. (2013, p. 199) highlighted a current monitoring gap in measuring the cumulative compensation activities at a regional, state and national level. Initiating a centralised information system for planning and monitoring requirements would contribute towards to integrating the outcomes.

A gap in monitoring data was then identified in Australia, England and South Africa. Common reasons for inadequate monitoring include a lack of resources, capacity or time on the part of regulators. In Australia, an environmental consultancy reported that there is very little data collected by the local authorities on the extent of compliance monitoring and enforcement due to a lack of coordinated systems (EDO, 2012, p. 22). A web-based information system, the Native Vegetation Tracking System (NVT), is in use by the Department of Sustainability and Environment to track vegetation removal but its data is incomplete and unavailable to the public. In England, several studies note the poor monitoring of compensation programs by the local planning authorities due to constrained financial resources, administrative strain and ambiguity of assessment methods (Treweek, 1999; Latimer & Hill, 2007, p. 156; Hill, 2008).
Consequently, the South African regional conservation agency conveys that provincial authorities do not have the capacity needed for monitoring, as activities are ‘not always what it should be’ (SANBI, 2007, p. 58; Turner, 2012, p. 15). 80% of biodiversity specialists who provided input were not registered with the regulatory body for natural sciences, despite it being legally required (Brownlie, Manuel, & De Villiers, 2006). Registration is mandatory under the Natural Scientific Professions Act 27 of 2003, which aims to overcome the inconsistent standards of education and training of natural scientists. This has led to concerns in quality of the sector, as SANBI (2007, p. 67) highlights that developers prioritise costs and often contract the consultant with the lowest price or those with a favourable decision. Without reliable monitoring data, there is a lack of evidence that any of these programs have achieved their ecological objectives.

4.2 Evaluation of Social Benefits

The indicators used in the compensation programs’ loss-gain measures focus on determining a site’s ecological value and how that can be exchanged. However, the loss-gain measures generally demonstrate a lack of consideration for the social value that these sites provide. The third objective of the CBD concerns access and benefit sharing which might be a major issue for ecological compensation, due to the spatial distribution of the impact and compensation site. If the compensation site is far away from the impact site, this may affect local communities’ livelihoods and recreational opportunities that call for social safeguards (Iuarte-Lima et al., 2014). The COP12 meeting had also noted the “possible risks and benefits of country-specific innovative financial mechanisms and safeguards” (CBD, 2014). When dealing with safeguards, different countries would require emphasis on certain issues according to their own context and local communities. To reduce risks of social harm, this paper identifies three main safeguards from the examined compensation programs’ policies: Local livelihoods, Access to recreation and Stakeholder participation.

4.2.1 Local Livelihoods

In a submission to the CBD, the Forest People’s Programme considers that ‘offsets may pose serious risks of social harm and rights violations unless rigorous safeguards and due diligence are guaranteed’ (FPP, 2011, p. 4). Concerns are raised that biodiversity offsets may apply exclusionary conservation approaches at the compensation site, which compromise people’s livelihoods. Ecological compensation programs are highly debated as social risks are experienced at both the development impact and proposed offset sites. Forest communities are dependent on collective and/or common property resources for their livelihoods at a local spatial scale; hence the removal of a site and its compensation somewhere else may be a disadvantage (BenDor et al., 2008, p. 342). If the offset site is located several kilometres away, its accessibility might be severely limited for forest communities.

Within the examined programs, this risk is addressed in South Africa’s policy as the livelihoods of certain communities are reliant on natural resources. Establishing compensation programs may affect the community’s access to resources at both the impact and compensation site, thereby requiring different kinds of social safeguards. One observed safeguard in the policy recognises the rights of local communities to benefit from the impact site’s natural resources. This examines the use and non-use values to the local communities and whether they would be left more vulnerable as a consequence of the development (DEA&DP, 2007, p. 55). A social specialist is recommended to determine if compensation is

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7 Although legislation may not refer to it as such.
sufficient to the stakeholders at the impact site, and whether the proposed compensation site would be ‘accessible and acceptable’ to the affected communities. Subsequently, another safeguard that considers the rights of the other community at the proposed compensation site, which may have potential conflicts with the first community. The approval of an offset plan requires an evaluation of negative effects on the other community at the proposed compensation site and how this is to be avoided (DEA&DP, 2007, p. 64). The distance between sites should be minimised to ensure that the same community lives at both sites to avoid competition for resources. Compensation programs must consider the distribution of social equity: which stakeholders would benefit from, and which would bear the cost, at both the impact and proposed site.

Alternatively, ecological compensation programs are also noted to have positive effects on livelihood at the compensation site. According to BBOP (2009b), a case study of a mining offset in South Africa contributes towards poverty alleviation a in low-income rural community by creating a wildlife reserve and eco-tourism lodge in the offset area. This provides employment through habitat management of the reserve and tourism opportunities. Subsequently, ownership of the lodge will be transferred to the community. In Germany and Australia, interviews with private landowners have noted the economic attractiveness of entering a long-term maintenance contract with compensation agencies, as it ensured a stable source of livelihood for local farmers (Tucker et al., 2013, p. 193; O’Connor NRM, 2009, p. 15).

4.2.2 Access to Recreation

The benefits of ecosystem services are appreciated on a local spatial and functional scale. Substantive safeguards are needed in the policy design to ensure the local people have reasonable access for enjoyment of habitats and wildlife. As noted in the safeguards for livelihoods, recreational benefits would also be optimised if ecological compensation is implemented close to the impact site. Safeguards for the recreation value of nature are noted in the German legislation as it calls for suitable compensation areas to be made publicly accessible, particularly in areas close to human settlements (BMUB, 2010, p. 8). The policy also requires compensation measures to be located in spatial context with the impact, which is carried out through local planners who direct the location of compensation sites within regional spatial plans (Küpfer, 2008, p. 2). Although there are no strict regulations that determine the distance between impacts and compensation, REMEDE (2008, p. 27) highlights that the sites are generally located within municipality boundaries. In England’s pilot areas, the local planning authority direct offset locations through an offsetting strategy that connects priority habitats. This enhances the recreational value through establishing parks or urban green spaces as part of the country’s ecological network (Lawton, et al., 2010, p. 26).

Concern for accessibility may be lacking in the U.S. program, as BenDor et al. (2008, p. 349) notes that the decision-making processes of the national regulatory body does not consider the compensation location in relation to human settlement. Mitigation banks are often located in less densely populated areas where land is less expensive, thereby decreasing recreation access of the urban population (King & Herbert, 1997). A ‘migration of wetlands across the urban-rural gradient’ is realised, with the variations in human population density suggesting that different populations are benefiting from the local ecosystem services. Ruhl and Salzman (2006, p. 13) suggest that banking incentives could be restructured, by adjusting credit allotments and offering a premium to banks that factor proximity to impact site.
4.2.3 Stakeholder Participation

Involvement of stakeholders has a strong influence on ensuring the success of compensation, from its initial conception to the long-term management (BBOP, 2009a, p. 22). Procedural safeguards are needed to address the local community’s rights to participate in decision-making. By cooperating with key stakeholders, the aim is to minimise negative impacts and achieve broad acceptance of ecological compensation by the local community. The U.S. regulations require a description of how stakeholder involvement is addressed in the mitigation bank’s long-term management strategies (EPA, 2008, p. 19681). In South Africa, procedural safeguards are highlighted as public participation has a significant role in the decision-making process. Two groups of stakeholders are identified: the directly affected parties and the general society. As part of the EIA, project developers are required by law to provide an opportunity for stakeholder engagement. Both the directly affected parties and the general society have the right to participate in the decision of which compensation type would best compensate their loss (DEA&DP, 2007, p. 46).

Germany has also recognised this as a key issue, the lack of landowner participation has been observed to hinder implementation. This proves to be one of the biggest challenges in establishing a compensation pool, as there is high demand on land for building purposes and landowners usually achieve lower prices instead when their sites are provided for compensation measures. The landowners are then less willing to provide sites for compensation measures (Wende et al., 2005, p. 110; Plyusnin & Müller, 2014, p. 24). The national regulator body has since emphasised safeguards by introducing certain quality standards to assess the formation of a pool, in which ‘acceptance by existing land users’ is an evaluation criterion (EFTEC, 2010, p. 84).

Timeliness of stakeholder participation, especially for those potentially affected by the development, should be initiated in the early decision-making stages. This allocates time for incorporating stakeholders input to the process, including the development of effective measures to ensure that the scheme respects the rights of people at the impact site and the proposed compensation site. In England, the government set out proposals for biodiversity offsetting and hosts consultations for public feedback during the decision-making process of its national offsetting policy (DEFRA, 2013). In Australia, stakeholder reviews on the clearing of native vegetation are incorporated into a consultation paper every two years, providing input to the government’s ongoing policy amendments (EDO, 2012).
Table 8: Summary of ecological compensation programs outcomes in terms of ecological and social benefits, corresponding to research objective 3. N/A means the benefit was not applicable to the particular program.

<table>
<thead>
<tr>
<th>Ecological Benefits</th>
<th>Wetland Banking, U.S.</th>
<th>BushBroker, Australia</th>
<th>Biodiversity Offsetting Pilots, England</th>
<th>Compensation Pools, Germany</th>
<th>Western Cape Biodiversity Offsets, South Africa</th>
</tr>
</thead>
<tbody>
<tr>
<td>74% of wetland mitigation projects in North Carolina had achieved their ecological goals (Hill et al., 2013)</td>
<td>Lack of a coordinated collection system for monitoring data</td>
<td>Lack of monitoring data due to constrained financial resources, administrative strain and ambiguity of assessment methods</td>
<td>45% of compensation pools had poorly defined goals. From the remaining pools with clearly defined goals, only 33% of the goals were achieved (Tischew et al., 2010)</td>
<td>Lack of monitoring data due to limited capacity of provincial authorities</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Social Benefits</th>
<th>Local Livelihoods</th>
<th>Access to Recreation</th>
<th>Stakeholder Participation</th>
</tr>
</thead>
<tbody>
<tr>
<td>N/A</td>
<td>Source of livelihood for private landowners</td>
<td>N/A</td>
<td>Description required of stakeholder involvement in the mitigation bank’s long-term management strategies</td>
</tr>
<tr>
<td>Lacking accessibility as wetland are relocated in less densely populated areas where land is less expensive</td>
<td>N/A</td>
<td>Offset locations are designated in a local offsetting strategy that connects priority habitats, establishing parks and urban green spaces</td>
<td>Stakeholder reviews are compiled every two years as input to the government’s ongoing policy amendments</td>
</tr>
<tr>
<td>N/A</td>
<td>Compensation areas to be made publicly accessible and close to human settlements</td>
<td>N/A</td>
<td>Consultations hosted by the government for public feedback during the decision-making process of a national offsetting policy</td>
</tr>
<tr>
<td>1. The rights of local communities to benefit from the impact site’s natural resources 2. Avoiding negative impacts on the rights of the other community at the compensation site</td>
<td>Lack of landowner participation due to high demand on land for building purposes</td>
<td>N/A</td>
<td>Lack of landowner participation due to high demand on land for building purposes</td>
</tr>
<tr>
<td>Directly affected parties and the general society have the right to participate in the process of determining an offset</td>
<td>N/A</td>
<td>N/A</td>
<td>N/A</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Stakeholder Participation</th>
<th>Wetland Banking, U.S.</th>
<th>BushBroker, Australia</th>
<th>Biodiversity Offsetting Pilots, England</th>
<th>Compensation Pools, Germany</th>
<th>Western Cape Biodiversity Offsets, South Africa</th>
</tr>
</thead>
<tbody>
<tr>
<td>Description required of stakeholder involvement in the mitigation bank’s long-term management strategies</td>
<td>Lack of a coordinated collection system for monitoring data</td>
<td>Lack of monitoring data due to constrained financial resources, administrative strain and ambiguity of assessment methods</td>
<td>45% of compensation pools had poorly defined goals. From the remaining pools with clearly defined goals, only 33% of the goals were achieved (Tischew et al., 2010)</td>
<td>Lack of monitoring data due to limited capacity of provincial authorities</td>
<td></td>
</tr>
</tbody>
</table>
5. Discussion

In regards to the first and second research objectives, this paper has reviewed the different approaches of designing ecological compensation programs in the U.S., Australia, England, Germany and South Africa. The analysis has identified three design aspects that have considerable influence on its outcome: (1) integration of compensation programs with conservation landscape planning, (2) commensurability of ecosystem functions and (3) an open access centralised reporting system.

First, the integration of compensation programs with landscape planning concerns two things: to ensure that development in no-go areas is avoided and to choose compensation sites to fulfil landscape planning objectives. In regards to no-go areas, the findings from Table 1 indicate that the transition of the mitigation hierarchy from design to implementation can vary and that avoidance practices can be easily overlooked in the implementation phase even when the mitigation hierarchy is noted in the program’s design. Despite being formally prioritised in the compensation program design of the U.S. and English legislations, the mitigation hierarchy was inadequately implemented for reasons such as a lack of resources and ecological capacity by the local authorities. With proper adherence to the hierarchy, ecological compensation programs should contribute towards slowing down the rate of biodiversity losses. Developers would be inclined to choose areas with less ecological value, since these would be cheaper to compensate for. Evidence of avoidance in implementation is noted in Australia, Germany and South Africa’s programs, which have all incorporated conservation landscape planning in its compensation design and implementation. As observed in these three countries’ land use plans, this provides guidance for the local authorities to identify no-go areas where impacts must not be offset and determine whether avoidance should be used.

In regards to the compensation site, landscape planning allows for areas to be strategically selected by the public authorities. This would consider the compensation location to contribute towards habitat connectivity, buffer zones and ecosystem function (McKenney & Kiesecker, 2010, p. 174; Gardner et al., 2013, p. 1259). The probability for a higher conservation return can be increased by locating the compensation sites in areas that support regional conservation goals. Landscape planning objectives can be fulfilled through the integration with regional spatial plans for compensation sites in Australia, England and Germany. This is also observed by locating offsets in South Africa’s ‘offset receiving areas’, which are priority biodiversity areas identified by landscape planners. In the U.S., the strategic selection of compensation areas cannot be realised since the design of the program delegates the power to the developer and the mitigation banks to determine the site location freely. As the U.S. policy does not address the distribution of wetland ecosystem services in their decision-making or monitoring, it is unlikely that mitigation banks have concern for this (Murphy et al., 2009, p. 330). Ruhl and Salzman (2006, p. 9) note that entrepreneurial banks are profit seeking and have economic incentives to select compensation sites on land that is least costly for wetland credits produced.

Second, the level of commensurability assumed between impact and compensation sites is a controversial factor of the instrument’s design (Bull et al., 2012, p. 172; Sullivan, 2013, p. 84) In England’s loss-gain metric, full commensurability is assumed between sites based on its number of ‘biodiversity units’. This value is derived from a subjective evaluation of habitat ‘condition’ and ‘distinctiveness’. Habitat types are then conceptually commensurable as site exchanges are guided by their assigned values, where two sites with the same subjective score are assumed substitutable for another. This has received criticism as different
aspects of nature may not be compensated for (EAC, 2013; Evans, 2013). On the other hand, assuming no commensurability between sites would only allow a habitat of rare flower species, or wetland used by cranes, to be replaced by a habitat that supports the same species. This limits the number of suitable compensation sites.

A balance is then needed to determine a suitable level of commensurability when assessing the equivalence amongst sites. To overcome this, the German program assumes commensurability but only within the same ecosystem function, such as water infiltration or pollination (Küpfer, 2008). This ensures that the distribution of ecosystem services is considered. For example, functional compensation allows for impacts on bodies of water to be offset by re-engineering of a riverbank (Wende et al., 2005, p. 108). Focusing upon ecosystem functions when determining equivalency, rather than assuming full commensurability of biodiversity values, may lead to securing more ‘ecologically appropriate’ sites with viable biodiversity gains (Gardner, et al., 2013, p. 1259).

The role of weighing the equivalence between sites is usually done by a government representative, but this responsibility may be shared with an independent, external organisation with particular expertise on the ecosystem at stake in order to ensure both impartiality and capacity. Alternatively, weighing the equivalence between sites can be a co-responsibility of the government and such external organisation. This is observed in South Africa, where specialists advise on whether the proposed site appropriately compensates for the loss (DEA&DP, 2007, p. 37). Additionally, establishing a dialogue between landscape planners and an independent organisation mentioned earlier would be beneficial to ensure both commensurability objectives and integration with regional planning scenarios.

Third, the development of an open access centralised reporting system at the national level would help to collect and organise the information flow on the projects undertaken within each compensation program. This would address the limitations that are found throughout the monitoring and reporting data at various scales for all of the compensation programs. Hill et al. (2013, p. 1085) highlight the absence of an ‘easily-accessible, complete listing of existing mitigation projects in North Carolina with up-to-date information regarding project location, quality, compliance and credit yield.’ Without reliable data, the outcomes cannot be properly evaluated and it is unclear whether policy goals are achieved. Even in the case of U.S. and Germany where certain data was available at different geographical scales, it is not necessarily evident whether the ecological outcomes were beneficial (Bull et al., 2012, p. 7).

A centralised reporting system would not necessarily require monitoring activities to be standardised at the expense of a decentralised structure and site-specific needs, but rather serve to harmonise the data to relevant policy goals. The system would coordinate the overall project outcomes and increase the availability of information on a local, regional or national scale. This provides better access to data for the public, regulators, compensation providers and developers. Performance monitoring and evaluation reports should be publicly accessible to improve transparency and accountability of compensation providers. Additionally, a centralised system reduces transaction costs by providing a platform for regulators to trace development permit applications and monitoring reports.

In regards to the third research objective, the paper then evaluates the compensation programs’ ecological and social benefits. Four biodiversity and social safeguards are identified to protect these benefits.
Biodiversity safeguards are needed to ensure the quality of the implementation and ecological outcomes of compensation programs. The quality control often falls to either the local authorities or the developers; however, the gap observed in examined programs’ monitoring data suggests that both parties lack the capacity and resources to do this. Teeffelen (2014, p. 68) notes that in the case where biodiversity is exchanged, such as within ecological compensation programs, gaps in ecological capacity can be overcome by engaging the scientific community and practitioners. To address this, one potential procedural safeguard is to share the ex-ante equivalence weighing and ex-post monitoring responsibilities with the independent, external organisation mentioned earlier. This organisation may be involved in weighting equivalence, monitoring and ensuring the ecological quality of compensation programs. Compensation programs that are independently monitored have been regarded to be ‘more trust-worthy’ than those conducted by the developer (Conway et al., 2013, p.117).

Besides biodiversity safeguards for ensuring impartial weighting equivalence, three main social safeguards are identified throughout the examined programs policies with varying degrees of importance for each country. Substantive safeguards that protect local livelihoods were more prominent in South Africa where local communities are often dependent upon the biodiversity and ecosystem services to sustain their livelihoods. Safeguards should be in place to ensure social equity at both the impact and proposed compensation site. However, the evidence is not always clear whether the offset is able to provide livelihood benefits that are comparable to the impacted ecosystem (Darbi et al., 2009, p. 20). In high-income countries, the social safeguards tend to emphasise access to recreation as observed in the German legislation where the compensation site location is considered in regards to residential areas.

In both middle- and high-income countries, procedural safeguards that ensure stakeholder participation were evident. This ensures that compensation policies are relevant to the local context. South Africa’s policy identifies two groups of stakeholders, consulting the directly affected community as well as engaging the general society during the process of deciding the offset (DEA&DP, 2007, p. 46). Burgin (2008) notes that biodiversity offsets deal with compensating complex ecosystems and the current practices are underdeveloped, therefore a collective involvement of stakeholders is required to improve the outcomes. Hence, social safeguards requesting participation may also benefit biodiversity outcomes. The biodiversity and social safeguards discussed here suggest that democratic decision-making and transparent implementation are important for good performance.

It must be noted that evidence of avoidance practices in the mitigation hierarchy are not always easy to detect and the proportion of denied permits may not be a good indicator. For instance, this data does not consider instances where developers had consulted with planners beforehand and avoidance was undertaken prior to the formal request of a permit. Alternatively, developers may have taken into account the additional costs needed for compensation and actively avoided ecologically expensive land, before requesting a permit. Further data on the conditions of which development permits are approved and what avoidance or minimisation steps were taken would contribute to a better implementation of the mitigation hierarchy.

Lastly, it is unclear whether ecological compensation programs would be effective in replacing the biodiversity and ecosystem services of complex ecosystems even under ideal conditions of program design and implementation. Further research is needed to develop the field of restoration ecology; there is doubt as to whether current restoration practices are capable of delivering biodiversity gains that are adequate to achieving the goals of
compensation programs (Hilderbrand et al., 2005; Gardner, et al., 2013, p. 1259). Recognising the limits of restoration ecology and the ‘restoration myth’ may help put ecological compensation programs into perspective, for us to realise that not all ecosystems are replaceable.

6. Conclusion
Compensation programs have contributed towards protecting land for conservation purposes but there is little empirical evidence to suggest that any of these programs have achieved their policy goals. There are ecological as well as institutional reasons for this. The complexity of ecosystems creates difficulties in reducing biodiversity features to a single measure of loss and gain. Even perfectly designed and implemented programs may fail to deliver, due to the ecological complexity of restoring ecosystems. Additionally, guidelines for the mitigation hierarchy must be carefully emphasised so that the instrument does not facilitate a ‘license to trash’ and allow development that would have otherwise not occurred. The use of this controversial instrument requires considerable regulation and capacity to be implemented adequately, with democratically decided performance criteria. Well-designed safeguards that are adapted to the local and national contexts are needed to protect against the ecological and social risks associated with compensation programs.
References


