UiT The Arctic University of Norway

Department of Arctic and Marine Biology

# Developing a mitigation hierarchy framework to conserve wetland biodiversity under pressure from development

Mathias Leines Dahle

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Cover page: Peatland plant diversity, Northern Norway. Photo by the author.

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**Supervisors:**

Vera Helene Hausner, UiT – The Arctic University of Tromsø

Francisco Javier Ancin Murguzur, UiT – The Arctic University of Tromsø

Anders Lyngstad, NINA – Norwegian Institute for Nature Research

# Forewords

This thesis is dedicated to nothing less than our home - Planet Earth. By many even referred to as *Mother Nature* – and for good reasons. After five years of biology studies, I have learnt that your provision of food, shelter, clean water, and fresh air is not an act of purpose nor kindness. It is a result of evolution and complex interactions among the millions of species that call you *home*. Maybe one would think that such an insight would remove the mystic and overwhelming admiration for the natural world, as experienced by a 3-year-old boy sitting in the corner of a playground, digging for insects. But it has not. The little boy has grown bigger, and so has my fascination for nature. I know now that we, *Homo Sapiens*, are “just another species in the forest” – and I love it. We belong to the phylum Chordata, the class Mammalia, and the family of Great Apes. And we are one among an estimated 10 million species. Unfortunately, I have also discovered that that one species is having a massive negative impact on the natural world. This has been the backdrop and motivation for conducting this study. I believe that we can change the trend, and I hope this thesis in some way can be a small contribution to the conservation of nature – our home.

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# Acknowledgement

Thank you so much to my supervisors. I want to thank you, Vera, for meeting me with enthusiasm for sustainability from the first day I knocked on your door, for guiding me towards the mitigation hierarchy framework (in which my thesis is based on), and for supporting and trusting me throughout the process. I want to thank you, Anders, for being my wetland expert and provide important insights and perspectives on both wetland ecology and master thesis workflow. Your feedback has always been clear and helpful, and I have felt safe having you as a supervisor. Last, but definitely not least – I want to thank you, Xabi, for being there all the time. I have never experienced a more open door than the one to your office. You have been a problem-solver, motivator, lunch-partner and psychologist. You are a multi-talent with a big heart. I hope you see the value of your support, and what you mean for those around you.

With the overwhelming task of “saving nature”, I am ever thankful to have been introduced to The Arctic Sustainability Lab. Here, I have experienced to be part of a team that not only sees and documents global challenges – but also tries to find solutions. I want to send a huge thanks to the whole group, for numerous Friday meetings, complex talks about how to save the world, and for great support throughout the two years of my master’s degree.

Doing a thesis is not an easy task. I have been lucky enough to be in the process with my soulmate and girlfriend, Kjerstin. Thank you for making my days at the University meaningful, a little less frustrating, and fun (as far as possible). You have ensured that I have taken breaks in-between my intense focus periods (we need that as well). And you make me human (in a good way). I admire you for keeping up with my boring, silent way of working for hours at a time, for not liking being interrupted, and for appreciating who I am. And of course for sharing my fascination and love for nature.

In the end I want to thank my family. Not because this thesis will change the world and your names should be here for credit, but because you were the ones introducing me to nature. You showed me how to dig for insects and you let me have a frog-hatchery in my room. Unfortunately for you, you now need to deal with my strict sustainable-living requirements - so maybe you regret? Anyways, I want you to know that I am thankful.

# Abstract

Wetlands are severely affected by human development. About 50 % of their original global extent has been lost, their populations of plants and animals have declined faster than for any other ecosystem, and 25 % of wetland-dependent species are threatened. As the main reason for these declines is habitat loss, often caused by infrastructure development, it is critical to develop conservation strategies targeting this particular pressure. The *mitigation hierarchy* (MH) provides for a promising solution, outlining how *no net loss* (NNL) of biodiversity can be achieved through the four steps avoid, minimize, restore, and offset impact losses. Although increasingly applied worldwide, the MH suffers from poor implementation and a lack of standardized methods, impeding practical application and successful conservation. In this study, I develop a landscape-scale MH methodology from scientific best-practices and demonstrate how it can be used to improve impact mitigation efforts in wetlands. By planning on a landscape scale, managers can consider how, and to what degree, many small development impacts together exert a cumulative pressure on natural environments. In the outlined approach, I calculate area-specific wetland conservation values from easily available data on species distributions, species threat level and ecosystem condition, and show how simple spatial analysis can be used to map areas of special importance for avoidance and offsetting. I also show how the same values can be used to determine offset sizes large enough to compensate for residual biodiversity losses. The landscape-scale mitigation planning approach presented here can provide for 1) a rapid assessment of wetland conservation value, 2) early anticipation of potential biodiversity impacts, 3) avoidance of valuable wetlands; reducing offset costs, 4) identification of potentially degraded wetlands for restoration offsets, and 5) a visualization of how NNL of biodiversity can be achieved.

**Keywords**: *mitigation hierarchy, conservation biology, no net loss, biodiversity, wetlands, habitat loss, landscape-level planning, development impacts*

# 1 Introduction

Wetlands cover only ~1 % of Earth's surface (Lehner and Döll 2004; Kingsford et al., 2016), but provide a disproportionately large amount of global biodiversity and ecosystem services (Dudgeon et al., 2006; Costanza et al. 2014; Kingsford et al., 2016). In addition to serving as critical habitats for the many species strictly dependent on them, wetlands are an important source of freshwater, food, and nursery-grounds for others (Kingsford et al., 2016). It is especially the complex mosaic of microhabitats with varying levels of inundation that gives rise to high levels of biodiversity in wetlands (Ward et al., 1999), and up to 40 % of the world's species is estimated to live and breed in this habitat (Mitra et al., 2003; Ramsar Convention Secretariat, 2018). Wetlands are also among the most productive ecosystems in the world (Mitsch and Gosselink, 1993; Ramsar Convention on Wetlands, 2018) and are vital to human livelihoods (Kingsford et al., 2016), providing us with a higher degree of ecosystem services than any other environment (Zedler and Kercher 2005; Russi et al. 2013; Costanza et al. 2014; Kingsford et al., 2016). Among the most prevalent of these is their ability to sequester and store enormous amounts of carbon (Kayranli et al., 2010), regulate water flow (Costanza et al., 2014; Kingsford et al., 2016) and purify water - providing clean drinking water for millions of people (Ramsar Convention on Wetlands, 2018). For these reasons, both IPBES (2017) and the Convention on Wetlands (2021) emphasize that wetlands are critical to the success of the United Nations Sustainable Development Goals.

Despite the critical role wetlands play for biodiversity and ecosystem services; wetlands have suffered tremendous losses, with estimates ranging from 33% to 87% since the 18$^{th}$ century (Davidson, 2014; Hu et al. 2017). Moreover, the rate of loss has been accelerating (Davidson, 2014) and between 1970 and 2015 the loss rate was found to be three times higher than for forests (Dixon et al. 2016; FAO, 2016). The Ramsar Convention (2018) also underscores that the *quality* of the remaining wetlands is decreasing.

The deteriorating state of global wetlands has been accompanied by a dramatic decline of wetland-dependent species (Davidson, 2018). By analyzing data from the IUCN Red List, the Ramsar Convention on Wetlands (2018) has calculated that 25 % of inland wetland-dependent species are globally threatened, of which 6 % are defined as critically endangered. 81 % of their total populations have decreased (Ramsar Convention on Wetlands, 2018), and the rate of decline the last 50 years has been substantially higher than for species living in other habitats (Millennium Ecosystem Assessment, 2005; Davidson, 2018; Convention on

Wetlands, 2021). As current conservation measures appear to be insufficient to halt biodiversity loss, there is an urgent need for improvements (Arlidge et al., 2018). In this context, there has recently been a call for a global framework that can provide a roadmap for conservation of nature (Watson and Venter 2017; Arlidge et al., 2018).

One of the most promising approaches for biodiversity conservation is known as “the mitigation hierarchy” (MH) (Kiesecker et al., 2010; Arlidge et al., 2018; Jones et al. 2022). It is a framework providing a complete step-by-step approach to reach no net loss (NNL) of biodiversity and has been increasingly applied by nations and corporate managers worldwide (Rainey et al., 2015; Maron et al., 2016; de Silva, 2019; Heiner et al., 2019). The MH is aimed primarily at curbing impacts resulting from infrastructure development (Phalan et al., 2018), although it can be used for a wider range of purposes (Arlidge et al., 2018). This makes the MH highly appropriate for addressing wetland biodiversity declines, whose main driver is habitat loss (Kingsford et al., 2016) primarily caused by infrastructure development and agriculture (MEA, 2005; IPBES, 2019; Ballut-Dajud et al., 2022).

To achieve NNL of biodiversity, the MH takes a highly pluralistic approach. It combines economic incentivization with legal protection and couples a variety of other aspects taken from the fields of conservation science, biology, strategic planning, law, and economics. The framework has also adopted parts from environmental and social impact assessment procedures (Pope et al., 2013; Arlidge et al., 2018; Bigard et al., 2020), although it goes beyond the scope of such assessments by extensively considering project *alternatives* – thereby resembling strategic planning practices (Phalan et al., 2018; Heiner et al., 2019; Bigard et al. 2020).

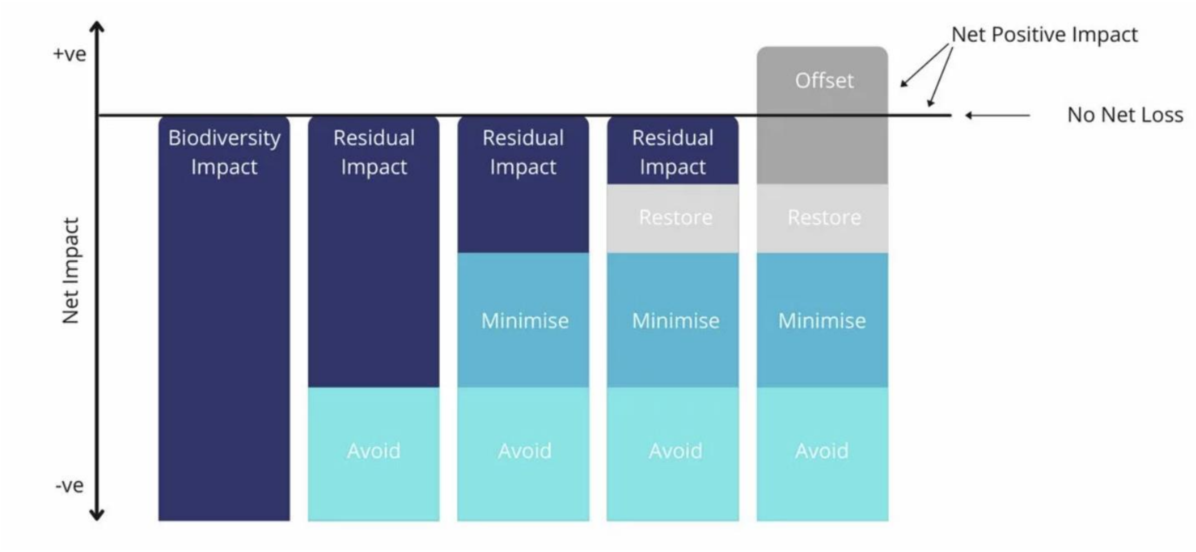


***Figure 1****. A visualization of how the mitigation hierarchy steps 1) avoid, 2) minimize, 3) restore and 4) offset act to reduce biodiversity impacts, eventually reaching No Net Loss or even a Net Positive impact. (Figure from The Biodiversity Consultancy, 2023)*

The core of the MH is four hierarchical steps to mitigate development impacts (Figure 1): 1) *avoid* impacts, 2) *minimize* unavoidable impacts, 3) *restore* as much as possible of the on-site impacts once construction work is done, and 4) *offset* all remaining impacts (McKenney and Kiesecker, 2010; Gardner et al, 2013). While most of these steps are self-explanatory, what offset means is less clear. It is defined by The Business and Biodiversity Offsets Program (2018) as “measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken”. Because impacts from developments cause a *loss* of biodiversity, the offset must generate a *gain* big enough to compensate for this loss. As shown in Figure 1, offsets can even contribute to a *net gain* of biodiversity. In practice, offsets typically take the form of either restoring a degraded ecosystem or legally protecting an intact ecosystem that would otherwise risk exploitation (so-called “averted loss” offsets) (Ekstrom et al. 2015; Jones et al. 2022). The task of realizing the offsets lay at the part responsible for the loss, who are often allowed to choose between conducting the offset themselves or pay a development fee to the state or a third-party company who take over this responsibility (Vaissière and Levrel, 2015). By allowing offsets through a development fee, the MH can generate substantial amounts of money to conservation (Kiesecker et al., 2009).

As its name implies, the MH steps are meant to be implemented hierarchically. This means that one must try to avoid impacts altogether (step 1) before considering specific minimization alternatives (step 2) and so on (Ekstrom et al., 2015). The hierarchic structure reflects the level of importance of each step; avoidance is considered the most effective way of reducing impacts to biodiversity (McKenney and Kiesecker, 2010; Ekstrom et al., 2015 Sonter et al., 2020), whereas offsets are seen as the least effective and should therefore only be used to compensate for *residual* losses after efforts have been made to avoid, minimize, and restore damage on-site (McKenney and Kiesecker, 2010; BBOP, 2018).

A major strength of the MH is the acknowledgement that avoidance of wetland loss cannot be accomplished in all cases. As such, alternative mitigating measures are outlined to ensure that overall biodiversity will not decline on a larger scale. This provides for a more effective approach to biodiversity conservation than legal protection alone, because managers tend to make exceptions to their laws of absolute avoidance without having formal alternative measures in place to compensate for the loss. For example, Norwegian authorities have recently decided to open for establishment of a highway through a river delta nature reserve, by altering its statutory regulation (The Norwegian Government, 2023). And although compensatory measures are discussed, there is no policy framework ensuring achievement of NNL. As such, the MH's in-built allowance for the "sacrifice" of some wetlands, under strict regulations, for the improvement of others, may provide a better overall outcome for wetland biodiversity.

Despite a solid theoretical foundation and increased application in different parts of the world, the MH faces many practical challenges (Heiner et al., 2019; Larsen et al., 2018) and failures are widely reported (Maron et al., 2012; Lindenmayer et al., 2017; Sonter et al., 2020). For example, avoidance - the first and most important step of the hierarchy, is often neglected (Clare et al., 2011; Villarroya et al., 2014), and there is a lack of formal methods (Quetier and Lavorel, 2011), causing a large variation among mitigation policies (Bennett et al., 2017). In turn, this has made it difficult to evaluate what factors determine conservation success (Sonter et al., 2020), ultimately making methodological advances difficult. Bartoldus found already in 1999 more than 100 methods to assess wetland value, but few have been shown to be applied in practice due to high cost and complexity (Kusler, 2006). For example, The Environmental Law Institute (2002) found that 60 % of 200 wetland mitigation banks (i.e., a third-party variant of offset realization) in the United States, determined offset size and development fees solely based on impact *size*, meaning that differences in wetland value was not considered at all. Although this challenge has received more attention by researchers the last decades, methodological advances are still needed before large scale implementation can be expected.

Lately, the need for upscaling the application of the mitigation hierarchy from a commonly used project-to-project basis to a broader landscape scale has also been emphasized (Kiesecker et al., 2010; Bigard et al., 2020; Jones et al., 2022). Such landscape scale assessments should not substitute, but rather complement project level mitigation efforts due to its ability to improve impact avoidance and offset localization (Arlidge et al., 2018; Bigard et al., 2020). In this context, it is important to recognize just how the two approaches differ; while the project level MH eventually seeks to fulfill NNL by replacing what is lost on the impact site on the offset site (e.g., species, habitats and ecological functions), landscape scale mitigation efforts primarily aims to facilitate this achievement for multiple projects by anticipating where negative impacts can be reduced to a minimum and where compensating efforts should be focused (Bigard et al., 2020). By doing so, the required compensatory actions needed to reach NNL of biodiversity on the project-level will be minimized.

Although several studies outline methods for landscape scale mitigation planning (e.g., Kiesecker et al., 2010; Bigard et al., 2020; Jones et al., 2022), most of them only consider a few MH principles - often seemed to be based on random choices depending on the authors own preferences, rather than systematic inquiry. In addition, the level of ecological resolution included in such assessments is generally small, with often less than eight different ecotypes distinguished (see e.g., Bigard et al. 2020; Jones et al. 2022). Due to large quality differences also *within* ecosystems, more detailed ecosystem evaluations are needed to ensure that impacts to high-quality habitats are prioritized for avoidance, and that such impacts cannot be easily offset in low-quality zones. Although useful for all ecosystems, such practices are critically needed for the most important and vulnerable ones. Wetlands, being recognized as the most valuable ecosystem to humans (Russi et al., 2013; Costanza et al., 2014) and the one declining at the fastest rate (Dixon et al., 2016; FAO, 2016), is thus a natural starting place.

This study describes the development and application of a landscape-level MH approach to wetland conservation. My aim is to address the abovementioned MH-challenges through the following objectives: 1) identify and map key mitigation hierarchy principles in use through a literature review, 2) use the identified principles to develop an easily applicable, landscape-level mitigation methodology for protecting wetlands threatened by development, targeted to decision-makers, and 3) demonstrate the methodology’s practical application through a case study.

In the case study I assess how potential wetland impacts of wind power development can be mitigated in the northernmost county of Norway. While Norwegian authorities are currently on the look for wind power opportunities to address climate change, such developments are pointed to as one of the biggest threats to wetlands in the country (Lyngstad et al., 2018). I address this challenge by building on recent research and best-practices to show how general mitigation planning principles can be tailored to wetland ecosystems and balance the need for both development and wetland conservation. More specifically, I analyze spatial data through Geographic Information System tools and highlight a path for achieving NNL of wetland biodiversity, focusing primarily on mapping priority avoidance areas and potential offset sites. The structure of the thesis is twofold; first, I describe the procedure and results of the literature review, before I explain the development and application of my MH methodology through the case study.

# 2 Identification of mitigation hierarchy principles

I conducted a literature review to identify commonly applied principles within each of the four MH steps (i.e., avoid, minimize, restore, offset). A *principle* was defined as a general aspect, method or recommendation proposed by researchers studying the mitigation hierarchy for nature conservation. The purpose of the review was to collect information on how the MH has been applied to conservation in general, and in relation to wetlands more specifically, which could serve as a scientific basis for developing a more specific wetland mitigation methodology. However, because of few search results for wetland-specific MH publications, the review was ultimately guided by a search for general MH principles relevant to all ecosystems. The exception was for restoration, for which a sufficient number of wetland related publications were found.

After searching the databases of Google Scholar, Web of Science, and the Biological Science Collection (see Appendix A for complete review procedure), I identified 65 different MH principles in total (Appendix A, Table S1 – S5). These were organized into their respective MH step according to how they were classified in the reference document, and thereafter grouped into larger categories referred to as “main principles”.

For impact *avoidance,* I found 13 specific principles which may be summarized by the five main principles “assess potential impacts”, “avoid priority avoidance areas”, “select avoidance strategy (four listed alternatives)”, “plan on both a large and small scale”, and “start avoidance consideration as early as possible” (Appendix A, Table S2). I also found some general criteria to move past the avoidance step, including full consideration of project alternatives, that social benefits outweigh environmental costs, delivery of no net loss of biodiversity, and fulfillment of national laws (Phalan et al., 2018).

For *minimization*, I identified 8 principles which I divided into the groups “implement formal development requirements”, “consult experts”, “minimize development area”, “minimize impact on development area” and “consider project alternatives” (Appendix A, Table S3).

For wetland *restoration*, I found 11 specific principles and arranged them into the broader groups “assess restoration needs and feasibility”, “set objectives and decide on measures”, “act before damage has occurred” and “use well-tested restoration techniques” (Appendix A, Table S4).

For *offsetting*, I identified 27 principles, mirroring the great abundance of studies addressing this particular MH step. The principles were summarized as “only use as last resort”, “develop offset goals”, “select offset site based on impact site characteristics”, “ensure additionality”, “avoid time-lag”, “determine offset size requirements”, “consider negotiation time”, “maximize offset longevity”, “ensure environmental justice”, “base methods on existing knowledge”, “monitor conservation success”, “plan on a landscape scale”, “select offset type” and “decide on offset implementation strategy” (Appendix A, Table S5).

# 3 Case study: development and demonstration of a MH methodology

## 3.1 Case study description

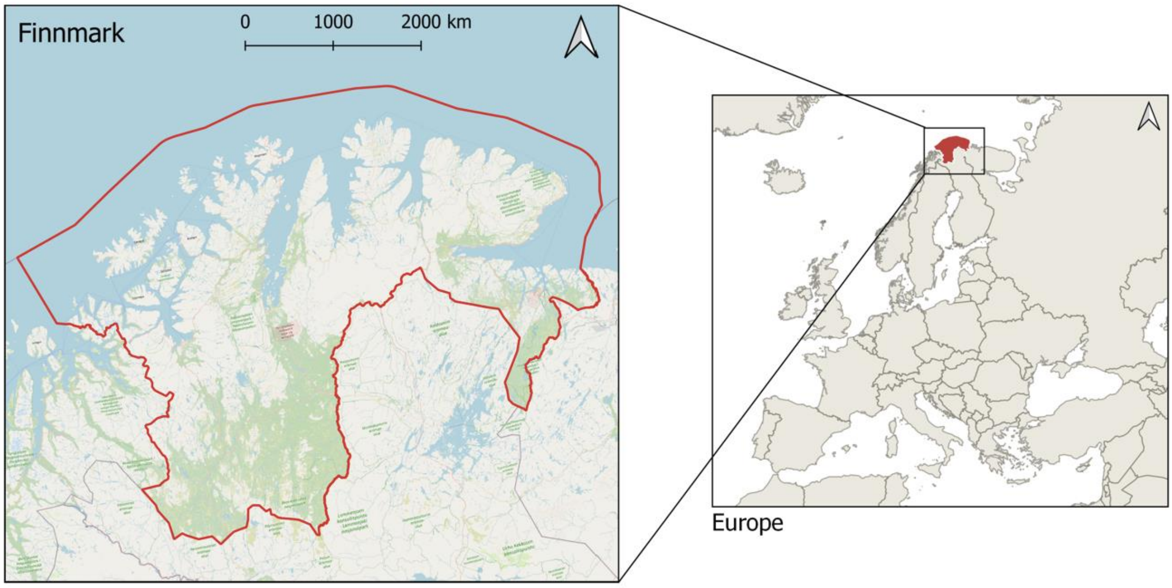


***Figure 2.*** *The study area of Finnmark, Northern Norway.*

The study area was limited to Finnmark, Northern Norway (Figure 2). Although currently a part of the larger Troms and Finnmark county, this region has historically been, and will from year 2024 be its own county. Such a jurisdictional unit was considered relevant for the study purposes because Norwegian wetland conservation policies are followed up by county-specific protection plans (Strann & Nilsen, 1996; Moen & Daverdin, 2010). Finnmark has great potential for wind power development because of its flat topography, high annual wind speed and vast uninhabited areas (although still used for reindeer herding). At the same time, wind power development is pointed to as one of the largest threats to wetlands (Lyngstad et al., 2018), largely due to the lack of regulations ensuring proper wetland protection (Norwegian Environmental Agency, 2022a). Current national wetland conservation strategies are mainly focused on three isolated measures: 1) laws prohibiting drainage of wetlands for agriculture, forestry, and peat extraction, 2) protected areas around a selection of especially valuable wetlands, and 3) restoration of a limited number of degraded wetlands, mostly within protected areas (NEA, 2020; Ministry of Climate and Environment, 2021; NEA, 2022a). Although highly valuable for reducing the *rate* of wetland loss, these efforts are not adequate to ensure NNL, as wetland impacts continue to exceed wetland restoration (NEA, 2022a). NNL could rather be achieved through a systematic application of the mitigation hierarchy, ensuring mitigation of all new harmful wetland impacts. The framework's potential assets for national wetland conservation have also been acknowledged by the Norwegian Environmental Agency (2022a).

Finnmark is largely placed within the forest tundra ecotone between the boreal forest and the arctic tundra biome, and the area is characterized by large, low-density forests of mountain birch (*Betula pubescens subsp. Czerepanovii),* as well as extensive wetlands. The wetlands cover huge areas due to multiple water systems, low temperatures causing little evaporation, and a flat topography reducing water flow. Early studies estimate that wetlands cover up to 30 % of the inner part of Finnmark (Strann & Nilsen, 1996), substantially more than for most other parts of Norway (NEA, 2020). Although there is a large variety of wetlands in Finnmark (Fylkesmannen i Finnmark, 2010), they generally differ from those elsewhere in the country due to their location in the North combined with a continental climate found in the inner parts (Strann & Nilsen, 1996; FMFI, 2010). This gives rise to characteristic "Eastern" vegetation (i.e., plants typically found further East) and unique wetland types, such as the endangered palsas with a core of permafrost (Øien et al., 2018). The wetlands also provide for Norway's largest and most important habitat and breeding ground for numerous bird species, many of which are threatened by extinction and found nowhere else in the country (Strann & Nilsen, 1996; FMFI, 2010).

## 3.2 Development and demonstration of a mitigation hierarchy methodology

### 3.2.1 Selecting mitigation hierarchy principles

The aim of my case study was to assess how the MH could be applied on a landscape-scale to conserve wetlands facing wind power development. As my literature review revealed a highly context-specific nature of minimization and restoration efforts (step 2 and 3 of the MH), impeding development of a standardized methodological approach, I concentrated on the avoidance and offset steps.

For avoidance and offset analysis, it was neither achievable nor necessary to incorporate all identified MH principles due to study constraints and that many principles can only be implemented by local decision-makers. For my purpose, I excluded principles that 1) required change in regulatory frameworks by politicians and lawmakers, 2) were primarily aimed at social or economic aspects of the MH, 3) were not possible to address within the scope of a

MSc thesis, 4) required unavailable data, and 5) were not relevant for spatial landscape-scale analysis.

For offsetting, I made the methodological choice of concentrating on mapping potentially degraded wetlands in need of *restoration* instead of valuable wetlands that could serve as *protection* offsets. I did this because 1) restoration offsets is the easiest way to ensure conservation *additionality* (i.e., offsets come in addition to measures that would have been implemented regardless of development impact), due to difficulties of determining when and how protecting existing biodiversity can be considered a gain (McKenney and Kiesecker, 2010; Wissel and Wätzold, 2010), and 2) the study area likely offers many restoration opportunities as Norway has lost 20% of its historic peatlands (Moen et al., 2010; Rekdal et al., 2016; Jakobsson & Pedersen 2020) and many are known to still suffer from old drainage ditches (Lyngstad et al., 2018).

Based on the exclusion criteria, I selected eight principles for avoidance (Table 1) and eight for offset (Table 2) from the complete lists of identified principles (Appendix A, Table S2 and S5). A brief description of how I applied these principles can be found in the tables below, while a more detailed explanation on how these were incorporated into a step-by-step approach for mitigation planning is described in the following sections (3.2.3 and 3.2.5).

***Table 1**. Avoidance principles applied in the case study. The principles were selected from the literature review results (Appendix A, Table S2), where more detailed descriptions can be found. The table includes description of how each principle was applied in the final MH methodology. Abbreviations in Table: PAA = Priority avoidance areas, DWPS = Designated Wind Power Site (described in section 3.2.3.3).*

| Main principle | Specific principle | Study application |
|---|---|---|
| **Avoid priority avoidance areas (PAA)** | Identify and minimize overlap between priority avoidance areas (PAA) and development site | Spatially visualize wetland PAAs and potential wind power sites (i.e., the DWPS) within study area. |
| | PAA: threatened species and ecosystems | Consider the national threat level for all included wetland plant species when determining avoidance and offset priorities |
| | PAA: high biodiversity areas | Consider habitat quality for a large amount of wetland species when determining avoidance and offset priorities. |
| | PAA: existing protected areas | PAs are already excluded from the DWPS by local authorities, making additional analysis unnecessary. |
| | PAA: areas in especially good ecological condition | Consider ecosystem condition when determining avoidance and offset priorities. |

| | | |
|---|---|---|
| **Choose avoidance strategy** | Avoidance through site selection | Use spatial mapping to visualize areas were wetlands are in conflict with wind power plans, thereby facilitating spatial avoidance planning. |
| **Plan on both large and small scale** | Two-phased: broad landscape-level planning + detailed project-level planning | Partly included by looking at both Finnmark (large-scale) and the smaller identified wind power site (the DWPS). But project-level assessment is not included. |
| **Start consideration early** | Start consideration as early as possible | Assess avoidance alternatives as the first step of the mitigation planning |

***Table 2***. *Offset principles selected for case study. The principles were selected from the literature review results (Appendix A, Table S5), where more detailed descriptions can be found. The table includes description of how each principle was applied in the final methodology.*

| Main principle | Specific principle | Study application |
|---|---|---|
| **Select offset type** | Restoration of a degraded site | Map potentially degraded wetlands that may serve as restoration offsets |
| **Ensure environmental justice** | Transparency in site selection process | Be clear in methodological choices and how offset site and conservation value is determined. |
| **Base methods on existing knowledge** | Build on scientific findings and principles. | All principles and methodologies are based on the findings from the research literature review. |
| **Plan on a landscape scale** | Offset site should be selected on a large scale to consider all options and maximize benefits | Analyze offset sites on a landscape scale. |
| **Select offset site based on impact site characteristics** | Ensure ecological equivalence | Generate maps that will facilitate this achievement by placing offsets in a) wetlands, b) the same bioclimatic zone and c) using a wetland replacement matrix to account for differences in wetland conservation value. |
| | Proximity to impact area | Outline potential offset sites in proximity to potential development sites. |
| **Maximize offset longevity** | Proximity to high-biodiversity areas | An aggregated habitat quality map for all Norwegian wetland plant species is used to determine conservation value, which in turn may be used for offset prioritization and size calculation. |
| **Determine offset size requirements** | Establish exchange rules/multipliers for determining offset size | Suggest specific impact-to-offset exchange ratios based on conservation value, ecological equivalence, and contemporary studies. |

### 3.2.2 General methodological choices

I assessed spatial avoidance priorities and offset opportunities by using the freely available Geographic Information System (GIS) software QGIS, version 3.22 Bialowieza. All included map layers were converted to the projected coordinate system ETRS-89 Lambert Azimutal Equal Area before analysis, as this projection gives accurate representation of area-based measurements.

All biological map layers included in the spatial analysis were raster data with an initial resolution of 500m * 500m, while the human pressure map (used to estimate ecological condition) was converted from vector to raster and could thus be adapted to any resolution. Although the pixel size of 500m is considered a relevant scale for assessing sedentary species and has been used by others conducting spatial biodiversity mitigation analyses (e.g., Gauthier et al., 2013; Bigard et al., 2020), the final map was downscaled to 200m * 200m to give a more accurate representation of the effect of human pressures on wetland conservation value. The downscaling was done in order to allow for fine-scale interpretation given by the linear distance from infrastructure; while the biological layers have a 500m pixel size, the continuous nature of the distance from disturbances makes it possible to increase the resolution.

### 3.2.3 Mitigation step 1: Identification of priority avoidance areas

I used the selected MH principles to develop the following three-step methodology for priority avoidance mapping: 1) identify wetlands within the study area, 2) calculate its conservation value and 3) assess overlaps between plans for wind power development and wetland ecosystems. The specific approach is summarized in Figure 2 and described in more detail in the following sections.

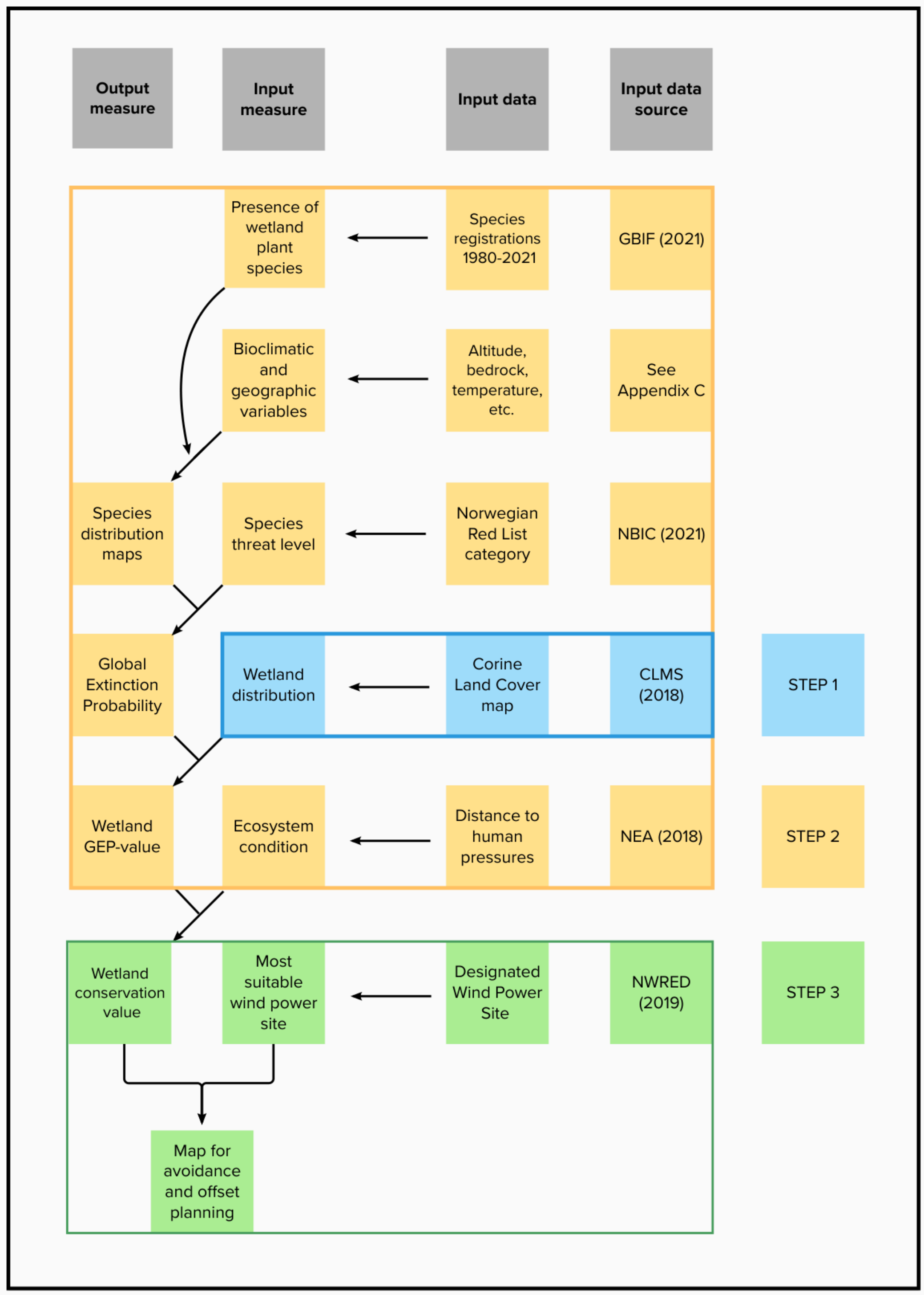


***Figure 3***. *Overview of the methodological approach and the data used to identify priority avoidance areas and potential offset sites. Step 1: Map wetlands in the study area. Step 2: Calculate wetland conservation value. Step 3: Assess conflict between wind power development and wetlands. Detailed data on the included wetland species can be found in Appendix B, and the variables used to predict species distribution is listed in Appendix C.*

#### 3.2.3.1 Mapping wetlands

I identified wetlands in the study area by using the Corine Land Cover dataset (European Environment Agency, 2018). It is based on satellite images used to categorize the global land surface into five major zones, including wetlands. I used the category of "inland wetlands", defined as:

> Areas flooded or liable to flooding during the great part of the year by fresh, brackish or standing water with specific vegetation coverage made of low shrub, semi-ligneous or herbaceous species. Includes water-fringe vegetation of lakes, rivers, and brooks and of fens and eutrophic marshes, vegetation of transition mires and quaking bogs and springs, highly oligotrophic and strongly acidic communities composed mainly of sphagnum growing on peat and deriving moistures of raised bogs and blanket bogs.
>
> (European Environment Agency, 2019)

Although this definition is broad and should be applied with care, the Corine Land Cover map provides easily accessible data that represents wetland extent in the study area in a meaningful way for landscape-scale analysis. For more small-scale assessments, other wetland mapping approaches should be considered to also account for differences among inland wetland types.

#### 3.2.3.2 Calculating wetland conservation value

Although ecosystem conservation value is commonly used to inform prioritization within mitigation planning, the exact metrics used vary widely (Brander et al., 2006; Kukkala and Moilanen, 2012; Bigard et al., 2020). I assessed wetland conservation value by combining measures of wetland habitat quality, species threat level and ecosystem condition (Figure 2). These parameters were selected because 1) they were identified in my literature review as important MH principles, 2) they are widely used for ecosystem evaluation purposes, 3) they can be addressed remotely with easily accessible data without requiring field work and 4) they make up a combined measure of *biodiversity*, directly relevant for biodiversity conservation, and indirectly relevant for assessing some levels of wetland function, as biodiversity is suggested a key driver of ecosystem services (BBOP, 2012; Cardinale et al., 2012; Harrison et al. 2014).

**Species distributions and threat level**

We created a list of all plant species naturally occurring in Norwegian wetlands based on the description provided in the handbook *Natur i Norge* (Bratli et al., 2022), before downloading data on spatial species occurrence from the GBIF database (see Appendix C). Species with less than 50 observations since 1980 were removed from the dataset to avoid biased estimations of species distributions. We also removed the species that had not been evaluated for inclusion on the Norwegian red list, using data from The Norwegian Biodiversity Information Centre (2021). Finally, 264 wetland plant species were included for further analysis (Appendix B).

As a proxy for species distributions, we modeled habitat quality for the selected species using MaxEnt as a statistical tool (Phillips et al., 2006). The GBIF observations was used as reference data to analyze the match between species occurrence and multiple bioclimatic and topographic variables, listed in Appendix C. In turn, this was used to estimate the habitat quality for each species in each data pixel in the study area.

Finally, we developed an overall habitat importance map based on the global extinction probability (GEP) estimate, as outlined by Kuipers et al. (2019). GEP combines the aggregated presence of species in a given area with the threat level of each species. As such, we combined the 264 independent habitat quality maps with the national red list threat level for each species to estimate the aggregated wetland habitat importance at a landscape scale (Equation 1). We used the threat level in a logarithmic scale in order to give more weight to threatened species.

$$\mathrm{GEP}_p = \frac{\sum_s \frac{o_{s,p} \times \mathrm{TL}_s}{\sum_p o_{s,p}}}{\sum_s \mathrm{TL}_s}$$

***Equation 1.*** *GEP equation, from Kuipers et al., (2019). GEP denotes Global Extinction Probability, where* ***o*** *is the occurrence of species* ***s*** *in cell (pixel)* ***p*** *and the threat level* ***TL****, as expressed in the Norwegian species red list.*

To exclude non-wetland ecosystems from the final GEP-map, the layer was clipped to match the identified wetlands. At last, the wetlands were split into three equally large groups (in terms of number of pixels = area), according to their GEP-value (Figure 4).

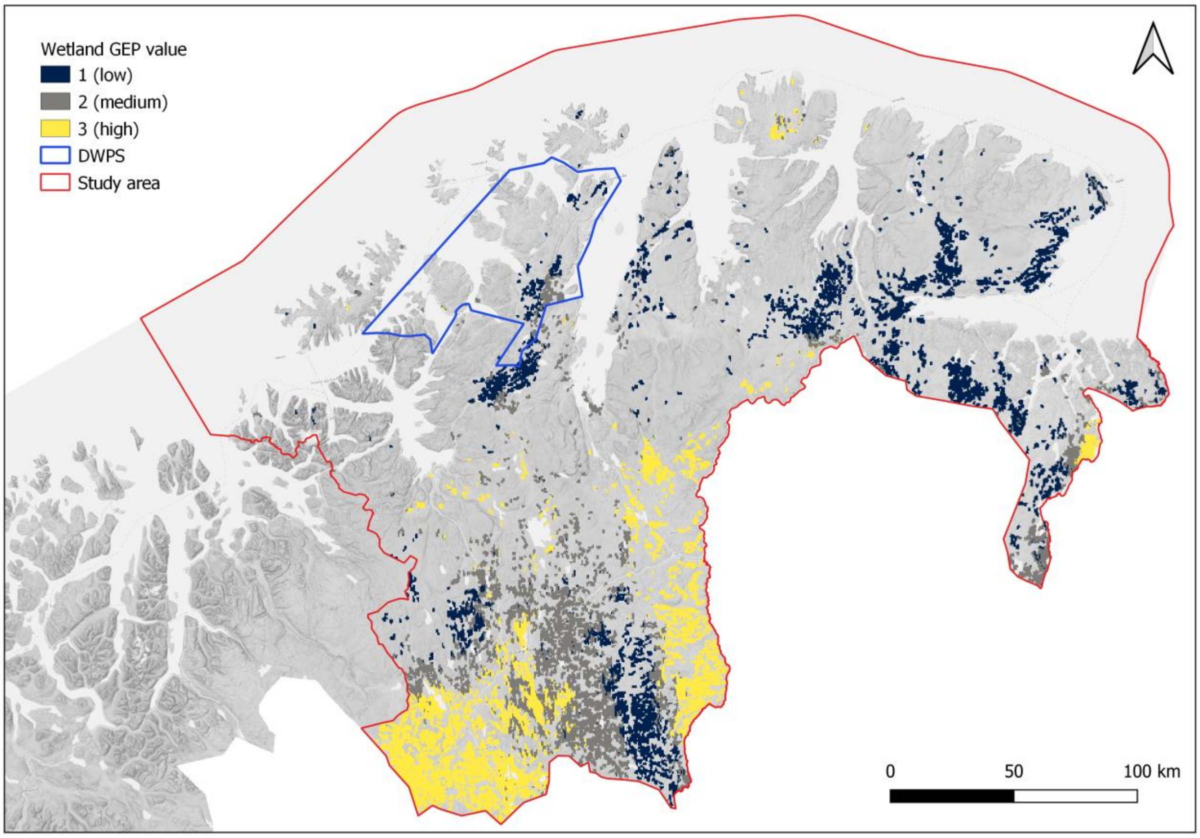


***Figure 4***. *Map of wetlands in study area, divided into three equally large groups according to their Global Extinction Probability index value. DWPS = Designated Wind Power Site.*

### Ecosystem condition

Because the ecological status of wetlands in the study area has not been mapped, wetland ecosystem condition was evaluated based on the level of human pressure. This has been shown to be a good proxy for ecosystem condition on a landscape scale, as the level of degradation is largely determined by the distance to human pressures, with areas close to such pressures being more degraded (Santos & Tabarelli, 2002; Laurance et al., 2009; Jones et al., 2022).

After assessing several publicly available data sets on human pressures, I found the best representation of my study area to be a map released by the Norwegian Environmental Agency (2023), showing distance-zones around human technical interventions (Figure 5). The validity of the maps was examined by visually evaluating the correlation between the assigned human pressure score and existing infrastructure visible on Google maps. I also emphasized the level of details (i.e., scale), favorizing high resolution.

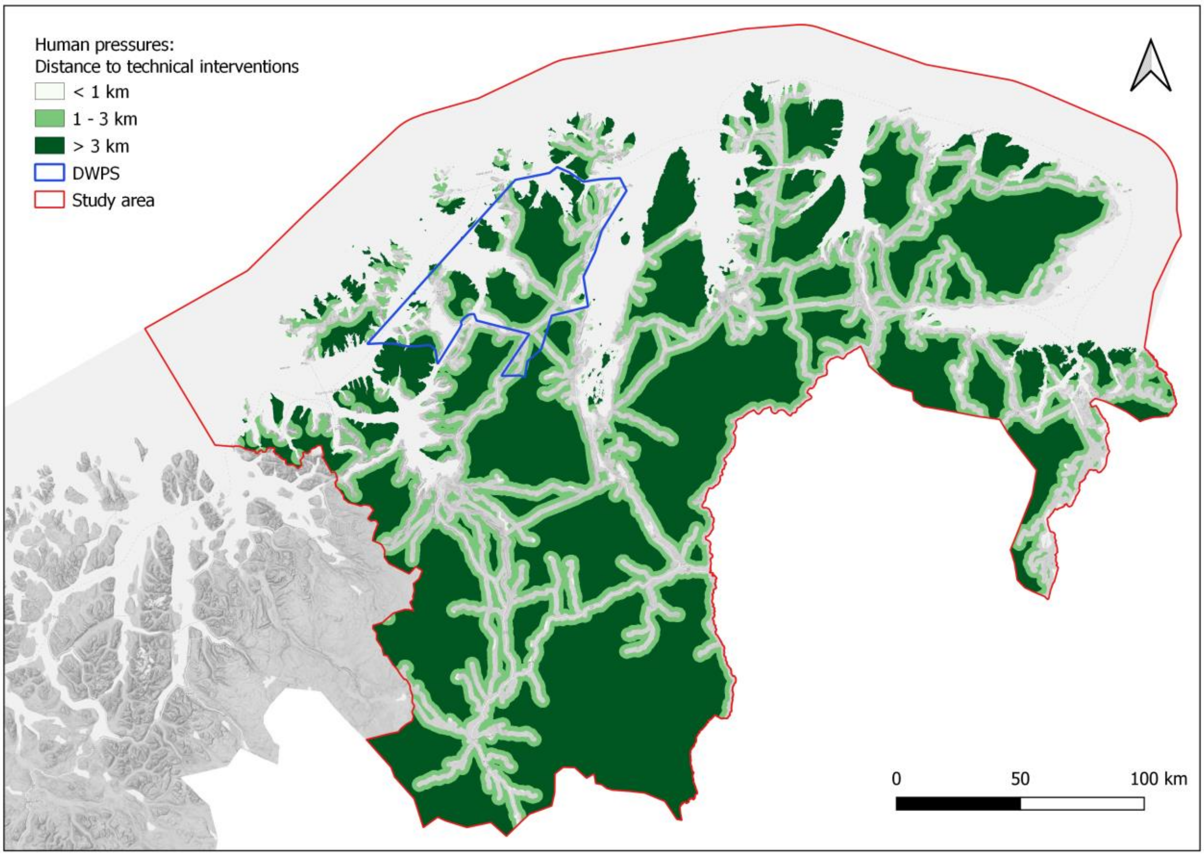


***Figure 5****. The human pressure map used as a proxy for ecosystem condition. The data layer shows three distance-zones around larger technical interventions, such as roads, buildings, and power lines. DWPS = Designated Wind Power Site.*

The selected map "Nature free of technical interventions" (Figure 5) is a vector layer highlighting areas more than 1, 3 and 5 km away from "larger technical interventions" such as roads, buildings, railways, major power lines and dammed rivers (Norwegian Environmental Agency, 2022b). Such placement of buffer zones around infrastructure is commonly used to account for indirect impacts extending beyond direct development footprints (Barber et al., 2014; Benítez-Lopez et al., 2017; Jones et al., 2022). Examples of indirect impacts include chemical and noise pollution, creation of habitat edges, movement barriers, and increased human access (Forman & Alexander, 1998; van der Ree et al., 2015; de jonge et al., 2022)

Impact distances vary widely among species, infrastructure type and local context (Benítez-López et al., 2010; de Jonge et al., 2022). To better reflect relevant impact distances for wetland species, I removed the 5 km zone, because the negative effects of technical intervention on wetland ecosystems was considered very small on distances above 3 km. 1 and 3 km buffer zones were regarded relevant after considering available data and that wetlands in the study area typically are affected by drainage due to infrastructure development, agriculture, or off-piste driving of ATVs, causing potentially far-reaching impacts (Fylkesmannen i Finnmark, 2010; Hansen & Jensen, 2010) .

Nature free of technical interventions is in itself considered valuable because of its provision of large, coherent natural habitats, important to biodiversity, recreation and climate change adaptation (NEA, 2022b). Yet, in Norway such undisturbed areas have experienced a dramatic decline since the 1800s (NEA, 2022b). Including this measure for evaluating wetland conservation value may thus have additional advantages in terms of decreasing the pressure on such areas, directing infrastructure development to areas already affected by humans.

**Final wetland conservation value**

I calculated a final wetland conservation value by combining GEP-value with wetland ecosystem condition according to the matrix in Table 3.

***Table 3**. Matrix showing how the measures of GEP value (i.e., habitat quality and species threat level) and ecosystem condition were combined to assign a total conservation score from 1 (lowest) to 5 (highest). The GEP-classes were created by dividing all wetlands in Finnmark into three equally large groups according to their value.*

| | **Wetland condition: Distance to technical interventions** | | | |
|---|---|---|---|---|
| **GEP value** | | **0-1 km** | **1-3 km** | **>3 km** |
| | **Low** | 1 | 2 | 3 |
| | **Moderate** | 2 | 3 | 4 |
| | **High** | 3 | 4 | 5 |

### 3.2.3.3 Assessing conflict between wind power development and wetlands

Although general priority avoidance maps can be useful for avoidance planning related to diverse development projects, mitigation planning involves looking more closely into *potential* project locations (Kiesecker et al., 2010; Bigard et al., 2020; Jones et al., 2022). Wind power development is only economically and socially viable in areas with a high annual wind speed, low cost of production, and low conflict-level with other stakeholders (NVE, 2019). As such, an assessment of suitable wind power plant areas is required before more detailed avoidance planning can be initiated.

In 2019, The Norwegian Water Resources and Energy Directorate (NVE) released a suggestion for a “National frame for wind power on land”, where they outlined the 13 most suitable large sites for wind power development throughout Norway. The selection was based on extensive analyzes of wind power potential, possible environmental effects, and stakeholder interests (NVE, 2019). One of these “designated wind power sites” (DWPS) was outlined in Finnmark. Thus, although wind power development can potentially occur outside of the DWPS, this site represents the area where developers are most likely to be given permission for such development, and subsequently where mitigation planning is primarily needed.

#### 3.2.3.4 Identified priority avoidance areas

Figure 6 highlights the identified wetland priority avoidance areas in the study area. As previously emphasized, all wetlands should ideally be avoided because of their unproportionally high value for humans and nature (Costanza et al. 2014; Kingsford et al., 2016), their substantial decline (Davidson, 2014), and because it is the safest way to ensure NNL of biodiversity (Phalan et al., 2018). However, in situations where this is not achievable, the wetlands’ conservation value, as shown in Figure 6, can be used for prioritization purposes.

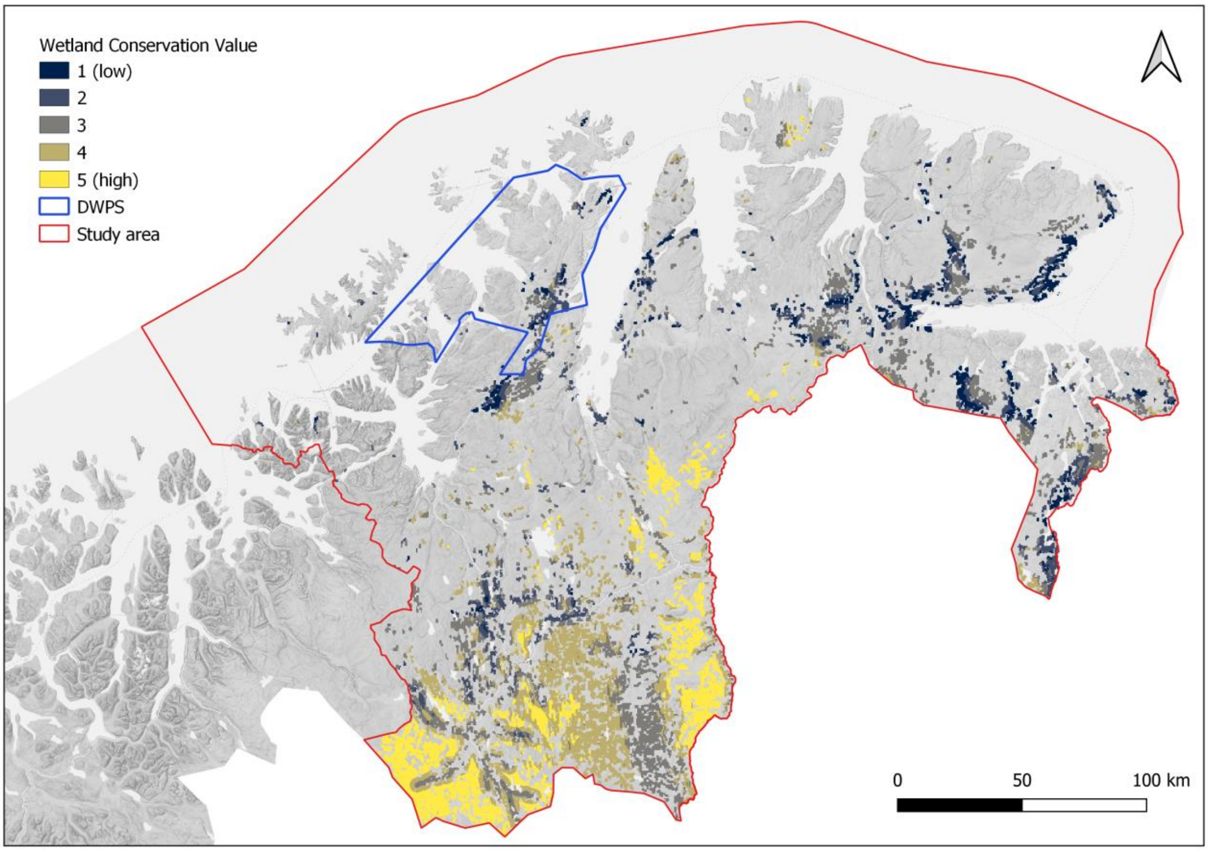


***Figure 6**. Priority areas for wetland avoidance in the study area, based on wetland occurrence and its calculated conservation value. DWPS = Designated Wind Power Site.*

Some general wetland distribution patterns are evident on the avoidance map (Figure 6); most wetlands are found in the low-laying southern part of the study area, along the border to Finland. This is also the area with most wetlands of the highest conservation value. In the East and along the coast, the values are generally lower, except for Nordkinn peninsula in the North, where we find wetlands of the highest conservation value. In total, the map displays 10160,15 $km^2$ of wetlands, accounting for 22,2 % of Finnmark's total land area, based on the Corine land cover assessment. The mean conservation value of these wetlands was found to be 3,4 and their distribution across conservation classes is shown in Table 4.

***Table 4****. Calculated wetland extent in the study area of Finnmark and within the Designated Wind Power Site (DWPS). Conservation class 1 = lowest conservation value.*

| Wetland conservation class | Area (ha) within Finnmark | Wetland proportion (%) within Finnmark | Area (ha) within DWPS | Wetland proportion (%) within DWPS |
|---|---|---|---|---|
| **1** | 77 134,9 | 7,6 | 5 064,5 | 20,3 |
| **2** | 144 541,0 | 14,2 | 11 909,0 | 47,8 |
| **3** | 300 363,1 | 29,6 | 7 648,7 | 30,7 |
| **4** | 281 621,3 | 27,7 | 280,0 | 1,1 |
| **5** | 211 699,0 | 20,8 | 0,0 | 0,0 |

Figure 7 provides a closer look at the Designated Wind Power Site for a better view of the wetlands within and near this area. This allows for easier separation among wetlands of different values. The DWPS is the site most likely to experience wind power development in the future, thus representing the area of greatest potential conflict and where avoidance planning is most critical to conserve wetlands. This site overlaps with 24 902,2 ha of wetlands. Interestingly, it contains no wetlands of class 5, only 1,1 % of wetlands class 4 (Table 4), and the mean value is 2,1. This indicates that the site contains wetlands of less conservation importance than the study area average. However, this does not mean that avoidance measures are unnecessary, and developers should nonetheless strive for avoidance. Fortunately, the map also shows that the wetlands within the DWPS are clustered in the South-Eastern and North-Eastern parts, and thus may easily be spatially avoided. In situations where this is not possible, the next three steps of the mitigation hierarchy must be employed.

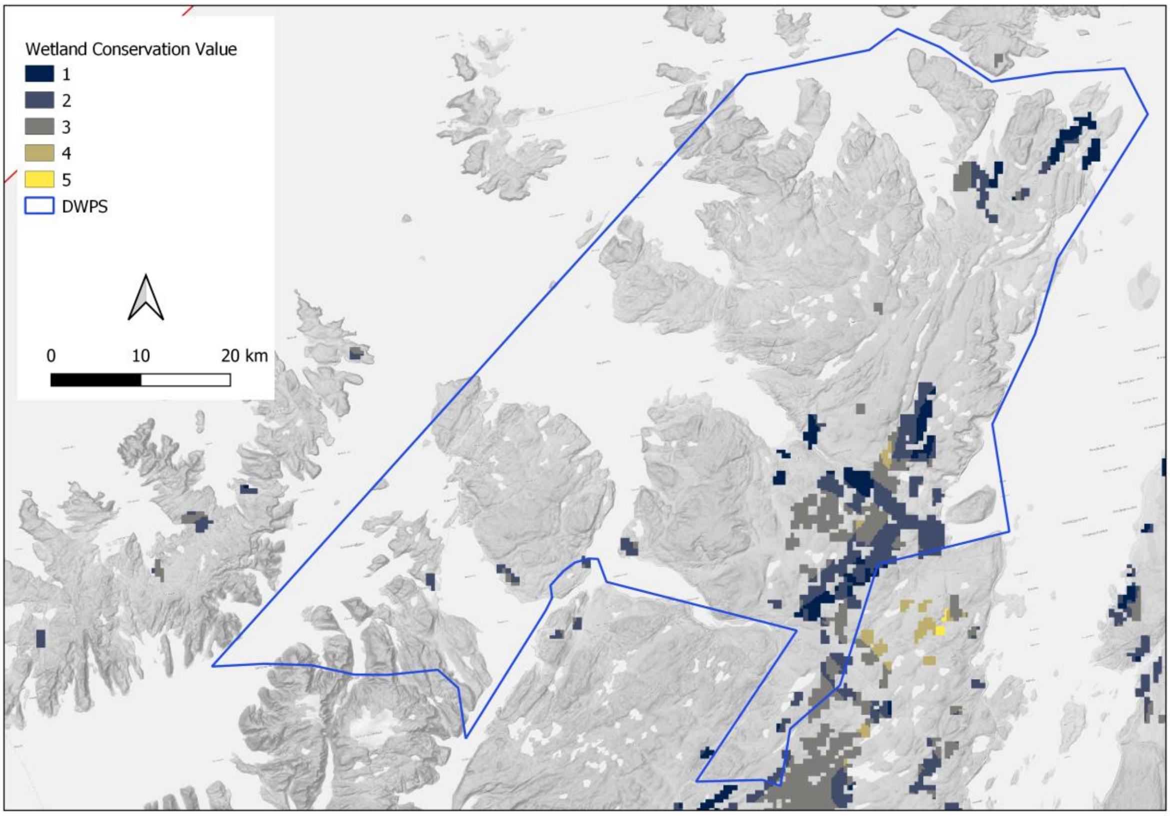


*__Figure 7__. A closer look at the wetland priority avoidance areas within and in proximity to the Designated Wind Power Site (DWPS). The map shows wetlands and their determined conservation values.*

It is important to note that outside the DWPS only *wetland* avoidance areas are displayed, whereas within this site many other factors relevant to wind power development are also considered (e.g., wind speed, production costs, protected areas, stakeholder interests). Thus, before making use of the avoidance map in Figure 6 for wind power planning outside of the DWPS, assessments of other important criteria must be conducted.

### 3.2.4 Mitigation step 2 and 3: Minimization and restoration measures

Given the highly project and site-specific nature of minimization and restoration measures, there is no reason to delineate more specific suggestions than the general principles outlined in Appendix A, Table S3 and S5. Once project location, size and other characteristics are determined, context-appropriate minimization and restoration strategies can be developed using these principles.

### 3.2.5 Mitigation step 4: Scoping for offset location and size

#### 3.2.5.1 Mapping degraded wetlands

Due to lack of data on degraded wetlands in the study area, I mapped wetlands with the highest *probability* of being degraded. A search for such a range of sites with the greatest potential to serve as offsets is an approach applied in similar studies (see e.g., Bigard et al., 2020; Jones et al., 2022). I used the priority avoidance map, adding the human pressure map to further visualize the wetlands in proximity to such pressures. As explained previously, wetland degradation is closely related to human pressures (Benitez-Lopez et al., 2010; Jones et al., 2022).

To use the same base map as already generated for avoidance planning gives a set of advantages, including less work for potential users, less chance of misunderstandings and user error, and it allows for an easy way to calculate offset *size* requirements by comparing impact and offset site characteristics directly on the same map (explained in more detail in section 3.2.5.3).

Schumann and Joosten (2008) emphasize that also restoration *feasibility* should be considered when determining offset site, because it highly affects the likelihood of restoration success. I considered this aspect by evaluating three overarching wetland parameters related to restoration feasibility: degradation probability, wetland accessibility, and the likely cause of degradation. While the former two factors are negatively correlated to distance to human pressures, thus indicating a higher offset feasibility in wetlands close to human pressures, I consider the latter aspect to favorize wetlands further away from such pressures. This is because the cause of degradation in wetlands closest to human pressures (i.e., technical interventions), is likely to be the technical intervention itself (e.g., roads, powerlines), which is difficult and expensive to alter (Jones et al. 2022) in a way that promotes wetland restoration. Degradation occurring further away from such pressures is more likely to be caused by factors more easily changed. To account for these “feasibility tradeoffs”, I have highlighted the wetlands within 1 and 3 km from technical interventions. I consider wetlands within this zone close enough that the probability of degradation and the accessibility remains high, yet far enough away that the likelihood of restoration success is high.

### 3.2.5.2 Identified potential offset sites

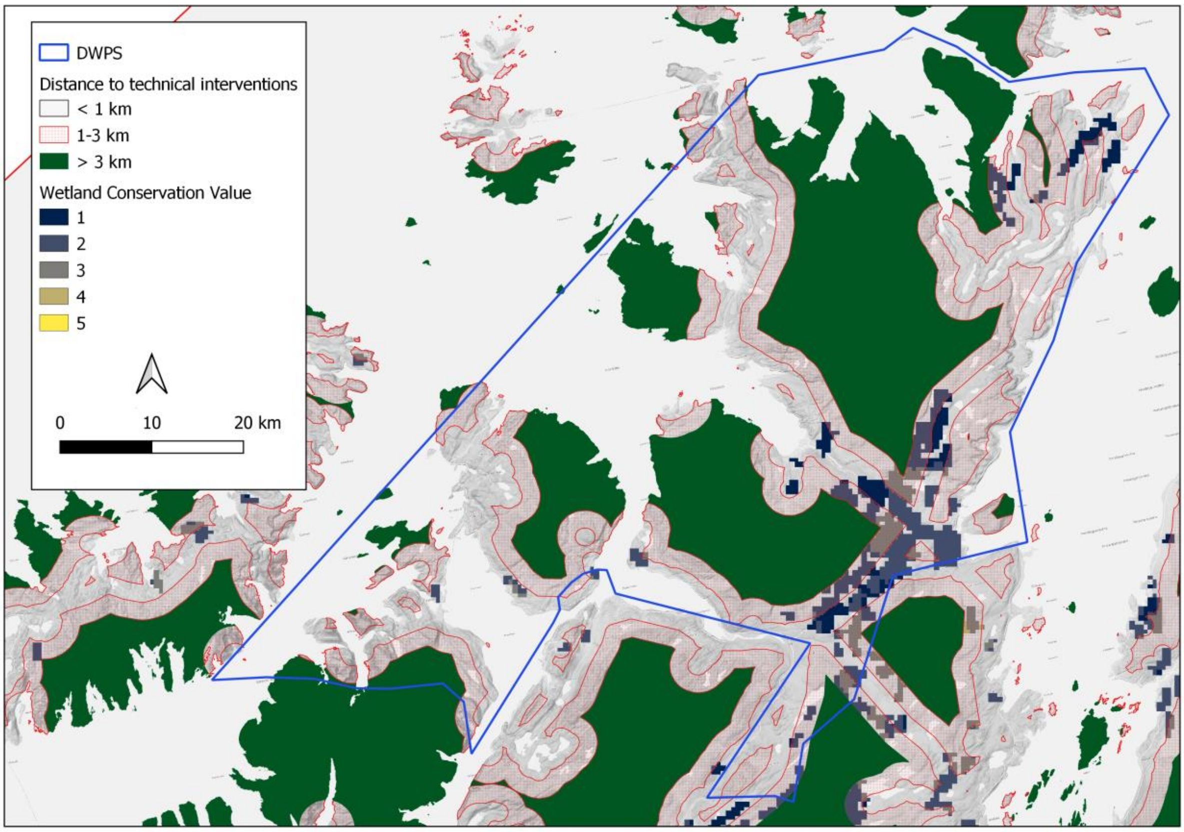


***Figure 8****. Potential wetland offset sites. The wetlands within the red zone are considered most feasible for restoration efforts. DWPS = Designated Wind Power Site.*

The wetlands within the red zone in Figure 8 represents the wetlands considered most feasible for restoration efforts. The map only displays wetlands within and adjacent to the DWPS, because offsets should be placed close to impact site (Kiesecker et al., 2009).

### 3.2.5.3 Selecting offset location based on impact site characteristics

Once development site is chosen, the map showing potential offset sites (Figure 8) can be used to outline a more specific offset location based on certain key features of the affected development area. There are especially two aspects relevant to consider: first, according to my literature review (Appendix A, Table S5), offsets should be placed as close as possible to the impact site to ensure that biodiversity gains accrue to the area affected by development (Kiesecker et al., 2009). Therefore, post-analysis field-surveys should start assessing restoration opportunities in this area. Second, ecological equivalence - the most fundamental offset principle (McKenney & Kiesecker, 2010; Quétier & Lavorel, 2011), can be achieved by offsetting in a) wetlands (Figure 8), b) in the same bioclimatic zone (Figure 9), and c) by accounting for differences in wetland conservation value. Factor a and b can be planned for by using Figure 8 and 9 in combination, whereas c can be considered by establishing "conservation value"-specific replacement ratios, as described in the following section.

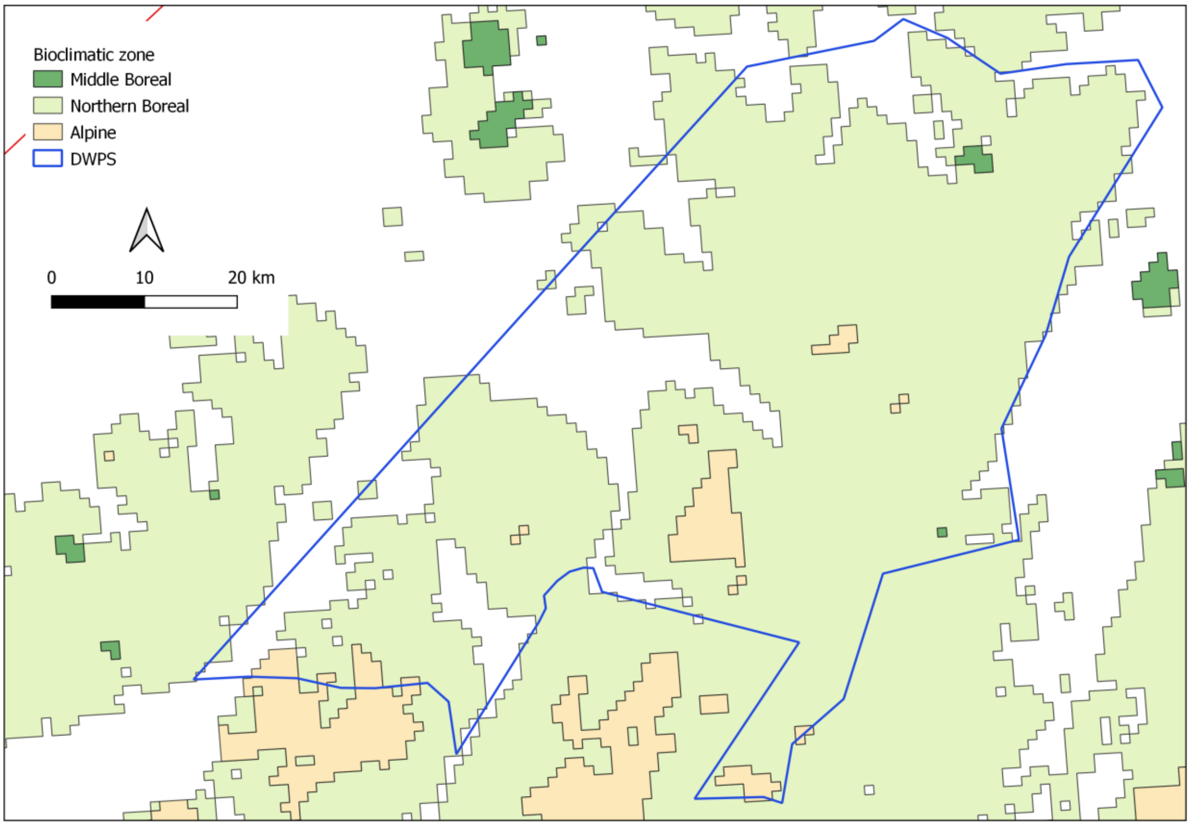


***Figure 9.*** *Bioclimatic zones for the Designated Wind Power Site, as determined by Bakkestuen et al. (2008). Data retrieved from NINA and NEA (2019). Within the DWPS most wetlands belong to the Northern Boreal zone and the likelihood of finding wetlands in need of restoration within this zone is therefore high. If impacting wetlands in one of the two other bioclimatic zones, offsets may be required to take place further away to meet the requirement of ecological equivalence.*

### 3.2.5.4 Determining offset size

To reach NNL of biodiversity, offset site gains must compensate for impact site losses (BBOP, 2018). Once offset site location is chosen, the accomplishment of this goal relies largely on offset *size* (Moilanen and Katiaho, 2018a). However, determining a reasonable offset size is a highly complex task (Moilanen and Katiaho, 2018a). This study outlines three central aspects that should be considered, and show how the determined wetland conservation value classes feed into these: 1) the size of impact site triggering offset requirements, 2) the quality difference (i.e., degree of ecological equivalence) between impact and offset site (Moilanen and Katiaho, 2018b), and 3) that offset gains are typically lower than impact site losses per unit area (BBOP, 2018).

### Size of impact site

Before measuring the size of impacted area, managers must first decide on what *type* of impacts that will trigger compensation requirements. Alternatives range from impacts above a certain threshold to all measurable impacts (Moilanen and Katiaho, 2018b). To fully reach NNL, offsets must be required for all measurable impacts. When the type of impacts triggering compensation is decided, field surveys and estimated indirect impact zones (such as proposed by Benitez-Lopez, 2010; Lembrechts et al. 2014 and de Jonge et al. 2022) can be used to calculate project impact size.

### Ecological equivalence

Ensuring ecological equivalence between impact and offset site is considered a key offsetting principle and is typically addressed by locating offsets to areas with similar ecological characteristics or values as impact site (McKenney & Kiesecker, 2010; Quétier & Lavorel, 2011). Therefore, offsets should ideally be placed in wetlands with the same conservation value as the impacted wetland. But because such strict like-for-like exchanges are not always possible, we must establish replacement ratios to account for differences in wetland conservation value. That is, to compensate for impacts to a wetland class 5 (highest value) in a wetland class 1 (lowest value) one must restore a larger area than what was impacted to ensure equivalent biodiversity gains.

### Offset gains versus impact losses per unit area

Offset gains per unit area is most often lower than impact losses per unit area (BBOP, 2018; Moilanen and Katiaho, 2018a). Such differences are caused by direct impact losses often being complete (i.e., meaning that local biodiversity is reduced to 0), whereas restoration gains is only partial - because restoration is almost never applied to a fully lost ecosystem, and is unlikely to recover the ecosystem fully back to its original state (Moilanen and Katiaho, 2018a). It can also be caused by failing to follow important offset principles (listed in Appendix A, Table S5), causing time delays, uncertainty, and lack of additionality, among other things (Moilanen and Katiaho, 2018b). To account for this, offset size must be relatively larger than impact size (BBOP, 2018; Moilanen and Katiaho, 2018a). Although the optimal multiplier that will ensure full compensation for such differences will vary widely among projects, studies indicate that it should be close to 10:1, or even higher (Moilanen and Kotiaho 2018a; Gibbons et al., 2016; Laitila et al., 2014; Moilanen et al., 2009). However, as these studies aimed to determine an adequate multiplier independent of differences in ecosystem conservation value, the ratio should be modified somewhat to account for this aspect. In my suggested wetland replacement matrix (Table 5) I have therefore used the 10:1 ratio as a "base line multiplier" for the ratio required to offset in wetlands of the same conservation value class as the impacted wetland but modified it for use in other situations.

**Suggested wetland replacement matrix**

Based on the considerations outlined above, I have developed a wetland replacement matrix (Table 5) that may serve as an example for how differences in 1) wetland conservation value and 2) typical rates of biodiversity losses at impact site vs. gains at offset site can be considered in offset size calculations. The table can be used to determine an adequate offset size after impact size is measured.

***Table 5**. The proposed wetland replacement matrix describing how many times bigger the offset area should be compared to the impact area to reach NNL of biodiversity. The suggested ratios are based on differences in wetland conservation values (from 1-5 in the Table) and typical contrasts between biodiversity losses at impact site and biodiversity gains at offset site.*

| **Conservation value of** | | **Offset wetland** | | | | |
|---|---|---|---|---|---|---|
| | | ***1*** | ***2*** | ***3*** | ***4*** | ***5*** |
| **Impacted wetland** | ***1*** | 10:1 | 9:1 | 8:1 | 7:1 | 6:1 |
| | ***2*** | 11:1 | 10:1 | 9:1 | 8:1 | 7:1 |
| | ***3*** | 12:1 | 11:1 | 10:1 | 9:1 | 8:1 |
| | ***4*** | 13:1 | 12:1 | 11:1 | 10:1 | 9:1 |
| | ***5*** | 14:1 | 13:1 | 12:1 | 11:1 | 10:1 |

# 4 Discussion

In this study I have identified commonly used mitigation hierarchy principles, from which I have developed a mitigation planning methodology for conserving wetland biodiversity under pressure from infrastructure development. I propose a framework that could support early anticipation of impacts on a landscape scale which will significantly increase decision-makers ability to prevent wetland biodiversity loss, in line with global targets of no net loss. I outline a complete step-by-step approach for identifying priority avoidance areas - considered the most important mitigation hierarchy step, as well as how potential offset sites can be mapped. I also include suggestions for how offset size can be calculated based on scientific principles.

## 4.1 Mitigation hierarchy principles

To deal with the abundance of existing mitigation hierarchy principles, as well as the lack of a formally recognized methodology, I conducted a comprehensive literature review identifying commonly applied principles (i.e., methods, content, and recommendations) for each of the four mitigation steps. For my study purpose, the review results (Appendix A, Table S1-S5) served primarily as a basis for creating a mitigation methodology. As the current body of research is fragmented, these findings may also be of practical value as an overview for managers aiming to develop their own mitigation policy. Although the search was focusing on wetland relevant principles, the identified principles are general enough to be applicable to conservation policies targeting diverse development projects and ecosystems.

In the literature review, I found the minimization and restoration principles to be highly context-specific and they therefore need to be developed on a project-by-project basis, whereas the avoidance and offset strategies can be more easily streamlined and adapted to mitigation planning of a wide range of wetland impacts on a landscape scale.

## 4.2 A proposed mitigation planning approach for wetland conservation

In short, my proposed avoidance-methodology consists of three steps: 1) detect wetlands using satellite-derived data, 2) assess wetland conservation value using measures of habitat quality, species threat level and ecosystem condition, and 3) spatially visualize and minimize overlaps between potential development sites and wetlands. The final “wetland conservation value”-map facilitates avoidance-planning by highlighting where the most valuable wetlands are located on a landscape scale. Applied in an early stage of project planning, this will help managers steer away from such areas and focus more labor-intensive surveys (e.g., environmental impact assessments and field work) on sites of lower conservation value and hence more appropriate for development. In turn, this will reduce planning costs, improve wetland conservation outcomes, and minimize offset requirements.

For locating potential offset sites, I found it practical to stick to the same three-step approach as developed for avoidance. In that way, the final avoidance map can serve as an offset map with only minor changes. I did this because 1) the same biological measures were considered relevant for evaluating offset placement, 2) there is a lack of more precise wetland degradation data that could add additional constraints for offset localization, 3) impact and offset site characteristics can thereby be compared directly on the same map for determining offset size, and 4) it will help keep the required workload down for decision-makers and other potential users, which has been shown important for practical application (Kusler, 2006).

Whereas step 1 and 3 of the developed methodology are applied in a similar fashion by most researchers scoping for avoidance and offset sites (see e.g., Kiesecker et al., 2010; Bigard et al, 2015; Jones et al., 2022), the way in which ecosystem conservation value is assessed (step 2) vary greatly (Brander et al., 2006). While cost-benefit analysis of potential development projects in wetlands are commonly informed by monetary valuation approaches (Brander et al., 2006), landscape level mitigation planning rather seeks to prioritize among areas. Calculating replacements costs or "willingness to pay" as commonly done in wetland *valuation* (Brander et al., 2006), is thus less relevant. But also within mitigation planning, there are various ways to determine wetland value. My approach differs from those requiring in-field surveys to assign a wetland score (see e.g., Alberta Government, 2015a; Department of Parks and Recreation, 2010). Whereas such evaluations may give a more accurate estimate of wetland value highly useful for *detailed* project planning, it requires considerably more time, data, and resources, meaning that such assessments are not practically feasible on a landscape level (Jones et al., 2022). Remotely based methodologies can analyze wetland value over vast areas and can therefore be easily integrated into a landscape-scale analysis at a lower detail level, meaning that detailed efforts can be focused on the areas of interest only. I have shown how data on species distributions, species threat level and ecosystem condition can address a number of essential avoidance and offset principles relevant for landscape level planning (Table 1 and 2). A major advantage with these parameters is that data sets are often publicly available. The GBIF database, from which data was collected to predict wetland species distribution, is part of a growing assembly of citizen-engaging platforms that have attracted users from all over the globe, generating a substantial amount of species observations. While this kind of data needs to be carefully considered in the context of species distribution modelling (Beck et al., 2014), the observations can be used to estimate habitat quality (or species presence likelihood) using a wide range of bioclimatic and topographic variables.

As a substitute for the lack of concrete data on degraded wetlands, I mapped wetlands with the highest *likelihood* of being degraded based on proximity to human pressures - in line with other landscape scale studies (Venter et al., 2016; Heiner et al., 2019; Jones et al., 2022). Thus, there is no guarantee that the outlined offset areas are degraded, and field work is required for verification. However, this type of analysis will limit the amount of wetland that needs to be assessed in more detail through such efforts. Collecting more precise data on degraded wetlands will improve offset planning significantly by constraining the mapped offset opportunities further, thereby reducing the amount of required post-analysis field work. The need for such mapping surveys has also been highlighted by Norwegian authorities as a priority for future national wetland conservation (NEA, 2020). However, this will likely require extensive analyses of aerial photos, Lidar-surveys, and other spatial data sets (NEA, 2020). My proposed method provides a promising alternative solution for offset mapping until this is in place. In other regions of the world, resources for extensive analysis of wetland degradation are not likely to be available in the near future. Therefore, a landscape-scale assessment based on already existing information, such as the method I propose here, might be necessary for strategic conservation planning using the mitigation hierarchy.

This study adds to the growing research literature showing the advantage of upscaling mitigation planning from project to landscape level (see e.g., Bigard et al. 2020; Jones et al., 2022; Kiesecker et al., 2010; Habib et al., 2013). Although the MH on a project level is important to mitigate specific impacts of individual projects, the MH on a landscape level can complement such efforts by improving overall impact avoidance and offsetting (Bigard et al., 2020; Jones et al., 2022). It also provides for a tool that may help decision-makers move away from small, disconnected mitigation efforts considered in isolation, towards a systematic, coherent approach that focuses on the bigger picture (Bigard et al. 2020). This is important because each development project may often not cause major negative impacts. Instead, it is the cumulate impact that is destructive for global biodiversity (Whitehead et al., 2017). By planning on a landscape scale, managers retain the capacity to consider how and to what degree many small development impacts together exert a collective pressure on natural environments and can act thereafter.

This study also shows how more detailed ecosystem assessments can be integrated with landscape scale mitigation approaches. Existing large-scale analysis are often developed for all present ecosystems simultaneously, which often comes at the expense of the level of detail for each ecosystem. For example, Jones et al. (2022), developing a mitigation hierarchy for a case study in Mozambique, determined ecosystem value solely on the basis of ecosystem *types* (although also ecosystem condition was considered in a later step), giving a high score to threatened ecosystems, without assessing further ecological differences. Although such methodologies are common (Bezombes et al., 2018; Jones et al., 2022), they are not capable of capturing deeper variations in ecosystem value - essential to consider when planning for no net loss of biodiversity. While acknowledging the difficulty of finding the optimal level of details in landscape-scale analysis, I claim that classifying each ecosystem type into a few value-groups, such as my proposed five wetland conservation classes, strengthens the credibility and usefulness of such planning practices significantly because of the increased opportunity to avoid important habitats and ensure ecological equivalence when selecting offset site.

Although developed for wetlands, the method can easily be adapted to other ecosystems. For example, priority avoidance and offset areas for pine forests can be mapped by replacing data on wetland extent and wetland-dependent species with pine forest extent and pine forest species. The applied human pressure map can either be used without any major changes or be adapted to reflect more relevant impact-distances for the targeted ecosystem. These adjustments require less effort and investment than developing scattered surveys in a landscape, providing more useful information in shorter time.

An important thing to note is that the two parameters used to determine final conservation value (i.e., GEP-value and ecosystem condition) are sometimes contradictory for a specific area. The wetlands with the highest GEP-values (promoting a high conservation score) are often found in areas with low ecosystem condition (promoting a low conservation score). The reason is found in a confounding variable; the *valleys* provide for both a warmer climate beneficial for high biodiversity, but also for increased human settlements (and thus human pressure) due to better farming opportunities, less exposure to harsh winter conditions, and natural guidelines for infrastructure development. On the other hand, the low-value areas for both species richness and human pressures are found in the mountains. In practice, this causes a low variation in the final conservation value among the wetlands in the study area. Only focusing on GEP-value would give a wrong representation of conservation value because many wetlands with high GEP value are degraded because of their proximity to human pressures - which is not accounted for in the GEP-score. Vice versa, focusing primarily on level of pressures in the landscape would result in assigning higher conservation scores in remote mountains that do not have high levels of biodiversity. This shows the importance of combining measures of potential biodiversity value developed from distribution models (e.g., GEP) with ecosystem condition metrics, unless measures of ecosystem condition is included when predicting suitable habitat.

## 4.3 Study limitations

As all frameworks, my proposed methodology has its limitations. Models simplify reality and will never capture every aspect relevant to nature conservation. I have not included all identified mitigation hierarchy principles from the literature review and thus, there are other aspects that could potentially be added to improve the framework. However, my aim has not been to develop a methodology that considers *all* aspects relevant to wetland conservation. Rather, I have concentrated on creating an easily applicable framework that is *sufficient* in its consideration of avoidance and offset sites, to be useful as a tool for helping decision-makers reaching their wetland conservation targets.

I focused on the wetland's importance to biodiversity, as well as its intactness to determine wetland conservation value. Although being commonly used metrics for such purposes, they do not cover all perspectives of wetland value. For example, wetlands offer several essential ecosystem services to human societies, and such services could potentially be considered when determining wetland conservation value. However, area-specific data on ecosystem services are rarely available, and typically require field surveys. Thus, it is more appropriate to consider these parameters in detailed project-based mitigation planning than in landscape scale analysis. For suggestions to how this may be done, I point to the wetland mitigation directive of the Canadian province of Alberta (Government of Alberta, 2015a). Here, managers have developed a protocol for dividing wetlands into four value categories by measuring a set of functions related to hydrology, water quality, ecological (habitat), and human use (Government of Alberta, 2015b). The wetland's contribution to carbon sequestration is another ecosystem service commonly included in such evaluations (Brander et al., 2006).

Furthermore, even the biodiversity parameter cannot be easily measured. As Kiesecker et al., (2009) points out, one must select a set of indicators, based on time, data, and resources, believed to represent biodiversity in a meaningful way, and it will thus be influenced by subjective decisions (Moilanen and Katiaho, 2018a). However, regardless of how ecological condition is assessed, such analyses are a first step and will always need to be validated through fieldwork (Jones et al., 2022). Hence, although my selected metrics represent an effort to capture meaningful variation in conservation value across wetlands, it should be tested through field validation.

Another factor to be aware of is that the study area boundaries chosen before dividing wetlands into three equally large GEP-value groups (as done to determine wetland conservation value) highly affects the final conservation score. For example, dividing the wetlands in all of Norway into three such groups would likely cause substantial differences in conservation score among major geographical gradients (Brinson, 1993). While this can be useful to inform national wetland conservation efforts, it may impede conservation in a local context by failing to capture small-scale variation in conservation value. Therefore, it is important to carefully consider what may count as a reasonable study area before calculating wetland conservation values. Factors worth to consider include natural borders (e.g., jurisdictional units and ecological gradients) and local wetland conservation goals. Nonetheless, what is considered a low or a high conservation score will always be *relative* – it depends on comparing areas across cases, and it is therefore not a challenge unique to this methodology.

Additionally, data on species distributions, threat level and ecosystem condition must be available to perform the suggested analysis, and the data quality will affect the modelling outcome. However, these aspects are considered by including data that should be easily available for most regions, and habitat quality maps are used instead of registered species distribution maps to minimize species observation bias.

## 4.4 Recommendations for implementation and future research

I suggest implementing this approach in collaboration with stakeholders. Especially wetland specialists, construction companies and affected local citizens should be included in the selection of development and offset site. In my case study area, most of the land is owned by the state and governed by the local management authority, but in other situations permission by private landowners is also essential (Bigard et. al., 2020).

Another important factor for implementation success is to adapt the methodology to the local context. The local authority may have additional policies regulating nature conservation. For instance, Norway has decided on certain specific regulations for wetland restoration and offsets, requiring cost-efficiency analysis, voluntariness of landowners and the fulfillment of a set of "fundamental prerequisites for restoration efforts" before measures can be initiated (NEA, 2020). These prerequisites restrict restoration efforts to locations not in conflict with agriculture or forestry, where plans for development do not exist and where local authorities have sufficient capacity for planning and implementation of the restoration efforts (NEA, 2020). Although these criteria are aimed at restoration work initiated independent of concrete mitigation efforts, local laws and restrictions must always be considered.

Based on the previously mentioned study limitations and implementation challenges, I suggest future research address the following aspects: 1) the possible gain and feasibility of considering more MH principles in landscape scale analysis, 2) field validation of the assigned wetland conservation score, 3) combining landscape-level planning with small-scale project mitigation efforts, and 4) developing a more standardized and meaningful way of measuring wetland impacts of a development project. This would be highly useful as *impacts* constitutes the key factor guiding the implementation of the MH, yet vary greatly among species, locations, and infrastructure types (Barber et al., 2014; Benítez-Lopez et al., 2017). In particular, there is a need to decide how the difference between direct and indirect impacts can be accounted for when determining offset requirements.

# 5 Conclusion

In this study I have outlined a path for how no net loss of wetland biodiversity can be planned for through the application of the mitigation hierarchy on landscape scale. When correctly implemented, the MH constitutes a powerful tool, balancing the needs of development projects with conservation goals. I have unraveled key principles important to consider in this context, and in line with the principal structure of the MH, I propose that developers are required to 1) *avoid* affecting wetlands altogether, alternatively avoid affecting wetlands of high conservation value, by following principles outlined Table S2, 2) *minimize* necessary impacts by following principles listed in Table S3, 3) *restore* on-site impacts through applying concepts explained in Table S4, and 4) *offset* remaining impacts by adhering to Table S5. The main contribution of this study is a landscape scale methodology developed based on key mitigation hierarchy principles, demonstrated by simple spatial analysis using existing data to inform avoidance and offset selection. Additionally, I have shown how an adequate offset size can be calculated based on ecological differences between impact and offset site to facilitate full compensation of unavoidable project losses. My thesis offers managers and decision-makers a basic tool to perform a rapid landscape level assessment of areas to be used in impact mitigation planning, thus providing the background needed to achieve no net loss of biodiversity in wetlands.

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# Appendix A

## Literature review procedure

I conducted a literature review identifying commonly applied principles within each of the four mitigation steps. I defined a principle as a general aspect, method or recommendation proposed by researchers studying the mitigation hierarchy for nature conservation. The purpose of the search was to collect information on how the MH has been applied to nature conservation in general and in relation to wetlands more specifically, which could serve as a scientific basis for developing a more specific wetland mitigation methodology.

I based my literature search procedure on review principles suggested by Xiao and Watson (2019) and Haddaway et al., (2015). Although this approach cannot be considered a "systematic literature review", as defined by most researchers, I built on lessons from such reviews to improve the transparency, objectivity, and repeatability of the search compared to traditional reviews (Haddaway et al., 2015).

I searched through the databases of Google Scholar, Web of Science and Biological science Collection, using combinations of the search words "mitigation hierarchy", "wetland/peatland/mire", "avoidance/avoid", "minimization/minimize/reduction/reduce", "restoration/restore", "offset/offsetting/compensate", "impact(s)/effect(s)", "infrastructure/development" and "wind power".

By reading through the title and abstract of the search results, documents fulfilling the following inclusion criteria were selected for further assessment: 1) related to conservation of biodiversity, 2) related explicitly to at least one of the four mitigation steps, 3) provide suggestions for the practical application of the MH, and 4) if place-specific, focused on an area in the Northern Hemisphere. A focus on wetlands was not chosen as a general inclusion principle, because of few relevant search results. However, there was a sufficient number of studies addressing wetland restoration, and for this MH step I therefore decided to narrow my focus to wetlands. Nonetheless, most principles were in the end expressed in a way that make them applicable to other ecosystems.

I identified MH principles by reading through the selected documents, starting with those most recently published and continuing until reaching a saturation point where no new principles were found when reading through new papers. At last, the principles were systematized into larger categories.

## Literature review results

I identified 66 different MH principles in total. 6 principles were related to the overall framework (Table S1), 13 principles were related to avoidance (Table S2), 8 principles to minimization (Table S3), 11 to restoration (Table S4) and 27 to offsets (Table S5). That most principles were found for offsetting, mirrors that the majority of the search result papers focused on this particular step of the hierarchy.

The identified principles were organized into MH steps according to how they were classified in the reference document. As such, some principles theoretically applicable to several mitigation steps, may sometimes only be listed under one of them. Also, some principles with similar meanings are listed under several mitigation steps, although I have tried to reduce overlaps to a minimum.

### Principles applicable to the MH framework as a whole

***Table S1***. *Identified principles applicable to the mitigation hierarchy as a whole.*

| Main principle | Specific principle | References |
|---|---|---|
| **Relevance** | Make use of relevant datasets | Ekstrom et al., 2015 |
| | Make use of maps and spatial information | Ekstrom et al., 2015 |
| **Targets and indicators** | Decide on a mitigation goal, quantitative targets, and biodiversity indicators | Kiesecker et al., 2010; Butchart et al., 2015; Arlidge et al., 2018; BBOP, 2018; |
| **Monitoring strategies** | Monitoring basic performance of staff and contractors | Ekstrom et al., 2015 |
| | Monitoring and evaluate the implementation, progress, and success of management plans | Rubec et al., 2009; Ekstrom et al., 2015; Phalan et al., 2018 |
| **Consultation** | Consult relevant stakeholders, experts and specialists for their advice and opinion | Ekstrom et al 2015; Arlidge et al., 2018; BBOP 2018; Phalan et al., 2018; |

For the mitigation hierarchy as a whole, I found six specific principles, which I grouped into four overall principles (Table S1). Characteristic for the mitigation framework is the importance of including spatial information (Ekstrom et al., 2015), and collect data on a broad range of topics, stretching from species distributions to stakeholder interests (Ekstrom et al. 2015; BBOP, 2018). The importance of deciding on overall conservation targets was emphasized by many (Kiesecker et al., 2010; Butchart et al., 2015; Arlidge et al., 2018; BBOP, 2018;), as such goals serve to guide the rest of the mitigation implementation.

## Principles to avoid impacts

***Table S2***. *Principles important for avoiding impacts, as identified in the literature review.*

| Main principle | Specific principle | References |
|---|---|---|
| **Avoid priority avoidance areas (PAA)** | Identify and minimize overlap between priority avoidance areas (PAA) and development site. | Bigard et al., 2020; Jones et al., 2022 |
| | PAA: threatened species and ecosystems | Arlidge et al., 2018; Jones et al., 2022; |
| | PAA: high-biodiversity areas | Arlidge et al., 2018; Jones et al., 2022 |
| | PAA: existing protected areas | Arlidge et al., 2018; Jones et al., 2022 |
| | PAA: areas in good ecological condition (i.e., areas with historically intact flora and fauna) | Arlidge et al., 2018; Jones et al., 2022 |
| **Assess potential impacts** | Consider direct and indirect cumulative impacts. | BBOP, 2012; Ekstrom et al., 2015; Phalan et al., 2018; |
| | Assess both environmental and social impacts | BBOP, 2012; Ekstrom et al., 2015; Phalan et al., 2018; |
| **Select avoidance strategy** | Alt. 1: Avoidance through site selection = spatial avoidance. | Ekstrom et al., 2015; Phalan et al., 2018; Jones et al., 2022 |
| | Alt. 2: Avoidance through project design. Resembles the minimization step of the MH. | Ekstrom et al., 2015; Phalan et al., 2018 |
| | Alt. 3: Avoidance through scheduling = temporal avoidance of especially important time periods (e.g., breeding periods for animals) | Ekstrom et al., 2015; Phalan et al., 2018 |
| | Alt. 4: Project cancellation. Applied when no viable alternatives exist. | Phalan et al., 2018 |
| **Plan on both large and small scale** | Two-phased: broad landscape-level planning + detailed project-level planning | Bigard et al., 2020; Jones et al., 2022 |
| **Start consideration as early as possible** | Start consideration before specific plans for project placement and design exists. | Ekstrom et al., 2015; Phalan et al., 2018 |

Avoidance is considered the most important step of the mitigation hierarchy (Ekstrom et al., 2015; Phalan et al., 2018; Jones et al, 2022), although often neglected in practice (Clare et al. 2011; Phalan et al. 2018). I identified 13 principles describing this MH step, and arranged them into five major groups (Table S2).

The main goal of the avoidance-step is to avoid impacting the ecologically most valuable areas – often referred to as priority avoidance areas (Bigard et al. 2020; Jones et al 2022). However, I found a large variation in the specific type of sites that are considered of special importance (Table S2).

I also found that avoidance can be achieved in four different ways, referred to as "avoidance strategies" in Table S2. While spatial avoidance and project cancellation is acknowledged as the most effective way of avoiding impacts altogether (Ekstrom et al., 2015; Phalan et al., 2018; Jones et al, 2022), temporal avoidance can provide an alternative to reduce pressure on wildlife or plants during their most vulnerable life stages, such as breeding periods or flooding events (Ekstrom et al 2015; Phalan et al. 2018).

## Principles to minimize impacts.

***Table S3**. Principles important for* minimizing impacts*, as identified in the literature review.*

| Main principle | Specific principle | References |
|---|---|---|
| **Minimize development area** | Plan for a small development area | Jones et al., 2022. |
| | Incentivize the use of small development areas. E.g., by setting high offset requirements. | Arlidge et al., 2018 |
| **Minimize impact on development area** | Develop minimization measures targeted specific ecosystems, species and infrastructure | Ekstrom et al., 2015 |
| | Make use of scientific best-practices and new technology | Ekstrom et al., 2015 |
| **Consider project alternatives** | Assess if the project can be changed in time, space, or design to lower impacts. (Similar to the avoidance-step) | Ekstrom et al., 2015 |
| **Consult experts** | Consult relevant experts to predict what impacts cannot be avoided and how these can be minimized. | Ekstrom et al., 2015 |
| **Set formal development requirements** | Establish certification requirements as a prerequisite for construction work in wetlands. E.g., specific courses, education, or documented competencies. | Arlidge et al., 2018 |
| | Require developers to submit a "minimization proposal" for approval by local authority before project start-up. Should include all measures taken to minimize damage. | Ekstrom et al., 2015 |

For the minimization-step, I identified eight principles and grouped them into five main groups (Table S3). In general, this was the mitigation step where I found the least amount of published literature. The identified minimization measures were formulated either very broadly or highly case specific. Thus, to not lose relevance for general applicability to a range of wetland impacts, my literature search only allowed for the identification of very broad principles.

In practice, minimization of impacts can be achieved in two ways; 1) by minimizing the size of development area (Arlidge et al., 2018; Jones et al., 2022), or 2) by minimizing the impacts on development area. Jones et al. (2022) suggests addressing the former through good planning practices, ensuring that infrastructure elements are placed in groups rather than dispersed over large areas, locating new infrastructure close to already existing ones, or by minimizing the size or length of infrastructure elements. Arlidge et al. (2018) propose to create incentives for developers to adhere to small areas. This can be done through strict legal requirements or putting a high cost on offsets.

The second solution – to minimize impacts on the development area, is suggested solved by making use of best-practices and by looking for innovative solutions and design (Ekstrom et al., 2015). Examples of such innovations include establishing wildlife corridors across roads and bury cables below-ground.

## Principles to restore impacts.

***Table S4.*** *Principles important for restoring impacts, as identified in the literature review.*

| Main principle | Specific principle | References |
|---|---|---|
| **Assess restoration needs and feasibility** | Identify cause of degradation | Schumann and Joosten, 2008 |
| | Identify level of degradation | Schumann and Joosten, 2008 |
| | Assess feasibility of restoration based on level and cause of degradation and economic and social constraints. | Schumann and Joosten, 2008 |
| **Set objectives and decide on measures** | Set restoration objectives | Schumann and Joosten, 2008 |
| | Decide on and implement specific measures based on cause of degradation and restoration objectives. | Schumann and Joosten, 2008 |
| | Restore historic water table. Many wetlands are degraded due to changed hydrology. | Schumann and Joosten, 2008 |
| | Implement measures as early as possible after damage has occurred. Restoration success decreases with time. | Ekstrom et al., 2015 |

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| **Act before damage has occurred** | Anticipate restoration needs before damage has occurred and plan for how to restore to original state. | Ekstrom et al., 2015 |
| | Establish reference point by taking measurements before disturbance to compare state before/after impact. | Ekstrom et al., 2015 |
| | Preserve substrate and genetic material from project site. Topsoil, plants, and seeds can be used to regrow damaged areas once construction work is finished. | Ekstrom et al., 2015 |
| **Use well-tested restoration techniques** | Use well-tested techniques and follow best practices on wetland restoration to maximize likelihood of success. | Ekstrom et al., 2015 |

I identified 11 restoration principles and grouped them into four main categories (Table S4). Most principles were found in the comprehensive work "Global peatland restoration: Manual" by Schumann and Josten (2008) and the handbook "A cross sector guide for implementing the Mitigation Hierarchy" by Ekstrom et al. (2015). Among the perspectives outlined by Schumann and Joosten (2008) is the importance of assessing the cause and level of degradation before initiating restoration efforts. The authors emphasize that degradation is a continuum between minor and heavy degradation that - together with the cause of degradation, economic and social constraints - determines restoration *feasibility*. Schumann and Joosten (2008) also propose methods to determine such wetland degradation stages, along with its estimated restoration potential.

Furthermore, the main cause of wetland degradation and biodiversity loss is human land-use changes causing habitat loss (MEA, 2005; IPBES, 2019; Ballut-Dajud et al., 2022). Such impacts typically reduce wetland quality by affecting wetland hydrology (Schumann and Joosten, 2008; NEA, 2020), and wetland restoration efforts should thus concentrate on restoring the ecosystem's natural (i.e., historic) water flow (Schumann and Joosten, 2008; NEA, 2020).

## Principles to offset impacts.

***Table S5**. Principles important for offsetting impacts, as identified in the literature review.*

| Main principle | Specific principle | References |
|---|---|---|
| **Only use as a last resort** | Only use offsets to compensate for residual impacts after efforts have been made to avoid, minimize, and restore as much as possible. | Kiesecker et al., 2010; Ekstrom et al., 2015; Arlidge et al., 2018; BBOP, 2018; Jones et al., 2022 |
| | Establish limits to what can be offset, and thus, what can be impacted. Some areas may be declared as out-of-limit for offsetting due to their superior value or species irreplaceability. | Arlidge et al., 2018; BBOP, 2018 |
| **Develop offset goals** | Determine what kind of impacts should require offsetting; either all impacts or only impacts above a certain threshold. Threshold can be based on the amount of area affected or the ecosystem type or threat-level of affected species. | Ekstrom et al., 2015 |
| | Decide on overall offset target. It normally takes the form of NNL of biodiversity or area neutrality (high ambition goals) or NNL of certain threatened species, critical habitat, and/or areas of high conservation value (low ambition goals). | Ekstrom et al., 2015 |
| **Select offset site based on impact site characteristics** | Ensure ecological equivalence: offset site should be ecologically similar to impact site to ensure offset gains accrue to the same ecological entities (e.g., species) as what is lost at impact site. | Kiesecker et al., 2010; McKenney & Kiesecker, 2010; Quétier and Lavorel, 2011; Ekstrom et al., 2015; Arlidge et al., 2018; Bigard et al., 2020; Jones et al., 2022; |
| | Proximity to impact area. This ensures that benefits accrue to affected area. | Kiesecker et al., 2010; Bigard et al., 2020 |
| **Ensure additionality** | The offset should come in addition to conservation measures that would be implemented regardless of development impact. | Kiesecker et al., 2010; Ekstrom et al., 2015; Arlidge et al., 2018; BBOP, 2018 |
| **Avoid time lag** | Reduce delay between impact occurrence and offset results to a minimum. | Arlidge et al., 2018 |
| **Determine offset size requirements** | Establish exchange rules (multipliers) to convert from impact area size to offset area size. Consider differences in ecological value and realistic offset biodiversity gains relative to impact biodiversity losses. | Stewart et al., 1996; Arlidge et al., 2018; Jones et al., 2022; |
| | Quantify residual impacts after efforts have been made to avoid, minimize, and restore damage. | Ekstrom et al., 2015 |
| **Consider negotiation time** | Prioritize offsets in areas already identified as priority conservation sites to help achieve national conservation | Ekstrom et al., 2015; Jones et al., 2022 |

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| | targets and reduce negotiation time and cost. | |
| | Consider number and type of landowners. It is easiest to make offsets on areas belonging to the public or few private landowners. | Bigard et al., 2020 |
| **Maximize offset longevity** | Consider potential impacts of climate change to offset site. Offset should ideally be placed in areas that may act as climate refugia, and at least not areas losing relevance in a warmer climate. | Jones et al., 2022 |
| | For restoration: Proximity to high-biodiversity areas. Important to maximize species recolonization rate. | Hodgson et al., 2011; Quetier et al., 2014; Bigard et al., 2020 |
| | Increase habitat connectivity. Helpful for species on the move due to climate change and to compensate for decreased connectivity due to global habitat loss. | Jones et al., 2022 |
| | Protect from future human damage. The positive effects of the offset should last at least as long as the negative effects of the development. | Kiesecker et al., 2010; Ekstrom et al., 2015, Arlidge et al., 2018; BBOP, 2018; |
| **Ensure environmental justice** | Transparency in offset site selection process. Design and implementation measures should be clear, and results should be communicated to the public | BBOP, 2018 |
| | Involve stakeholders. Assess how an offset will affect stakeholders and how to include them in the decision-making process. | Ekstrom et al., 2015; Jones et al., 2022; BBOP, 2018 |
| **Base methods on existing knowledge** | Build on scientific findings and principles. | BBOP, 2018 |
| | Build on traditional and local knowledge. | BBOP, 2018 |
| **Monitor conservation success** | Monitor success by using biodiversity indicators. It is recommended to use multiple and compound indicators for measuring change in biodiversity over time. | Schumann et al., 2008; Arlidge et al., 2018 |
| **Plan on a landscape scale** | Offset site should be selected on a large scale to consider all options and maximize benefits. | Kiesecker, 2009; BBOP, 2018; Bigard et al., 2020. |
| **Select offset type** | Alt. 1: Legal protection of an ecologically important site. I.e., by establishing a new protected area. Additional criteria applied by some: sites that without protection would be lost (“averted loss” offsets) | Ekstrom et al., 2015; Jones et al., 2022; |

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| | Alt. 2: Restoration of a degraded site. Degraded habitats and ecosystems are restored to its original state. | Ekstrom et al., 2015¸ Jones et al., 2022; |
| | Alt 3: Creation of a new valuable ecosystem in a location where it has not been before. E.g., by creating a wetland on a farmland. | Ekstrom et al., 2015 |
| **Decide on offset implementation strategy** | Direct: developer is responsible for conducting the offset. | Vaissière and Levrel, 2015 |
| | Indirect: developer pays a fee to a conservation fund or external company who overtake the responsibility for conducting the offset (also known as "mitigation banking", "nature fee" or "in lieu fee"). | Vaissière and Levrel, 2015 |

I identified 27 offset principles in the literature and arranged them into 14 broader categories (Table S5). I found that the most fundamental concept of offsetting is *ecological equivalence* (McKenney & Kiesecker, 2010; Quétier & Lavorel, 2011), which refers to ensuring that the same ecological properties and values lost at the impact site is gained at the offset site (Queter and Lavorel, 2011; Jones et al. 2022;). The exact ecological entities considered in this context vary from species and ecosystems to ecological condition and the potential of the site to fully compensate for impact losses (Quetier and Lavorel, 2011; BBOP, 2012; Jones et al. 2022).

# Appendix B

***Table S6.*** *The wetland plant species included in calculation of wetland conservation value.*

| Species name (scientific) | Species name (Norwegian) | Red List status 2021 |
|---|---|---|
| *Agrostis canina* | hundekvein | LC |
| *Alchemilla glabra* | glattmarikåpe | LC |
| *Alchemilla glomerulans* | kildemarikåpe | LC |
| *Alnus glutinosa* | svartor | LC |
| *Alnus incana* | gråor | LC |
| *Alopecurus geniculatus* | knereverumpe | LC |
| *Andromeda polifolia* | hvitlyng | LC |
| *Anemone nemorosa* | hvitveis | LC |
| *Aneura pinguis* | fettmose | LC |
| *Angelica sylvestris* | sløke | LC |
| *Anthelia julacea* | ranksnømose | LC |
| *Anthelia juratzkana* | krypsnømose | LC |
| *Arabis alpina* | fjellskrinneblom | LC |
| *Athyrium filix-femina* | skogburkne | LC |
| *Aulacomnium palustre* | myrfiltmose | LC |
| *Bartsia alpina* | svarttopp | LC |
| *Betula nana* | risbjørk | LC |
| *Betula pubescens* | bjørk | LC |
| *Bistorta vivipara* | harerug | LC |
| *Blepharostoma trichophyllum* | piggtrådmose | LC |
| *Blindia acuta* | rødmesigmose | LC |
| *Botrychium boreale* | fjellmarinøkkel | LC |
| *Botrychium lunaria* | marinøkkel | LC |
| *Brachythecium rivulare* | sumplundmose | LC |
| *Brachythecium rutabulum* | storlundmose | LC |
| *Calamagrostis canescens* | vassrørkvein | LC |
| *Calamagrostis neglecta* | smårørkvein | LC |
| *Calliergon cordifolium* | pjusktjernmose | LC |
| *Calliergon giganteum* | stauttjernmose | LC |
| *Calliergonella cuspidata* | sumpbroddmose | LC |
| *Calluna vulgaris* | røsslyng | LC |
| *Caltha palustris* | bekkeblom | LC |
| *Campylium stellatum* | myrstjernemose | LC |
| *Cardamine amara* | bekkekarse | LC |
| *Cardamine nymanii* | polarkarse | LC |
| *Cardamine pratensis* | engkarse | LC |
| *Carex atrofusca* | sotstarr | LC |
| *Carex bigelowii* | stivstarr | LC |
| *Carex buxbaumii* | klubbestarr | LC |
| *Carex canescens* | gråstarr | LC |

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|---|---|---|
| *Carex capillaris* | hårstarr | NT |
| *Carex capitata* | hodestarr | LC |
| *Carex chordorrhiza* | strengstarr | LC |
| *Carex demissa* | grønnstarr | LC |
| *Carex dioica* | særbustarr | LC |
| *Carex echinata* | stjernestarr | LC |
| *Carex elongata* | langstarr | LC |
| *Carex flava* | gulstarr | LC |
| *Carex glareosa* | grusstarr | LC |
| *Carex globularis* | granstarr | LC |
| *Carex hostiana* | engstarr | LC |
| *Carex lachenalii* | rypestarr | LC |
| *Carex lasiocarpa* | trådstarr | LC |
| *Carex lepidocarpa* | nebbstarr | NT |
| *Carex livida* | blystarr | LC |
| *Carex loliacea* | nubbestarr | NT |
| *Carex mackenziei* | pølstarr | LC |
| *Carex maritima* | buestarr | LC |
| *Carex nigra* | småstarr | LC |
| *Carex panicea* | kornstarr | LC |
| *Carex pauciflora* | sveltstarr | LC |
| *Carex pseudocyperus* | dronningstarr | NT |
| *Carex pulicaris* | loppestarr | LC |
| *Carex rariflora* | snipestarr | LC |
| *Carex remota* | slakkstarr | LC |
| *Carex rostrata* | flaskestarr | LC |
| *Carex rufina* | jøkelstarr | VU |
| *Carex salina* | fjærestarr | LC |
| *Carex subspathacea* | ishavsstarr | LC |
| *Carex vaginata* | slirestarr | LC |
| *Carex vesicaria* | sennegras | LC |
| *Cerastium cerastoides* | brearve | LC |
| *Cetraria islandica* | islandslav | LC |
| *Cetrariella delisei* | snøskjerpe | NT |
| *Chrysosplenium alternifolium* | maigull | LC |
| *Cinclidium stygium* | myrgittermose | LC |
| *Cirsium heterophyllum* | hvitbladtistel | LC |
| *Cirsium oleraceum* | kåltistel | VU |
| *Cirsium palustre* | myrtistel | LC |
| *Cladonia ecmocyna* | snøsyl | LC |
| *Climacium dendroides* | palmemose | LC |
| *Comarum palustre* | myrhatt | LC |
| *Conostomum tetragonum* | hjelmmose | VU |
| *Corallorhiza trifida* | korallrot | LC |
| *Cratoneuron filicinum* | kalkmose | LC |
| *Crepis paludosa* | sumphaukeskjegg | LC |

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|---|---|---|
| *Dactylorhiza incarnata* | engmarihand | LC |
| *Dactylorhiza maculata* | blekmarihand | LC |
| *Dactylorhiza majalis* | kongsmarihand | LC |
| *Deschampsia cespitosa* | kvassbunke | LC |
| *Distichium capillaceum* | puteplanmose | LC |
| *Draba alpina* | gullrublom | LC |
| *Drosera anglica* | smalsoldogg | LC |
| *Drosera rotundifolia* | rundsoldogg | LC |
| *Eleocharis quinqueflora* | småsivaks | LC |
| *Eleocharis uniglumis* | fjæresivaks | LC |
| *Empetrum nigrum* | krekling | LC |
| *Epilobium alsinifolium* | kildemjølke | LC |
| *Epilobium anagallidifolium* | dvergmjølke | LC |
| *Epilobium hornemannii* | setermjølke | LC |
| *Epilobium lactiflorum* | hvitmjølke | LC |
| *Epilobium palustre* | myrmjølke | LC |
| *Equisetum arvense* | åkersnelle | LC |
| *Equisetum fluviatile* | elvesnelle | LC |
| *Equisetum palustre* | myrsnelle | LC |
| *Equisetum pratense* | engsnelle | LC |
| *Equisetum sylvaticum* | skogsnelle | LC |
| *Equisetum variegatum* | fjellsnelle | LC |
| *Erica tetralix* | klokkelyng | LC |
| *Eriophorum angustifolium* | duskull | LC |
| *Eriophorum latifolium* | breiull | LC |
| *Eriophorum scheuchzeri* | snøull | LC |
| *Eriophorum vaginatum* | torvull | LC |
| *Euphrasia wettsteinii* | småøyentrøst | EN |
| *Festuca rubra* | rødsvingel | LC |
| *Filipendula ulmaria* | mjødurt | LC |
| *Fissidens adianthoides* | saglommemose | LC |
| *Frangula alnus* | trollhegg | LC |
| *Fuscocephaloziopsis albescens* | bremose | NT |
| *Galeopsis bifida* | vrangdå | LC |
| *Galium aparine* | klengemaure | LC |
| *Galium palustre* | myrmaure | LC |
| *Galium trifidum* | dvergmaure | LC |
| *Galium uliginosum* | sumpmaure | LC |
| *Geum rivale* | enghumleblom | LC |
| *Glyceria fluitans* | mannasøtgras | LC |
| *Glyceria lithuanica* | skogsøtgras | VU |
| *Gymnadenia conopsea* | engbrudespore | LC |
| *Gymnocolea inflata* | torvdymose | LC |
| *Harpanthus flotovianus* | kildesalmose | LC |
| *Humulus lupulus* | humle | LC |
| *Hylocomium splendens* | etasjemose | LC |

| | | |
|---|---|---|
| *Iris pseudacorus* | sverdlilje | LC |
| *Juncus articulatus* | ryllsiv | LC |
| *Juncus biglumis* | tvillingsiv | NT |
| *Juncus castaneus* | kastanjesiv | LC |
| *Juncus conglomeratus* | knappsiv | LC |
| *Juncus effusus* | lyssiv | LC |
| *Juncus filiformis* | trådsiv | LC |
| *Juncus stygius* | nøkkesiv | LC |
| *Juncus triglumis* | trillingsiv | LC |
| *Kiaeria starkei* | snøfrostmose | NT |
| *Koenigia islandica* | dvergsyre | LC |
| *Loeskypnum badium* | messingmose | LC |
| *Lysimachia europaea* | skogstjerne | LC |
| *Lysimachia thyrsiflora* | gulldusk | LC |
| *Lysimachia vulgaris* | fredløs | LC |
| *Lythrum salicaria* | kattehale | LC |
| *Malaxis monophyllos* | knottblom | EN |
| *Marchantia quadrata* | skjøtmose | LC |
| *Meesia uliginosa* | nervesvanemose | LC |
| *Melampyrum pratense* | stormarimjelle | LC |
| *Mentha arvensis* | åkermynte | LC |
| *Menyanthes trifoliata* | bukkeblad | LC |
| *Mesoptychia bantriensis* | kildeflik | LC |
| *Micranthes stellaris* | stjernesildre | LC |
| *Micranthes tenuis* | grannsildre | LC |
| *Mnium hornum* | kysttornemose | LC |
| *Molinia caerulea* | blåtopp | LC |
| *Montia fontana* | kildeurt | LC |
| *Myosotis laxa* | sumpforglemmegei | LC |
| *Myosotis scorpioides* | engforglemmegei | LC |
| *Myrica gale* | pors | LC |
| *Narthecium ossifragum* | rome | LC |
| *Neottia ovata* | stortveblad | LC |
| *Oligotrichum hercynicum* | grusmose | LC |
| *Omalotheca norvegica* | setergråurt | LC |
| *Omalotheca supina* | dverggråurt | LC |
| *Oxycoccus microcarpus* | småtranebær | LC |
| *Oxyria digyna* | fjellsyre | LC |
| *Paludella squarrosa* | piperensermose | LC |
| *Parnassia palustris* | jåblom | LC |
| *Pedicularis oederi* | gullmyrklegg | LC |
| *Pedicularis palustris* | myrklegg | LC |
| *Petasites frigidus* | fjellpestrot | LC |
| *Phalaris arundinacea* | strandrør | LC |
| *Phegopteris connectilis* | hengeving | LC |
| *Philonotis fontana* | teppekildemose | LC |
| *Phippsia algida* | snøgras | VU |

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|---|---|---|
| *Phippsia concinna* | sprikesnøgras | NT |
| *Phleum alpinum* | fjelltimotei | LC |
| *Phragmites australis* | takrør | LC |
| *Picea abies* | gran | LC |
| *Pinguicula vulgaris* | tettegras | LC |
| *Pinus sylvestris* | furu | LC |
| *Plagiomnium ellipticum* | sumpfagermose | LC |
| *Plantago maritima* | strandkjempe | LC |
| *Pleurozium schreberi* | furumose | LC |
| *Poa alpina* | fjellrapp | LC |
| *Poa trivialis* | markrapp | LC |
| *Pohlia drummondii* | rødknoppnikke | LC |
| *Pohlia wahlenbergii* | kaldnikke | LC |
| *Polytrichastrum sexangulare* | snøbinnemose | VU |
| *Polytrichum commune* | storbjørnemose | LC |
| *Potentilla erecta* | tepperot | LC |
| *Prunella vulgaris* | blåkoll | LC |
| *Prunus padus* | hegg | LC |
| *Racomitrium lanuginosum* | heigråmose | LC |
| *Ranunculus glacialis* | issoleie | NT |
| *Ranunculus nivalis* | snøsoleie | VU |
| *Ranunculus pygmaeus* | dvergsoleie | LC |
| *Ranunculus repens* | krypsoleie | LC |
| *Rhizomnium punctatum* | bekkerundmose | LC |
| *Rhododendron tomentosum* | finnmarkspors | LC |
| *Rhynchospora alba* | hvitmyrak | LC |
| *Rubus chamaemorus* | molte | VU |
| *Sagina nivalis* | jøkelarve | LC |
| *Sagina saginoides* | seterarve | LC |
| *Salix aurita* | ørevier | LC |
| *Salix cinerea* | gråselje | LC |
| *Salix glauca* | myrvier | LC |
| *Salix hastata* | bleikvier | LC |
| *Salix herbacea* | musøre | LC |
| *Salix lapponum* | lappvier | LC |
| *Salix myrsinifolia* | storvier | LC |
| *Salix myrsinites* | myrtevier | LC |
| *Salix pentandra* | istervier | LC |
| *Salix phylicifolia* | grønnvier | LC |
| *Salix polaris* | polarvier | NT |
| *Sanionia uncinata* | klobleikmose | LC |
| *Sarmentypnum exannulatum* | vrangnøkkemose | LC |
| *Sarmentypnum sarmentosum* | blodnøkkemose | LC |
| *Saussurea alpina* | fjelltistel | LC |
| *Saxifraga aizoides* | gulsildre | LC |
| *Saxifraga cernua* | knoppsildre | NT |
| *Saxifraga oppositifolia* | rødsildre | NT |

| | | |
|---|---|---|
| *Saxifraga rivularis* | bekkesildre | LC |
| *Scapania uliginosa* | kildetvebladmose | LC |
| *Scapania undulata* | bekketvebladmose | LC |
| *Scheuchzeria palustris* | sivblom | LC |
| *Schoenus ferrugineus* | brunskjene | VU |
| *Scorpidium cossonii* | brunmakkmose | LC |
| *Scorpidium revolvens* | rødmakkmose | LC |
| *Scorpidium scorpioides* | stormakkmose | LC |
| *Scorzoneroides autumnalis* | føllblom | LC |
| *Scutellaria galericulata* | skjoldbærer | LC |
| *Selaginella selaginoides* | dvergjamne | LC |
| *Sibbaldia procumbens* | trefingerurt | LC |
| *Silene acaulis* | fjellsmelle | LC |
| *Solanum dulcamara* | slyngsøtvier | LC |
| *Solorina crocea* | safranlav | LC |
| *Sorbus aucuparia* | rogn | LC |
| *Stachys palustris* | åkersvinerot | LC |
| *Stellaria crassifolia* | saftstjerneblom | LC |
| *Stellaria palustris* | myrstjerneblom | VU |
| *Straminergon stramineum* | grasmose | LC |
| *Succisa pratensis* | blåknapp | LC |
| *Tayloria lingulata* | myrtrompetmose | LC |
| *Thalictrum alpinum* | fjellfrøstjerne | LC |
| *Tofieldia pusilla* | bjørnebrodd | NT |
| *Tomentypnum nitens* | gullmose | LC |
| *Trichophorum alpinum* | sveltull | LC |
| *Trichophorum cespitosum* | bjørneskjegg | LC |
| *Triglochin maritima* | fjæresauløk | LC |
| *Triglochin palustris* | myrsauløk | LC |
| *Tussilago farfara* | hestehov | LC |
| *Urtica dioica* | stornesle | LC |
| *Vaccinium myrtillus* | blåbær | LC |
| *Vaccinium uliginosum* | blokkebær | CR |
| *Vaccinium vitis-idaea* | tyttebær | LC |
| *Valeriana sambucifolia* | vendelrot | LC |
| *Veronica beccabunga* | bekkeveronika | LC |
| *Viola epipsila* | stor myrfiol | LC |
| *Viola palustris* | myrfiol | LC |

# Appendix C

***Table S7**. Groups of variables used in the habitat suitability modeling to calculate the GEP value for each mapping unit.*

| Data | Description | Source |
|---|---|---|
| **Bioclimatic variables** | Annual mean temperature<br>Mean diurnal temperature range<br>Isothermality<br>Temperature seasonality<br>Maximum temperature<br>Minimum temperature<br>Annual temperature range<br>Mean temperature in the wettest quarter<br>Mean temperature in the driest quarter<br>Mean temperature in the coldest quarter<br>Mean temperature in the warmest quarter<br>Total annual precipitation<br>Precipitation in the wettest quarter<br>Precipitation in the driest quarter<br>Precipitation variability<br>Precipitation in the wettest quarter<br>Precipitation in the driest quarter<br>Precipitation in the warmest quarter<br>Precipitation in the coldest quarter | Hijmans, R. J., Cameron, S. E., Parra, J. L., Jones, P. G., & Jarvis, A. (2005). Very high resolution interpolated climate surfaces for global land areas. *International Journal of Climatology*, *25*(15), 1965–1978. https://doi.org/10.1002/joc.1276 |
| **Altitude** | Norwegian DEM | Norwegian mapping authority (Kartverket.no), DTM 10 Terrengmodell |
| **Distance to coast** | Calculated from the Norwegian coastline map | Norwegian mapping authority (Kartverket.no), N250 Data |
| **Distance to fresh water** | Calculated from the Norwegian river inventory | Norwegian mapping authority (Kartverket.no), N250 Data |
| **Bedrock** | Bedrock categories according to the Norwegian Geological Society (NGU) | Norges geologiske undersøkelse. 20-12-2021. https://geo.ngu.no/kart/berggrunn_mobil |
| **Species occurrence data** | GBIF occurrences of human observations of plant species of interest from 1980 to the 10th of December 2021 | GBIF.org. (2021). *GBIF Occurrence Download*. Global Biodiversity Information Facility. https://www.gbif.org/occurrence/download/0065725-200613084148143 |