Monitoring of Biodiversity Offsetting in Sweden

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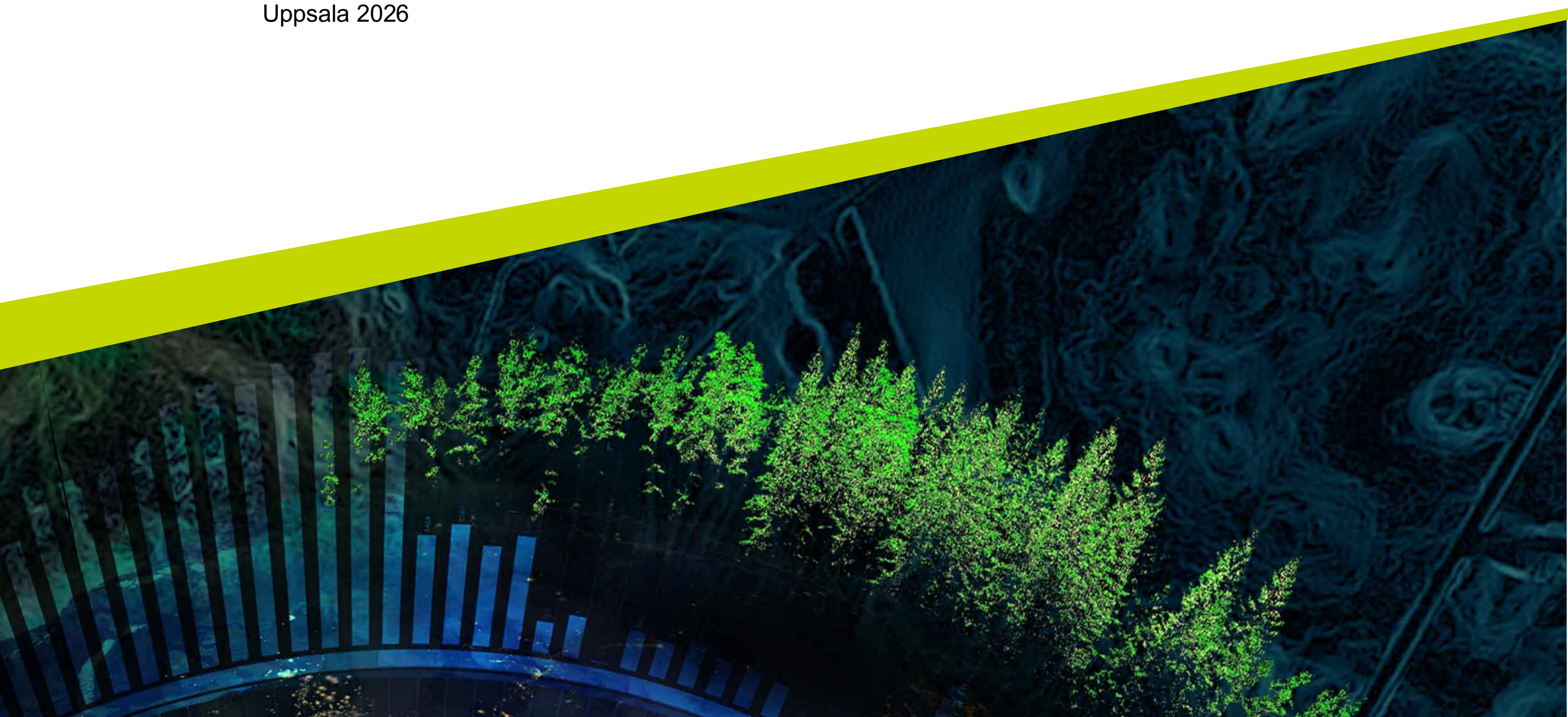

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Abstract

Biodiversity offsetting is a debated, yet increasingly used method to compensate for biodiversity loss caused by human activity, by creating gains somewhere else. However, offsets frequently fail to achieve their intended outcomes, and weak monitoring has been identified as a major contributor to this pattern. In Sweden, municipalities can apply voluntary offsetting under the Planning and Building act. Several Swedish municipalities have developed or are currently developing guidelines for biodiversity offsetting. The aim of this master thesis was to investigate how monitoring of outcomes of biodiversity offsetting, as applied by a Swedish municipality could be designed, using Uppsala municipality as an example. To address this question, I conducted two literature studies: a policy review of established offsetting frameworks, and a structured literature synthesis focusing on challenges and recommendations related to offset monitoring. My results confirmed the previously established finding that monitoring requirements are often limited even in well-established offsetting policies. The findings also showed that decisions made in the planning stage, such as the definition of counterfactual baselines and the choice of biodiversity metrics, highly affect the probability of offset success. A key finding was that simple habitat-based metrics commonly used to estimate losses and gains in biodiversity offsetting often fail to capture species-level biodiversity patterns, which may complicate the evaluation of offset outcomes. Therefore, an integrated approach is recommended, where monitoring design is closely linked to planning-stage decisions. Based on these findings, I provide recommendations related to offset monitoring in Sweden, including long-term monitoring, the use of biodiversity metrics matched to offset objectives, robust study designs coupled with control sites, and improved transparency and data availability.

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Abbreviations

Abbreviation	*Description*
NG	Net gain
NNL	No net loss
CLIMB	Changing Land Use Impact on Biodiversity
The SBM	The Statutory Biodiversity Metric

1. Introduction

The increasing biodiversity loss is one of the most urgent environmental challenges of our time (IPBES, 2019). Still, economic growth and a growing population have led to increased land-use change, further driving biodiversity declines (IPBES, 2019). Biodiversity offsetting (hereafter, offsetting) is a method for compensating human-caused impacts leading to ecological loss, by creating ecological gain at another site (Bull *et al*, 2013). This can be done through creation of new protected areas or preservation of already existing habitats (Gibbons & Lindenmayer, 2007). The origins of offsetting can be traced to wetland compensation policies established under the U.S. Clean Water Act in the 1970s, while international principles and guidelines were published by the Business and Biodiversity Offsets Programme (BBOP) in 2009 (BBOP, 2012: Bull *et al*, 2013). Offsetting is the last step of the mitigation hierarchy, stating that impacts should first be avoided, then minimized, then restored, and only as a last resort compensated for through biodiversity offsets (BBOP, 2012). One important principle of offsetting is additionality, meaning the biodiversity gains resulting from offset projects need to exceed what would otherwise be attained (BBOP, 2012). Moreover, offset projects commonly formulate goals of *no net loss* (NNL), or more recently *biodiversity net gain* (BNG) (BBOP, 2012; DEFRA, 2025). This means that the biodiversity loss at the impact site should not exceed the biodiversity gain at the offset site, or the biodiversity gain at the offset site should be greater than the loss from the impact, respectively. Offsetting projects should aim for ecological equivalence, or *like-for-like*, meaning that losses should be compensated with gains of the same type of biodiversity (BBOP, 2012; Bull *et al*, 2013). *Like-for-unlike* offsets, where losses of one biodiversity feature are compensated with gains of another, may be permitted when they are estimated to have higher conservation value.

Offsetting is controversial because it treats nature as replaceable units, and critics argue that this perspective can undermine the notion that nature has intrinsic value (Moreno-Mateos *et al*, 2015). Furthermore, ecological values are often unique and impossible to fully replace (Moreno-Mateos *et al*, 2015). However, advocates for offsetting suggest that ecosystems will be further exploited, and in the absence of better alternatives, offsetting can be a necessary tool to compensate for unavoidable losses (Maron *et al*, 2025).

Offsets have frequently been observed to fail (Lindenmayer *et al*, 2017; Theis *et al*, 2020). One of the main reasons behind this pattern is weak offset design (Theis *et al*, 2020). Moreover, poor offset implementation, including inadequate monitoring, has been identified as a major factor behind offset failure (Bull *et al*, 2014; Carreras Gamarra & Toombs, 2017; Moilanen *et al*, 2024). According to the BBOP, offsets should deliver long-term outcomes, which need to be monitored and evaluated (BBOP, 2012). Monitoring of offsetting outcomes is important not only to evaluate the individual offsetting case, but also to learn about which monitoring actions are most effective and under which circumstances (Ghijselinck *et al*, 2026). Improved monitoring design has been stressed in the scientific literature as key for offsets to demonstrate NNL (Josefsson *et al*, 2021; Souza *et al*, 2023; Zu Ermgassen *et al*, 2019).

1.1 Offsetting in Sweden

No official Swedish offsetting policy exists today, but the Swedish Environmental Protection Agency released guidelines on offsetting in 2016 (SEPA, 2016). Offsetting can be applied under the Swedish environmental code (1998:808) and the Planning and Building Act (2010:900). Within the environmental code, compensation is required when protected areas are affected by exploitation, for example nature reserves or Natura-2000 areas. Offsetting can also be applied voluntarily under the Planning and Building act, primarily through municipal planning processes and local strategies aimed at conserving biodiversity (Hanson *et al*, 2021). In these cases, offset design is decided by the responsible authority (Olsson & Hansson, 2023).

There is an increasing interest among Swedish municipalities to work with offsetting (Olsson & Hansson, 2023). For example, in Lomma municipality previous offsetting projects include planting 25 solitary trees to compensate for a lost meadow (*like-for-unlike*) and planting 2667 m^2 of forest to compensate for 2000 m^2 of exploited forest land (*like-for-like*). An evaluation of offsetting in Lomma municipality found that a lack of data made it difficult to evaluate offsetting outcomes, resulting in one third of the included projects being excluded from the analysis (Morell *et al*, 2026). Uppsala municipality decided in June 2024 on *Guidelines for Nature Consideration and Ecological Compensation in Land-Use Change* (author's translation), which established a standardized approach for minimizing biodiversity loss associated with development and land exploitation (Uppsala kommun, 2024). The purpose of this master thesis was to examine how Swedish municipalities can design monitoring of outcomes of offsetting, using Uppsala municipality as an example. Recently, a new method for quantifying biodiversity losses and gains was introduced in Sweden, *Changing Land Use Impact on Biodiversity* (CLIMB) (EcoGain, 2023). The method is inspired by the English 'Biodiversity Net Gain' system, and could potentially become an important tool for offsetting in Sweden in the future.

Unlike traditional ecological monitoring, offset monitoring aims to inform on biodiversity patterns relative to losses that have already occurred. As a result, offset monitoring design is highly connected to decisions made during the planning stage. For example, using the same biodiversity metric in both planning and evaluation is preferable to ensure comparability between losses and gains (Cousins, 2016). With this in mind, I chose to evaluate offset monitoring not only by looking at program design, but also in relation to factors in the planning stage that may affect the possibility of reaching NNL/NG. 2 literature studies were conducted to answer the following questions:

1: How is offset monitoring designed in well-established offsetting policies?
2: How does CLIMB work?
3: What are the key challenges associated with offset monitoring?

4: What suggestions are given in the scientific literature to overcome these limitations?

5: Based on these findings, how can offset monitoring be designed in Swedish municipalities?

2. Methods

I conducted two separate literature searches: one policy-oriented literature study (Literature study 1), and one structured literature-based synthesis study (Literature study 2). Literature study 1 aimed to answer Research questions 1 and 2, while Literature study 2 aimed to answer Research questions 3 and 4. Research question 5 will be addressed in the discussion.

2.1 Literature Study 1

To answer Research question 1, relevant offsetting policies were identified through the reading of publications included in Literature study 2, where certain offsetting policies were discussed repeatedly. The selected policy frameworks were from: England, France, The U.S and Australia (State of Victoria). Also, 11 evaluations of these policy frameworks were identified. 9 through backward reference searching of the publications included in Literature study 2, while 2 were already included in Literature study 2. Research question 2 was addressed by identifying and reviewing relevant documents published by the company EcoGain, developers of CLIMB.

2.2 Literature Study 2

Literature study 2 was conducted as a structured literature-based synthesis study, supplemented with backward reference searching (Wohlin, 2014). A search string was created with inspiration from Josefsson *et al* (2021). I searched titles, abstracts and keywords in Scopus and Web of Science (WoS) with the following search string:

("ecological compensat*" OR "biodiversity offset*" OR "no net loss" OR "net gain" OR "conservatory mitig*" OR "restoration*" OR “conserv*”) AND ("monitor*" OR "ecological impact assessment" OR "assess*" OR "evaluat*" OR "effect*" OR "outcome*") AND (“biodiversity metric*” OR “biodiversity account*” OR “offset metric*” OR “biodiversity measure*”) AND NOT ("carbon offset*" OR "carbon accounting*")

The initial search resulted in 330 hits in Scopus. After title screening, it was observed that many publications were unrelated to offsetting. Therefore, I decided to remove the word “conserv*” from the search string. The revised search resulted in 76 hits in Scopus. The same search in WoS resulted in 4 additional publications.

All titles were screened, and 32 publications were selected for abstract screening. Publications were included if they were published after 2015 and addressed challenges or proposed solutions related to offset monitoring or planning. Publications focusing on carbon offsetting were excluded. In addition, studies focusing primarily on ecosystem services were excluded, as this study focused specifically on biodiversity outcomes.

Following abstract screening, 19 publications met the inclusion criteria. An additional 15 publications were chosen through backward reference searching of the included studies. Also, 5 articles published after the initial search in spring 2025 were included.

3. Results

3.1 Literature study 1

3.1.1 Monitoring in well-established offsetting policies

England

A provision in the English Environment Act from 2021 states that land development in England should result in a Biodiversity Net Gain (BNG) - meaning development resulting in lost natural habitat has to be compensated for with biodiversity offsets (DEFRA, 2025b). According to the provision, new developments can only be approved if the developers report how a 10% BNG will be obtained in their application. The 10% BNG requirement has been legally binding since February 2024.

To measure biodiversity quantitatively, project developers are required to use the Statutory Biodiversity Metric (SBM) (DEFRA, 2025a). The SBM assigns biodiversity units to various habitat features based on how strongly they indicate overall biodiversity. The habitat features are: Habitat area; habitat condition; measured as good, moderate or poor; habitat distinctiveness, describing the ecological importance of the habitat; strategic significance, meaning how well the habitat aligns with local or national biodiversity strategies. These units are then multiplied to obtain a final biodiversity unit. The metric can be adapted to offsets specifically by adding variables, for example units based on the time required for a created habitat to reach the goal condition (maturity time), and the difficulty to reach that condition (DEFRA, 2025a).

To ensure 10% NG, the offset sites have to be monitored over 30 years, using a Habitat Management and Monitoring Plan (HMMP). Natural England has published a template for HMMPs, however using the template is not mandatory - yet advisable (Natural England, 2023). The template does not specify how often the site should be monitored, or using what method. It is instead mentioned that the person using the template is responsible for determining the monitoring design, which should be based on ecological expertise (Natural England, 2023).

While many individual offsetting projects claimed 10% net gain, the BNG program as a whole showed net loss when evaluated (zu Ermgassen *et al*, 2021; Rampling *et al*, 2023). This was partly explained by the trade of big, low-scoring impact areas for smaller offset areas, based on the expectation that the smaller areas would achieve higher values in the future (Ermgassen *et al*, 2021). Insufficient monitoring of on-site offsets was also identified as a reason for NL (Rampling *et al*, 2023). However, the policy was updated in 2024 and now includes mandatory monitoring of on-site offsets (DEFRA, 2025b). No evaluations addressing the updated policy were found.

Australia - The State of Victoria

A policy including offsetting was introduced by the State of Victoria in 2002 (NRE, 2002). In the current system, biodiversity losses and gains are measured in General and Species Habitat Units (HUs) (DELWP, 2025). These are based on: ‘Habitat Hectares´ (HH), which is a metric combining vegetation condition and area relative to benchmarks for Ecological Vegetation Classes (EVCs); A landscape factor, which is based either on a score of the ‘strategic biodiversity value’ of a site, or on a ‘habitat importance score’ for threatened species. Multipliers are also added to account for the uncertainty of the estimated future gain (DELWP, 2025).

In terms of monitoring, the landowners need to demonstrate compliance by filling in an offset managing plan (OMP) annually during a ten year period (DEECA, 2025). The OMP is largely constructed around the Management standards for native vegetation offset sites, which are provided by the Department of Energy, Environment and Climate Action in Victoria (DEECA) (DEECA, 2025). These standards include several mandatory actions to ensure the protection of the habitat hectares of the offset site, including appropriate fencing, control of weeds and pests, and retention of native vegetation. Compliance is also monitored on a program level using voluntarily submitted data on approved impacts and offsets, field visits and internal compliance records (e.g. OMPs) (DEECA, 2023). However, how biodiversity should be monitored more specifically, for example which metrics should be used to ensure NNL, was not stated in these documents.

One evaluation showed that the State of Victoria did not reach the NNL goal (VAGO, 2022). The report mentioned insufficient biodiversity assessment and monitoring as two of the main weaknesses of the policy. Furthermore, DEECA's own evaluation found that the low submission rate of voluntary data, with only 40% of responsible authorities providing information, made it difficult to evaluate NNL (DEECA, 2023).

The U.S. - Conservation Banking and Wetland Mitigation Banking

The U.S. offsetting system is largely based on conservation banking and wetland mitigation banking (U.S. Fish and Wildlife Service, 2003; USACE & EPA, 2008). Conservation banking is described in a 2003 policy, and has the purpose of compensating land exploitation impacts affecting species protected under the Endangered Species Act (U.S. Fish and Wildlife Service, 2003). A conservation bank is a piece of permanently protected land of ecological importance that can be sold as credits to companies seeking to compensate for biodiversity losses off-site. Wetland mitigation banking was introduced in 1995, and enabled credits bought to compensate for smaller impacts to be merged, enabling the creation and enhancement of bigger aquatic habitats, who often have a higher long-time value compared to smaller sites (Federal Interagency Wetland Mitigation Banking Guidance, 1995; USACE & EPA, 2008). Unlike conservation banking, wetland mitigation banking focuses on the compensation of lost aquatic habitats particularly, rather than the loss of habitats important for certain endangered species. Accordingly, it is motivated in the wetland mitigation Banking policy that preservation of existing habitats should be discouraged (USACE & EPA, 2008). This is because protecting existing wetlands does not generate additional ecological functions, but merely prevents further loss. Conversely, due to the focus on specific species of conservation banking, it is encouraged to preserve existing habitats in this policy (U.S. Fish and Wildlife Service, 2003). Neither the wetland mitigation nor the conservation banking policy describe how biodiversity should be measured in the impact area and/or in the bank site. However, using the same method in the impact area as was used on the bank site is advisable in the conservation banking policy, to ensure comparability between the sites (U.S. Fish and Wildlife Service, 2003). A broader compensatory mitigation policy introduced in 2016 expanded the frameworks to include NG objectives, but these were later removed in a revised policy published in 2023 (U.S. Fish and Wildlife Service, 2016; 2023).

Both the Wetland Mitigation and the conservation banking policy emphasize the importance of management and monitoring of the bank site to ensure NNL (USACE & EPA, 2008; U.S. Fish and Wildlife Service, 2003). The conservation bank owner is responsible for monitoring the site, and the banking agreement should include a plan describing how the monitoring will be carried out and how often. It needs to be specified in the banking agreement: how the biological goals will be evaluated, hence what indicators or metrics will be used; how compliance will be verified; how the results from the evaluation will be used to enable adaptive management. According to the Wetland Mitigation policy, monitoring should be undertaken for at least 5 years (USACE & EPA, 2008).

Several studies found that mitigation banking often leads to NL (Levrel *et al*, 2017; Murphy *et al*, 2006; Theis *et al*, 2022). A 2022 evaluation of 12,756 mitigation bank transactions across the U.S. found that 25% of transactions failed to achieve no net loss, largely due to the frequent use of preservation and re-establishment, as well as poor ecological equivalency between impacted and offset sites (Theis *et al*, 2022). Another evaluation from 2017 found that although 58,575 ha of wetlands in Florida were restored through mitigation banking between 2001 and 2011, there was a net annual loss of 5,600 ha of wetland habitat, possibly because many impacted wetlands were not protected under the Clean Water Act (Levrel *et al*, 2017). Levrel *et al* also recognized that evaluating mitigation banking is challenging partly due to the lack of monitoring (Levrel *et* *al*, 2017). Furthermore, a conservation banking evaluation from 2017 showed that while most individual banks monitored their sites, the absence of systematic monitoring made it impossible to evaluate whether the program succeeded as a whole (Carreras Gamarra & Toombs, 2017).

France

France has been working with offsets since 1976, when the mitigation hierarchy was incorporated into French environmental law (Quétier *et al*, 2014). Yet, the French offsetting system was poorly applied until an updated policy - following the Environmental Impact Assessment Directive (EIA) - came out in 2012 (MEDDE, 2012; Quétier *et al*, 2014). The 2012 French policy includes some of the key concepts present in other offsetting policies, such as ecological equivalency, additionality and requirements for long-term monitoring and management to ensure the durability of offsets (MEDDE, 2012). However, the policy has received criticism for not providing enough detail about how these concepts should be implemented in practice (Quétier *et al*, 2014). Unlike the English BNG system and the Victorian policy, guidance about metrics used to compare biodiversity losses with gains is not given in the French offsetting policy. Moreover, although monitoring of offset sites is formally required under the French 2012 policy, it is rarely applied in practice (MEDDE, 2012; Bezombes *et al*, 2019). This is likely linked to the lack of baseline data, the lack of standardized ecological indicators, and the absence of standardized management plans (Bezombes *et al*, 2019).

In 2016, France introduced the *Sites Naturels de Compensation* (SNC) system, inspired by the U.S. mitigation banking model (French Ministry of Ecological Transition, 2023; Pagney, 2022). Similar to the American model, the purpose of the SNC system was to enable biodiversity gains to be attained before the loss, and avoiding offset patches by merging smaller units into one. However, a 2022 evaluation found that the SNC system faced challenges in showing ecological equivalency, partly due to less strict requirements than those applied in the American system (Pagney, 2022).

3.1.2 CLIMB

Similar to the English Statutory Biodiversity Metric, in CLIMB habitat features are assessed and translated into biodiversity units (CLIMB units) (EcoGain, 2023). CLIMB is an aggregated multi-metric centered around the area, conservation value and landscape value of an impact/offset site, each of which are assigned factors based on their indication for overall biodiversity (Table 1). Below is a summary of the method (For full equations see supplementary table 1 in Appendix):

The conservation value follows the Swedish standard for biodiversity assessment (SS199000:2023). In this framework, each biotope within a site is assigned a 'conservation value class' from 1 to 7, where lower numbers indicate higher ecological values. These classifications are primarily based on expert field assessments rather than quantitative ecological data. In CLIMB, the assigned conservation value class of each biotope is translated to a conservation value factor, where the factor decreases by half for each class in the scale from 1 to 5. The conservation value factor decreases more for class 6 and 7 due to the low biodiversity associated with these classes. The landscape value in CLIMB is also based on the Swedish standard for biodiversity assessment (SS199000:2023). In the Swedish standard, a 'valuable landscape' (värdelandskap) is characterized as "a landscape area of special significance for biodiversity" (SS199000:2023, author's translation). This means adding a spatial layer when evaluating a habitat, by considering landscape features important to support biodiversity and maintain it over time, such as high connectivity and low fragmentation (EcoGain, 2023). A 15 % increase in CLIMB units are added to the biotope when it is located within a 'valuable landscape' of the same nature type/biotope. Furthermore, CLIMB also considers: delivery time, where a deduction of CLIMB units is applied the longer the estimated time is for a biotope to move from a lower 'nature value class' to a higher one; difficulty level, meaning CLIMB units are removed when it is estimated as difficult for a biotope to reach its goal state; indirect effects, where CLIMB units are removed if a biotope is located adjacent to an impact site.

CLIMB can be adapted to compensation projects specifically by adding a factor to account for when the offset site is located far away from the impact site, and for additionality. The factors are then combined, producing a final CLIMB unit.

No evaluations of CLIMB were found, which is not surprising considering the young age of the method.

Table 1:

Table showing the different variables in CLIMB (EcoGain, 2023).

Variable	***Explanation***	***Specification***
Area	Area in hectares.	Area in hectares.
Conservation value	Based on SS 199000:2023. A biotope is assigned a class (1-7), where lower classes indicate higher ecological values.	The 'conservation value factor' increases exponentially from 1 (class 5) to 16 (class 1)
Landscape value	Based on SS 199000:2023. A rough measure of connectivity and fragmentation.	CLIMB units increase with 15% if a biotope is within a 'valuable landscape'
Delivery time	Time for a biotope to move between 'conservation value classes'.	Estimated in intervals, and simplified to number of years. For each year until the goal state is reached, 1 % of the factor is removed.
Difficulty level	An estimate for how difficult it is for a biotope to reach a goal state.	Measured as easy (factor=1), intermediate (=0.67) or difficult (=0.33).
Indirect effects	An estimate accounting for indirect effects in areas adjacent to impact sites.	33% of the CLIMB units are removed from the affected area.
Off- site: Distance to impact site	Distance to impact site.	If the offset site is located more than 50km away from the impact site, the factor will decrease with 5% (from 1), and decrease additionally if the site is also in a different biogeographical region.
Off-site: Additionality	The practitioner decides if the planned offset actions are estimated to lead to additionality.	If yes, additionality factor=1, if no, ``=0

3.2 Literature study 2

3.2.1 Offset planning

The mitigation hierarchy

Several studies highlighted the importance of following the mitigation hierarchy with care to achieve NNL (Marshall *et al*, 2022; Moilanen *et al*, 2024; Souza *et al*, 2023). The mitigation hierarchy is a fundamental part of offsetting, yet the steps of the hierarchy have been reported as unclear (Ghijselinck *et al*, 2026). Ghijselinck *et al* used the term "typological drift" to describe the gradual blurring of the four steps of the mitigation hierarchy - avoidance, minimization, restoration and compensation - when they are treated as interchangeable rather than applied as a true hierarchy, thereby undermining the function of the hierarchy (Ghijselinck *et al*, 2026). Perhaps most notably, avoidance - the first hierarchical step - is rarely used (Ghijselinck *et al*, 2026), despite that core habitats cannot be offset through either habitat-based or species-based metrics (Marshall *et al*, 2022).

Avoided loss

A number of articles pointed out limitations regarding avoided loss (Ermgassen *et al,* 2019; Gordon *et al*, 2025; Maron *et al*, 2015; Moilanen & Kotiaho, 2018b; Peterson *et al*, 2018). Avoided loss or averted loss is a type of offsetting where already existing habitats are protected, thereby preventing future degradation (Gordon *et al*, 2025). This could be especially beneficial for specialist species, since habitats that already have high ecological values and provide important niches can be protected (Moilanen & Kotiaho, 2018b). A recent review found that none of the evaluated avoided loss offsets showed NNL (Ermgassen *et al*, 2019). This outcome is likely partly explained by the inevitable loss of habitat area associated with avoided loss (Ermgassen *et al*, 2019). Furthermore, the difficulty establishing a counterfactual baseline could hinder avoided loss offsets to achieve NNL, and also makes correct evaluation - and therefore monitoring - more difficult (Gordon *et al*, 2025; Maron *et al*, 2015). Avoided loss can only accomplish NNL when areas risking future degradation are protected (Gordon *et al*, 2025). However, previous findings show that predicted background losses were frequently overestimated, leading to NL and thereby further biodiversity declines (Ermgassen *et al*, 2019; Maron *et al*, 2015). For example, Maron *et al* found that most Australian offsetting policies assumed background losses of between 0.36-4.2% a year, while real deforestation rates were estimated as 0-0.5% annually (Maron *et al*, 2015). Increased transparency about the uncertainty of counterfactual baselines, as well as more accurate estimations of them, were mentioned as important steps to improve the effectiveness of avoided loss (Maron *et al*, 2015; Moilanen & Kotiaho, 2018a).

Multipliers

High multipliers were recurrently discussed in the literature as a significant mechanism for achieving NNL (Ghijselinck *et al*, 2026; Moilanen & Kotiaho, 2018a; Moilanen *et al,* 2024), and their use has been reported as a reason behind successful offsetting outcomes (Ermgassen *et al*, 2019). A multiplier is a correction factor used in offsetting to compensate for uncertainties, and is typically defined in terms of habitat area, requiring the offset site to be larger than the impact site (Moilanen & Kotiaho, 2018a). However, multipliers can also be expressed in other ways, for example as biodiversity units in the English SBM (Marshall *et al*, 2023; Moilanen & Kotiaho, 2018a). Offsetting involves many uncertainties, such as the inability to ensure permanent protection of offsetting sites, the unclear recovery of populations, and the often long time lags for offset sites to reach a mature state (Ghijselinck *et al*, 2026; Moilanen & Kotiaho, 2018a; Souza *et al*, 2023). The poor understanding of the effects of monitoring actions also adds to the complex and unpredictable outcomes of offsetting (Moilanen *et al*, 2024). If adding multipliers to all uncertainties, the offsetting ratios would increase rapidly, with estimates suggesting ratios as high as 10:1 (Moilanen & Kotiaho, 2018b).

Even though the uncertainty of offsetting is widely recognized, one review found that multipliers were only applied in half of the included offsetting policies (Marshall *et al*, 2024). While recognizing multiplies as a useful tool for compensating the lack of certainty in offsetting, the authors also noted that the definition of multipliers differed broadly between policies, and that the different approaches made evaluation of their efficiency more difficult (Marshall *et al*, 2024).

3.2.2 Metric choice

Limitations of habitat-based metrics

Habitat-based metrics are biodiversity metrics that use vegetation attributes as proxies for biodiversity and are more frequently used in offsetting compared to conservation planning and the ecological literature (Marshall *et al*, 2020). Area-based metrics were reported in more than 70% of offset policies, compared to 56% and 59% in conservation planning and 49% and 15% in the ecological literature, respectively (Marshall *et al*, 2020). Habitat-based metrics are often used in offsetting due to their simplicity, which is not trivial, since practitioners have highlighted the need for biodiversity metrics that are time-efficient and cost-effective (Carreras Gamarra *et al*, 2018; Marshall *et al*, 2023). However, their use is widely criticized in the literature (Bush, 2024; Cousins, 2016; Duffus *et al*, 2025a; Duffus *et al*, 2025b; Ermgassen *et al*, 2019; Ermgassen *et al*, 2021; Marshall *et al*, 2020; Marshall *et al*, 2021; Marshall *et al*, 2024; Millon *et al.*, 2021; Rampling *et al.*, 2024; Simpson *et al*, 2022). One recurrent argument is that simple habitat-based metrics fail to represent individual species, especially those dependent on more complex habitat features (Duffus *et al*, 2025a; Marshall *et al*, 2024; Marshall *et al*, 2021). Most studies evaluating how well habitat-based metrics explain species diversity were done in England, evaluating the SBM (Duffus *et al*, 2025a; Hawkins *et al*, 2022; Marshall *et al*, 2024). The SBM did not explain the biodiversity of species of conservation priority in England (Hawkins *et al*, 2022), ground invertebrates (Duffus *et al*, 2025a), butterflies or birds

(Marshall *et al*, 2024). For example, invertebrates and birds are dependent on habitat heterogeneity, which might not be captured by the simple design of the SBM condition score (Duffus *et al*, 2025a; Duffus *et al*, 2025b; Marshall *et al*, 2024). One study was carried out in an Australian context, finding that simple habitat-based metrics did not offset population declines (Marshall *et al*, 2021).

Another common critique of habitat-based metrics is the heavy focus on habitat area (Ermgassen *et al*, 2019; Ermgassen *et al*, 2021; Rampling *et al*, 2019). The species-area relationship is well-known in ecology, describing how species richness increases with habitat area, typically modelled as a power function (S=Ca^z) (Connor & Mccoy, 1979). However, habitat area alone captures neither habitat quality nor species identity (Zu Ermgassen *et al*, 2019). In offsetting, simple area-only metrics are expected to favor low scoring but big offset areas, as this is most economically beneficial for the landowners (Simpson *et al*, 2022). Beyond influencing landowner decisions, the focus on area can also affect how offset success is evaluated. In a 2019 global review Ermgassen *et al* found that many offsets reporting NNL were assessed using area-only metrics, and were commonly associated with high offset multipliers, meaning offset sites were bigger than the corresponding impact sites (Zu Ermgassen *et al*, 2019).

The opposite scenario - where large impact areas are traded for small offset areas - can occur when area is treated as an isolated factor within an aggregated multi-metric (Zu Ermgassen *et al*, 2021). For example, the English SBM treats area linearly, such that biodiversity units scale proportionally with site size (DEFRA, 2025a). Consequently, a 10-ha site with a condition score of 3 will have the same final score as a 20-ha site with a condition score of 1.5 (DEFRA, 2025a). An evaluation of the BNG in England showed 20% NG, but a 34% loss of greenspace, explained by the trade of low-scoring but big impact areas for high-scoring but small offset areas (Zu Ermgassen *et al*, 2021). Notably, the high scores of the small offset sites were based on promises of future gains, creating a risk that the gains will never be delivered. The strong emphasis on on-site delivery of the BNG could also contribute to the pattern of trading big areas for small, as on-site offsets tend to create multiple small habitat patches within the same development (Zu Ermgassen *et al*, 2021; Rampling *et al*, 2019). Moreover, as developments are often located in urban areas, these small patches are subject to intense human pressure, potentially putting their long-term ecological value at risk (Zu Ermgassen *et al*, 2021; Rampling *et al*, 2019).

Improve habitat-based metrics with additional habitat and species data

In response to the limitations of habitat-based metrics used in offsetting, a recurring recommendation was to complement existing metrics with species data (Duffus *et al*, 2025b; Marshall et al, 2020; Marshall *et al*, 2021; Marshall *et al*, 2024; Moilanen & Kotiaho, 2018a; Moilanen *et al*, 2024). Currently, simple presence/absence data is required in many policies, but actual species data - that could make the absence/presence data more meaningful - is rarely used (Marshall *et al*, 2023; Marshall *et al*, 2024). Species richness is included in the NSW policy (Gibbons *et al*, 2018), but species richness has known disadvantages, like being sensitive to scale and failing to capture important information such as species abundance and identity (Fletcher *et al*, 2025). To refine habitat-based metrics, it was suggested to incorporate species-specific metrics, such as habitat suitability and abundance (Marshall *et al*, 2022), as well as metrics focused on selected target species (Duffus *et al*, 2025b). One recommendation was to align species-based metrics with conservation/offsetting goals (Marshall *et al*, 2020). Moilanen *et al* highlighted using species-based metrics in offset monitoring to correctly quantify gains, however they also noted that species-based metrics can miss species-interactions, leading to unintended consequences for non-target species (Moilanen *et al*, 2024). Moreover, generating species data is resource-intensive, and can therefore be difficult to incorporate into offsetting in a robust way (Moilanen & Kotiaho, 2018b).

In addition to adding species data, it was suggested to refine habitat-based offsetting metrics with additional habitat data (Cousins, 2016; Duffus *et al*, 2025A; Duffus *et al*, 2025b). Today, the English SBM only considers structural connectivity indirectly through the strategic significance multiplier. Also, the measurements are simplified, and do not look into for example functional permeability, making them weak proxies (Duffus *et al*, 2025b). It was emphasized to consider more complex habitat characteristics in the SBM, such as habitat heterogeneity and more thorough measurements of structural connectivity (Duffus *et al*, 2025a; Duffus *et al*, 2025b)

Alternative metrics

Beyond strengthening existing habitat-based offsetting metrics with additional species and/or habitat data, 2 publications presented novel offsetting metrics (Cousins, 2016; Bush *et al*, 2024). For example, Cousins *et al* tested combinations between over 45 different biodiversity metrics, producing a multi-metric index including the following 4 metrics: beta diversity (vascular plants), temporal risk (time to maturity), habitat rarity and structural connectivity (Cousins *et al*, 2016). Notably, the multi-metric index did not correlate with the English SBM when tested across several different habitat types (Cousins *et al*, 2016). One study suggested using irreplaceability as an offsetting metric (Bush et al, 2024). Irreplaceability is a metric used in conservation, where a site is valued according to its ecological contribution relative to other sites across the landscape, while also considering what is economically most efficient (Bush et al, 2024). Using irreplaceability would enable offsetting to move beyond 'like-for-like' offsets, which could increase the probability of NNL by assessing biodiversity from a landscape perspective, and not only within a specific site (Bush *et al*, 2024).

Assessment of Biodiversity metrics

In many European countries, for example France, it is not specified which biodiversity metrics to use in offsetting (Bezombes *et al*, 2018). This leads to unclear guidance and complicates comparison of outcomes between projects, which is important to understand the reasons behind project failure (Bezombes *et al*, 2018; Carreras Gamarra *et al*, 2018). This calls for standardized evaluation tools, aiding practitioners in choosing and assessing biodiversity metrics (Bracy

Knight *et al*, 2019). For example, a Metric Decision Tree has been developed, where the best suited offsetting metric is recommended based on the focus of the offset (Carreras Gamarra *et al*, 2018). Also, at least one Biodiversity Metrics Framework has been proposed, aiding practitioners in comparing biodiversity mitigation metrics with scientific evidence, identifying potential gaps between conservation goals and the metrics used to assess them (Bracy Knight *et al*, 2019). Both the Metric Decision Tree and the Biodiversity Metrics Framework are based on Noss' (1990) well-established hierarchy of biodiversity, which defines biodiversity through three main attributes (composition, structure and function), across four scales ranging from genetic to landscape level (Noss, 1990). The credibility and effectiveness of offsetting could be increased by combining tools such as for example the Biodiversity Metrics Framework and the Metrics Decision Tree (Bracy Knight *et al*, 2019).

3.2.3 How and when

How should offset monitoring be designed?

Besides the central aspect of *what* is monitored in offsetting, *how* offsetting is monitored is also of great importance (Moilanen *et al*, 2024). While many of the reviewed studies included in this thesis discussed this indirectly, for example by highlighting the importance of correct design to ensure NNL, only a few studies directly addressed *how* offset monitoring should be designed (Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024). From these, two challenges were identified. First, biodiversity losses were rarely (if ever estimated, making it impossible to correctly quantify gains (Josefsson *et al*, 2021). Second, the lack of control sites prevented the establishment of a counterfactual baseline, creating a risk that natural variation was mistaken for biodiversity gains (Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024).

Ecological monitoring programs can be designed in a number of ways (Moilanen *et al*, 2024). Common approaches include: After-only, the affected sites are sampled only after the impact; Before-after (BA), impacted sites are sampled both before and after the impact; Control-impact (CI), impacted sites are sampled only after the impact, but control sites are added to reduce background noise; Before After Control Impact (BACI), impacted sites and control sites are sampled before and after the impact; RCT, plots (or sites) are randomly assigned to impact or control groups before the impact is applied. All plots are then sampled after the impact (Christie *et al*, 2019). For offsetting monitoring, BACI-designs have been recommended for ideal statistical accuracy (Josefsson *et al*, 2021; Moilanen *et al*, 2024). By sampling both impact and control sites, real biodiversity differences can be distinguished from variations that would have occurred anyway (Josefsson *et al*, 2021; Moilanen *et al*, 2024). While being a robust study design with a low risk of bias, BACI-designs require multiple replicates, meaning it can take many years to obtain usable results (Moilanen *et al*, 2024). This potentially complicates its application in offsetting monitoring, since offsetting sites usually are monitored for a limited number of years (Moilanen *et al*, 2024).

How long and how often?

It was frequently advised in the literature to apply long-term monitoring to ensure NNL/NG (Cousins, 2016; Marshall *et al*, 2022; Moilanen *et al*, 2024; Souza *et al*, 2023). Ecological responses commonly take multiple years - often decades - to mature (Cousins, 2016). For example, population gains can take over 30 years to become detectable (Marshall *et al*, 2022). If offset sites are monitored shorter than the maturity times, time-lags can hide ongoing declines, creating a risk that gains are overestimated (Marshall *et al*, 2022; Moilanen *et al*, 2024). Moreover, it was stressed that adaptive management should be an integrative part of long-term monitoring, as it enables correction of offset actions, thereby increasing the chance of achieving NG (Moilanen *et al*, 2024). An alternative way to mitigate long time-lags without extending monitoring periods is the use of habitat banking (Cousins, 2016; Rampling *et al*, 2023). Yet, because habitat banking in some cases can be viewed as avoided-loss offsets, demonstrating additionality can be more challenging (Cousins, 2016). It was also suggested to add additional multipliers if the estimated maturity time exceeds the feasible monitoring or evaluation time (Moilanen & Kotiaho, 2018a).

None of the publications included in this study discussed how often offset sites should be monitored. However, Moilanen *et al* emphasized that adaptive management requires accounting for temporal variability, such as yearly fluctuations in populations (Moilanen *et al*, 2024).

3.2.4 Public databases for increased offset knowledge and transparency

Currently, there is a knowledge gap between what offsets claim to achieve on paper, and what they deliver in practice (Josefsson *et al*, 2021). To bridge this gap, the establishment of public databases has been proposed, where outcomes and monitoring data from offsets would be publicly available (Gordon *et al*, 2025; Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024). Such an approach offers two key benefits. First, it would enhance transparency, addressing one of the most common critiques of biodiversity offsetting (Kujala *et al*, 2022). Second, it could improve the effectiveness of offsets by enabling knowledge sharing about which actions have proven successful and which have not (Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024).

4. Discussion

4.1 Limited monitoring requirements in well-established offsetting policies

Neither of the offsetting programs included in this study achieved NNL when evaluated. The lack of systematic monitoring was recurring in policy evaluations across all offsetting policies. For example, in the State of Victoria, a lack of voluntarily submitted data from practitioners made it difficult to evaluate NNL, and in the U.S the lack of systematic monitoring made program evaluation impossible (Carreras Gamarra & Toombs, 2017; DEECA, 2023). Furthermore, in England, a offsetting program evaluation found that while individual offsets frequently claim 10% NG, the program as a whole still resulted in NL (Zu Ermgassen *et al*, 2021). These findings highlight the importance of monitoring on different scales, both the monitoring of individual offsetting projects, but also monitoring of entire programs (SEPA, 2023).

Policy support for monitoring was scarce in the French and American offsetting programs (Bezombes *et al*, 2019; Levrel *et al*, 2017; Murphy *et al*, 2006; Theis *et al*, 2022; Quétier *et al*, 2014). This was also reflected in practice. For example, in France 25% of the reviewed cases did not provide a management plan, indicating that no monitoring was conducted (Bezombes *et al*, 2019). England and the State of Victoria offer monitoring templates, which can provide clarity on what is expected according to policy (DEECA, 2025; Natural England, 2023). However, in England using the template is not mandatory, and the template itself leaves several aspects of the monitoring design to the practitioner, including metric choice and monitoring frequency. Although some flexibility is necessary given the variation between offsets, the lack of mandatory reassessments using the SBM during the 30-year monitoring period can be problematic, since the 10% NG target is based on that metric (DEFRA, 2025). In the State of Victoria in Australia the focus of monitoring is compliance, meaning to ensure the protection of the habitat units. The assumption that protection and management of the habitat units lead to NNL can be viewed as a clear policy weakness. Together, these results show that high-quality monitoring instructions in offsetting policies is key not only to enhance its use, but also to increase the probability of NG/NNL. Considerations related to the design of such monitoring programs will be further discussed in the section “Offset monitoring in Sweden”.

4.2 Offset planning

Several limitations regarding offset planning were identified in the literature. Neither habitat-based nor species-based habitats could offset core habitats, stressing the importance of following the mitigation hierarchy with care, where avoidance is the first step (Ghijselinck *et al*, 2026; Marshall *et al*, 2022). This is partly reflected in the guidelines from Uppsala municipality, where Class 1 sites are excluded from offsetting (Uppsala Kommun, 2024). Furthermore, establishing a correct counterfactual baseline is a key step in offset planning (Maron *et al*, 2015). Estimating counterfactual baselines for avoided loss offsets is complicated as this involves evaluations about what would happen to these sites without the offset, evaluations which have shown to often be overestimated (Zu Ermgassen *et al*, 2019; Maron *et al*, 2015). Also, avoided loss offsets will always lead to a NL of habitat area, further hindering the probability of these offsets to reach NNL (Zu Ermgassen *et al*, 2019). For these reasons, it would be advisable to avoid avoided loss offsets when possible.

The use of multipliers was recommended to compensate for the many uncertainties in offsetting (Ghijselinck *et al*, 2026; Moilanen & Kotiaho, 2018a; Souza *et al*, 2023). Offset monitoring is currently poorly understood, though this also depends on the monitoring action, where some are more studied than others (Moilanen *et al*, 2024). One way forward could be to add additional multipliers when the predicted outcomes of monitoring actions are unknown. However, while multipliers are intended to compensate for the uncertainties of offsetting, they are based on offsetting metrics - such as habitat area or biodiversity units of aggregated multi-metrics (e.g. the SBM and CLIMB) - that have well-documented limitations, which in turn may make the multipliers themselves uncertain. Therefore, I suggest addressing uncertainties directly, and use multipliers only when direct approaches are not possible.

4.3 Species-based vs habitat-based metrics

From the publications included in literature study 2, two weaknesses regarding habitat-based offsetting metrics stood out. First, habitat-based metrics did not capture species diversity, including invertebrates, butterflies and birds (Marshall et al, 2024; Duffus et al, 2024). Many species are dependent on complex habitat features, such as heterogeneity, which may not be sufficiently represented in simple habitat-based metrics (Duffus *et al*, 2024; Marshall *et al*, 2024; Marshall *et al*, 2021). Second, habitat-based metrics in offsetting commonly have a heavy focus on area, which can lead to other aspects of biodiversity being overlooked (Zu Ermgassen *et al*, 2019). When used in isolation area tends to favor high offset area ratios, but within aggregated metrics it can instead enable the replacement of large habitats with smaller, higher-scoring patches (Zu Ermgassen *et al*, 2019; Zu Ermgassen *et al*, 2021). The latter explained the discrepancy between the claimed 20% NG and a 34% loss in greenspace found in England (Zu Ermgassen *et al*, 2021). Metric aggregation has previously been identified as problematic, because it enables the trade of biodiversity indicators, potentially hiding the different ecological relevance of individual elements (Maseyk *et al*, 2016).

Habitat-based metrics could be refined by adding species data, which currently is scarcely used in offsetting (Duffus *et al*, 2025b; Marshall *et al*, 2020; Marshall *et al*, 2022). However, generating species data is an expensive and time-consuming task, and consequently its use has to be clearly motivated (Moilanen & Kotiaho, 2018b). It has been suggested to align the inclusion of species-based data with conservation goals (Fletcher *et al*, 2025). Here, the goals of the offset could guide practitioners in determining when species data is necessary, which in turn increase the demands for clearly set goals.

Beyond adding species data, it was suggested to add additional habitat-based metrics such as habitat heterogeneity, as well as landscape features like structural connectivity (Cousins, 2016; Duffus *et al*, 2025b; Duffus *et al*, 2025b). Habitat heterogeneity has previously shown promise as a biodiversity surrogate in boreal forests (Hekkala *et al*, 2023; Johansson *et al*, 2025). Incorporating a heterogeneity score into existing offsetting metrics could improve the metrics by accounting for species that depend on complex habitat features (Duffus *et al*, 2025b).

Given that habitat-based metrics are relatively easy to apply, understanding in which situations they can reliably act as surrogates for species-based metrics could improve the efficiency of offset monitoring without losing credibility. There have been studies comparing species-based vs habitat-based surrogates, showing that habitat-based surrogates outperformed species-based as proxies for species richness of arboreal marsupials and species of conservation concern in boreal forests (Lindenmayer *et al*, 2014; Johansson *et al*, 2025). However, to my knowledge, no review synthesizing these type of studies has been conducted. This could be a natural next step to further understand the dynamics of biodiversity metrics and surrogates.

4.4 How and when - Limited results

How offset monitoring should be designed in terms of time and scale was rarely discussed in the literature. As an example, no study talked about how frequently offset sites should be monitored. One explanation for this could be that I in this study only looked at publications about offsetting specifically, meaning publications about conservation was excluded.

However, some publications discussed monitoring duration time. Today, monitoring duration times vary between policies. For example, the State of Victoria applies mandatory monitoring for 10 years, while England applies 30

years. As maturity times are usually long - sometimes longer than 30 years - long-term monitoring was recommended to ensure NNL/NG (Cousins, 2016; Marshall *et al*, 2022; Moilanen *et al*, 2024; Souza *et al*, 2023). If the duration time is too short, time-lags can be hidden, increasing the risk that biodiversity gains are overestimated (Marshall *et al*, 2022; Moilanen *et al*, 2024). Since the concept of offsetting encompasses that habitats are exploited, meaning biodiversity to some extent is forever lost, one could argue that NNL/NG can only truly be claimed if offset sites are monitored forever. However, permanence of offset sites is also an economical question (Souza *et al*, 2023). Therefore, some sort of trade-off between what is desirable is theory and what is possible in practice is necessary. One suggestion was to add multipliers if the estimated maturity time is longer than the possible monitoring period (Moilanen & Kotiaho, 2018b). However, relying on multipliers determined at the planning stage could add further uncertainties as the outcome of the offset would be unknown. Another suggestion to compensate for long time-lags was to use habitat banking (Rampling *et al*, 2023). One clear advantage of a banking system is that the compensation is implemented before the impact occurs, thereby avoiding both the temporal loss and the uncertainty surrounding ecological outcomes associated with traditional offsetting practices (Theis *et al*, 2022). Moreover, because banking enables several small transactions to be merged into a single larger unit, it can protect more extensive and connected habitats, thus avoiding the creation of small, fragmented offset patches that often have lower ecological resilience (Theis *et al*, 2022). However, habitat banking can also be viewed as a type of averted loss, which often has a hard time demonstrating NNL (Cousins, 2016).
For these reasons, long-term monitoring is still essential (preferably extending beyond 30 years), with multipliers used as a complementary measure.

A few publications examined how monitoring should be designed. They identified two key problems with monitoring design in offsetting as it stands today: the lack of control sites, which made it difficult to establish a counterfactual baseline, and the rare estimation of biodiversity losses, hindering accurate estimation of biodiversity gains (Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024). To address these issues, using a BACI-model was recommended for ideal statistical power (Josefsson *et al*, 2021; Moilanen *et al*, 2024). This is in line with precious research showing that simple study designs can lead to misleading results (Christie *et al*, 2019). However, it was also noted that applying a BACI design in offsetting can be challenging, as it requires multiple replicates, which might not be viable when sites are monitored for only a few years (Moilanen *et al*, 2024). This means that for BACI-designs to be meaningful they need to be coupled with long-term monitoring. Using BACI-designs is highly preferable, but when resources are insufficient for a full BACI-design, an option is to use a BA-design. BA-designs do not include control sites, weakening the data significantly. But they include quantifying biodiversity loss, making comparison between loss and gain possible, which is fundamental for estimating outcomes at all.

4.5 Public databases for knowledge sharing and increased transparency

Establishing public databases was recommended to bridge the current gap in offsetting between what offsets claim to achieve on paper, and what they deliver in practice (Gordon *et al*, 2025; Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024). Such databases could enhance the knowledge about monitoring actions (Josefsson *et al*, 2021; Kujala *et al*, 2022; Moilanen *et al*, 2024), and could also offer a practical toolbox for implementation of adaptive management. Furthermore, by publicly documenting offset sites they can be protected from future exploitation (Rampling *et al*, 2024). Since the databases would be available to the public, they could also increase the transparency of offsets (Kujala *et al*, 2022). One could argue that transparency in the light of the ethic dimensions of offsetting becomes particularly important. Offsetting treats nature as replaceable units, which from a communication perspective is a very clear tool - 2 is more than 1 and so forth. However, as shown many times previously, as well as here, demonstrating NNL with complete certainty is maybe an impossible task. Therefore, transparent communication both about the uncertainty of the used metrics (which are always proxies) and about other uncertainties, such as counterfactual baselines, time lags, what has not been included in the measurements e.g., is important to make offsetting more credible as a whole.

4.6 Offset monitoring in Sweden

Since the goal of offsetting is often NNL, and in some cases even 10% NG, as in England and Uppsala municipality, the monitoring methods used in offsetting need to have high statistical power to detect small changes. Methods that can detect small changes have also previously been described as key for robust monitoring design (Lindenmayer *et al*, 2020). This could complicate using CLIMB as a method for offset monitoring in Sweden. This is because the conservation value in CLIMB is derived from evaluations of 'conservation value classes', which have only 7 categories (class 1-7), making the steps between categories big and vague.

Despite that CLIMB in its current form might not be suitable to use in offset monitoring, the method is based on the English BNG system and it is reasonable to think that CLIMB will be used in Sweden in the future when quantifying offset losses and gains. Therefore, even if CLIMB would not be used in offset monitoring, it would be preferable if monitoring actions would lead to an increase in CLIMB units in the long term. In this context, it could be important to refine the habitat-based features of CLIMB in a similar way that was suggested for the SBM. This includes adding a habitat heterogeneity score, and to account for structural complexity more than an aspect within the 'landscape value factor', that currently leads to a general 15% increase in CLIMB units. Furthermore, it would be of high weight to evaluate the 'conservation value classes' within the Swedish Standard. The classifications are primarily based on field visits with little quantitative data. This creates a risk of subjectivity that may be hidden when translated into numerical values in a quantitative method such as CLIMB. Subjective measurements can be problematic as they may lead to inconsistent results (Zu Ermgassen *et al*, 2021). According to Uppsala municipality's offsetting guidelines, class 1 sites should be excluded from offsetting under the avoidance step of the mitigation hierarchy (Uppsala kommun, 2024). Consequently, inconsistencies in conservation class classification may have practical implications, where class 1 sites risks being exploited.

Based on my findings, I recommend the following for offset monitoring in Swedish municipalities:

1: Set clear offsetting goals, and plan the monitoring actions according to these. There can be goals at different ecological scales, for example ‘increase the diversity of species group X’ and ‘increase the population size of species X’.

2: Follow the mitigation hierarchy carefully, applying avoidance to the greatest extent possible, while using offsetting only as a last resort when the previous steps in the hierarchy are not feasible.

3: Choose biodiversity metrics according to the set goals, including species-based metrics if the goals deem them necessary.

4: If CLIMB is used in the planning stage, it should also be used in offset evaluation to ensure comparability between losses and gains. Repeated CLIMB evaluations should therefore be conducted alongside selected monitoring metrics (but with longer time intervals) to ensure that improvements in the monitored features are reflected in long-term increases in CLIMB units. If CLIMB and the selected monitoring metrics do not align over time, adaptive management should be applied.

5: Use additional multipliers when the outcomes of the selected offsetting actions are poorly studied, and when the estimated maturity time of the site is longer than the possible monitoring period.

6: Adapt the monitoring time according to the estimated maturity time of the site. 30 years minimum would be desirable, but to declare 10% NG monitoring should go on for as long as possible (ideally forever).

7: Establish public databases to enable knowledge sharing about which monitoring actions have proven successful and which have not, and to increase transparency by communicating uncertainties of biodiversity metrics, counterfactuals baselines etc.

8: Avoid avoided loss offsets when possible.

9: Use BACI-designs whenever possible. If a full BACI-design with control sites is not feasible, at least use a BA-design, so that gains can be compared to losses.

Conclusion

Here, I have shown that offset monitoring as currently applied has several limitations. I have also highlighted the importance of an integrated approach, where monitoring actions are closely linked to offset goals and planning-stage decisions. Based on my results, I have provided recommendations for the design of offset monitoring in Swedish municipalities. It is important to recognize that implementing all of these recommendations in practice may not always be possible, given the financial and practical constraints faced by municipalities. However, as noted by Moilanen *et al* (2024), poor monitoring design can not only be a waste of resources, but also risks creating an illusion of offset success. This could hinder learning for the future, and ultimately contribute to further biodiversity loss. Therefore, incentives for robust monitoring are needed to encourage municipalities to prioritize monitoring efforts, thereby increasing the likelihood of offset success.

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5. Populärvetenskaplig sammanfattning

Den biologiska mångfalden minskar i oroande hög takt, och en stark bidragande faktor är ändrad markanvändning till följd av mänsklig aktivitet. Ekologisk kompensation är ett allt vanligare verktyg, där förluster av naturvärden på en plats kompenseras genom att skapa vinster på en annan plats. Ekologisk kompensation är samtidigt ett omdebatterat verktyg, delvis eftersom det bygger på antagandet att förluster idag kan kompenseras med vinster imorgon, vilket medför många osäkerheter. Trots att ekologisk kompensation används i allt högre utsräckning visar flera utvärderingar att många kompensationsprojekt inte uppnår sina mål. En aspekt som återkommande lyfts i forskningen är vikten av väl utformad uppföljning för att kunna avgöra om kompensationsåtgärderna faktiskt fungerar.

I Sverige kan kommuner applicera frivillig ekologisk kompensation inom Plan- och Bygglagen. I Uppsala kommun antogs år 2024 riktlinjer för ekologisk kompensation med målet om 10 procent nettovinst av biologisk mångfald. Syftet med detta examensarbete var att utvärdera hur svenska kommuner kan designa uppföljning av ekologisk kompensation, genom att använda Uppsala kommun som ett exempel. För att utvärdera detta utfördes två litteraturstudier. Den första fokuserade på hur uppföljning utförs inom etablerade kompensationssystem i England, USA, Frankrike och Australien. Den andra sammanställde vetenskaplig litteratur om utmaningar och rekommendationer kopplade till uppföljning av ekologisk kompensation.

Mina resultat bekräftar tidigare observationer om att uppföljningen ofta är bristfällig, även i länder med väletablerade kompensationssystem. Ett återkommande problem är att det ofta saknas tillräckliga data för att avgöra om målen om ingen nettoförlust eller nettovinst för biologisk mångfald faktiskt uppnås. Ett viktigt fynd var också att aspekter som är en del av planeringsfasen, till exempel vilka biodiversitetsmått som ska användas för att beräkna vinster och förluster, i hög grad påverkar möjligheten att sedan utvärdera resultaten på ett korrekt sätt. Resultaten visar vidare att det också är viktigt hur uppföljningen genomförs och hur länge den pågår, eftersom många naturvärden utvecklas långsamt och förändringar kan därför vara svåra att upptäcka på kort sikt.

Utifrån mina resultat föreslås bland annat långsiktig uppföljning (minst 30 år), stark studiedesign där bakgrundseffekter minimeras, och att uppföljningsåtgärder och biodiversitetsmått är nära kopplade till tydligt satta kompensationsmål. Dessutom rekommenderas inrättandet av offentligt tillgängliga databaser där kompensationsprojekt dokumenteras. Dessa kan på sikt bidra till ökad kunskap om uppföljningsåtgärder, men kan även ha ett viktigt syfte i att öka transparansen kring ekologisk kompensation och dess osäkerheter.

Appendix

Supplementary table 1:

Adapted from EcoGain, 2023. Table summarizing the equations used in CLIMB. Abbrivations: A = Area in hectares; NV = Factor for 'conservation value'; L = Factor for landscape value; S = Factor for difficulty level; T = Factor for time; I = Factor for indirect impact; t0 = before impact/action; t1 = after impact/action; Distance = Factor for distance between the impact area and the compensation area.

Equation	***Explanation***
At0 · NVt0 · Lt0	Net CLIMB units
At1 · NVt1 · Lt1 · S · T	CLIMB units created through habitat creation
((At1 · NVt1 · Lt1) − (At0 · NVt0 · Lt0)) · S · T	CLIMB units created through habitat improvement
(At1 · NVt1 · Lt1) − (At0 · NVt0 · Lt0)	CLIMB units lost through habitat deterioration
(AI · NVt0 · Lt0) · (1 − I)	CLIMB units lost through indirect impact
On-site CLIMB units created by habitat creation + on-site CLIMB units created by habitat improvement + on-site CLIMB units retained − CLIMB units lost through habitat deterioration − CLIMB units lost through indirect effects − CLIMB units lost	On-site change
(Off-site CLIMB units created by habitat creation + off-site CLIMB units created by habitat improvement + off-site remaining CLIMB units that lead to additionality − CLIMB units lost through habitat deterioration − CLIMB units lost through indirect effects − CLIMB units lost) × Distance	Off-site change

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