# Compensatory Wetland Mitigation in North America: Wetland Mitigation Banking as a Form of Greenwashing

Major Research Paper

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# Glossary

| Term/Abbreviation | Definition |
|---|---|
| Hydrogeomorphic Classification | Hydrogeomorphic classification approach assumes that wetlands of different types will function differently. This approach facilitates the comparison of mitigation and natural wetlands within a functional context at a landscape level. |

# Abbreviations and Acronyms

| Abbreviation/Acronym | Meaning |
|---|---|
| CRAM | California Rapid Assessment Method |
| CWM | Compensatory Wetland Mitigation |
| HGM | Hydrogeomorphic Method |
| ILF | In-Lieu Fee |
| PRM | Permittee-Responsible Mitigation |
| WMB | Wetland Mitigation Banking |

# 1.0 Introduction

Our planet is on the brink of a sixth mass extinction, with current extinction rates exceeding those expected from the fossil record (Barnosky *et al.* 2011). The primary reason behind the current biodiversity crisis is widespread destruction of natural habitats (Koh *et al.* 2004), which is fragmenting the land leading to genetic and evolutionary consequences (Hanski, 2011). There is an urgent need for effective conservation of biodiversity hotspots. Teaming with diverse biotic life, wetlands are recognized as biodiversity hotspots (Dertien *et al.* 2020; Keddy *et al.* 2009). These ecosystems have been identified as one of the most productive ecosystem types (Salimi *et al.,* 2021), recognized globally as being important for both wildlife and human productivity (Horwitz *et al.,* 2012).

Wetlands serve as vital ecosystems by performing physical, chemical, and biological processes that provide essential goods and services, ultimately supporting basic life necessities (Bond *et al.* 1992). Their shallow depths, nutrient levels, and primary productivity make wetlands ideal ecosystems for developing the base of the food web. Wetland ecosystems improve water quality, provide habitat for an immense variety of species and protection against coastal erosion and floods, maintain the global water and nutrient cycles, and moderate the global climate through carbon sequestration (EPA, 2024b; EPA, 2023; NCC, 2023). Additionally, wetlands provide us with medicines, opportunities for recreation (e.g., bird watching), and aesthetically pleasing environments (EPA, 2024b).

Despite the significance of wetlands in maintaining biodiversity and human well-being, nearly 35% of the world's wetlands have vanished since the 1970s, primarily due to land-use changes. Accordingly, there has been a legislative and regulatory shift towards wetland conservation (OECD, 2016). Wetland degradation is still ongoing and widespread, however, sustaining the loss of critical wetland ecosystem services. Failing to conserve the world's remaining wetlands could jeopardize the attainment of the United Nations' Sustainable Development Goals, especially those related to climate change, biodiversity, and disaster risk reduction. Policymakers must strike a balance between economic development and the preservation of these critical ecosystems (Convention on Wetlands, 2021).

## 1.1 Relevant Wetland Conservation Legislation in North America

The first legislation for protecting against further wetland loss was passed in the 1970s, once society began to shift away from the dominant historical narrative of wetlands as "wastelands" that become valuable when drained (Cox & Grose, 2000), towards an increased understanding of how human well-being is connected to biodiversity and ecosystem functions (Lele *et al.* 2013; OECD, 2016). This first wetland conservation legislation was passed in the United States of America (USA), with the *Clean Water Act (CWA),* 1972, making it illegal for anyone to discharge dredged or fill materials into most wetlands in the USA (OECD, 2016).

In 1989, a goal of No Net Loss (NNL) of wetland area and function was established at both the federal and state levels in the USA under Section 404 of the CWA (EPA, 2024). This section of the CWA established a regulatory program requiring developers to secure a permit for any unavoidable wetland losses and provide the appropriate compensation (i.e., offset the loss of wetland area and function caused by their development projects) (EPA, 2024). This form of offsetting is known as Compensatory Wetland Mitigation (CWM) (Ambrose, 2000), and, in the USA, is administered by the Army Corps of Engineers (ACE) in cooperation with the Environmental Protection Agency (EPA) (EPA, 2024).

The goal of NNL became one of the most commonly employed tools for offsetting environmental impacts (Ermgassen *et al.* 2019), not only across the USA but across North

America (Cox & Grose, 2000). In Canada, the *Federal Policy on Wetland Conservation* (FPWC) was mandated in 1991, which aimed to sustain the ecological and socio-economic functions of wetlands now and in the future (ECCC, 2023). The FPWC aligns with the USA's NNL policy under Section 404 of the Clean Water Act. This Canadian policy takes less of a regulatory approach, however, and relies more heavily on performance objectives. The Canadian Wildlife Service has the responsibility of offering mitigation advice, and the FPWC is merely a recommendation, as the federal government lacks enforcement capabilities beyond its jurisdictional authority. Nonetheless, several jurisdictions across Canada have implemented the goal of NNL in policy with the understanding that compensation, if used, should replace lost wetland functions (Rubec & Hanson, 2009).

The NNL policies in both the USA and Canada force developers to consider alternative projects by following a mitigation hierarchy in a sequential manner with the aim of wetland conversion being an absolute last resort. First, developers must prioritise the avoidance of impacts, taking all possible measures to prevent adverse effects on wetlands. If avoidance proves impossible, they then must seek to minimise impacts by exploring alternative project designs and locations. Finally, developers must compensate or offset for impacts that they prove cannot be minimised or avoided. With this last option in the mandated mitigation hierarchy, developers must offset their impacts to wetland area and function through wetland restoration, creation, and/or enhancement (Bennett *et al.,* 2017; Cox & Grose, 2000; OECD, 2016; Rubec & Hanson, 2009).

## 1.2 Compensatory Wetland Mitigation

There exist three mechanisms for which CWM may be implemented in the USA, including Permittee-Responsible Mitigation (PRM), In-Lieu Fee mitigation (ILF), and Wetland

Mitigation Banking (WMB) (Refer to Figure 1 below for the WMB and PRM process). In the PRM method, mitigation actions are determined and executed on an individual basis. Developers must identify, implement, and monitor compensatory measures to offset their wetland impacts. They submit permit applications that are reviewed and sanctioned by the ACE and EPA to ensure alignment with federal and regional compensation standards. With the ILF approach, multiple permit holders pool fees to a third-party entity. Using these pooled funds, the third-party organization (e.g., conservation organization) or government agency assumes the role of executing several mitigation initiatives. Under the WMB system, "banks" are designated sites intended for wetland restoration, establishment, enhancement, or conservation. Banks generate pooled wetland mitigation credits regulated by the ACE, which determines the credit allocation for each bank based on monitoring data. Bank sponsors are required to monitor banks for at least five years to ensure that the wetlands within them meet the required quality for offsetting losses. Once credits are distributed and sold, bank sponsors are responsible for transferring the property to an organization tasked with its ongoing management (Corps and EPA, 2008; OECD, 2016; Vaissière *et al.,* 2017). While many wetland CWM projects implemented within Canada are under the PRM system, some provinces including Alberta and Québec have considered WMB as an alternative (Rubec & Hanson, 2009).

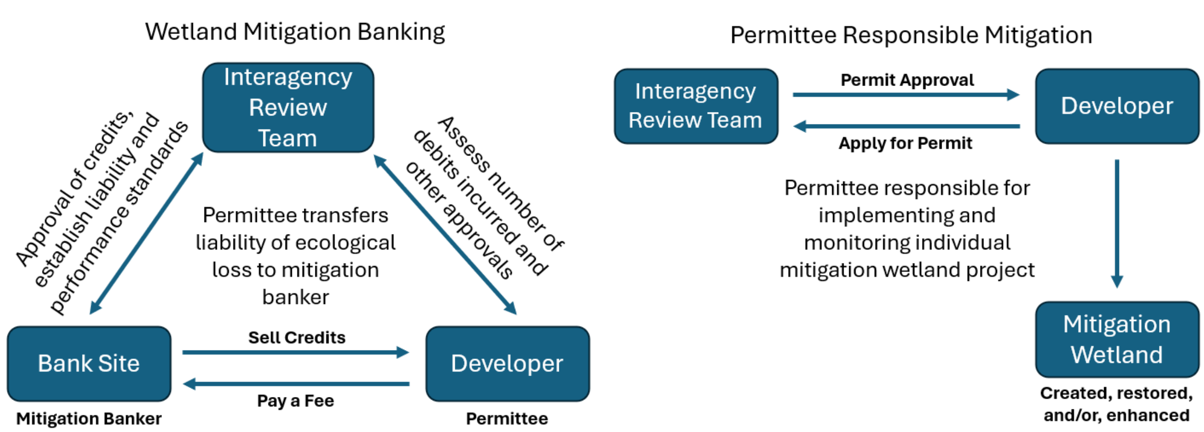


***Figure 1***. The processes of implementing WMB and PRM.

## 1.4 Rationale and Research Questions

Several assumptions underlying the regulatory framework of CWM may hinder the achievement of the goal of NNL. CWM projects are deemed sufficient when they achieve functional equivalence with natural wetlands in the surrounding area. The ACE assumes that sites meeting performance standards will maintain ecological equivalence in the long-term, an assumption that has been challenged by Zedler & Callaway (1999). Moreover, the goal of NNL aims to address both area and functional loss, yet wetland assessments and credits under WMB are often area-based (Ermgassen *et al.* 2019). While the ACE favors functional assessment methods, they are not mandatory. When these methods are not used, a 1:1 area replacement is deemed adequate by the ACE (Corps & EPA, 2008), overlooking the natural variability in biodiversity, habitat quality, and ecosystem services (Kate *et al.*, 2004). Additionally, it is important to consider an assumption regarding the effectiveness of WMB. The CWM strategies have evolved due to downfalls associated with traditional PRM. That is, limited ecological and economic effectiveness as well as low compliance rates (Vaissière *et al.*, 2017). In response, WMB was developed, which aimed for improved cost-effectiveness and compliance monitoring efficiency (EPA, 2023b).

Notably, the advantages of WMB over PRM as recognized by the ACE are purely economic, lacking any focus on the ecological effectiveness of WMB over PRM. Regardless, the ACE has since stated WMB as its preferred mechanism for implementing CWM (Corps & EPA, 2008), resulting in a substantial increase in its adoption (IWR, 2015). There is, however, a concerning level of uncertainty as to whether WMB can actually outperform PRM by creating wetlands within banks that are functionally equivalent to natural wetlands and achieving the broader goal of NNL in practice (Burgin, 2009; Levrel *et al.,* 2017; Mitsch & Hernandez, 2012; OECD, 2016; Zinn, 1997).

There are inherent characteristics within the regulatory framework of WMB that may hinder the achievement of true ecological equivalence and the goal of NNL. In particular, the market-driven nature of WMB could shift the focus from ecological preservation towards merely economic outcomes (Calvet *et al.,* 2015). Additionally, the pooling principle within WMB reduces asset specificity unlike in PRM which has a one-to-one wetland replacement requirement. This pooling principle in WMB may therefore potentially be simplifying wetlands over time, creating more homogeneity across the landscape (Vaissière *et al.*, 2017). A concerning finding was revealed from a simulation study conducted by Tillman & Matthews (2022), whereby even with a 1.5:1 mitigation ratio, WMBs within the Chicago District, USA, failed to replace even half of the native plant species found within the wetlands impacted by development projects. This finding highlights the critical need to assess the ecological outcomes of WMB to ensure it is effective for conserving the remaining wetlands that we depend on.

My study is a qualitative review that will synthesize the existing data on the success and ecological outcomes of wetlands created, restored, or enhanced through WMB within North America to answer the following research questions:

(1) Is the required condition of equivalence under the regulatory framework of WMB for the issuance of mitigation credits being met?

(2) Can the goal of NNL of wetlands be achieved through the current systems of WMB?

If WMB is capable of creating, restoring, or enhancing wetlands to a functionally equivalent level with natural reference wetlands within the surrounding area of development projects, then the current regulatory framework may support the goal of NNL. If most studies are finding that ecological performance standards are not being met, however, with landscape homogeneity increasing over time and functional loss occurring beyond ecologically defined watershed boundaries, then the current regulatory framework may very well fail at curbing the loss of wetlands and their critical functions. The outcomes of WMB should realistically be matching the vegetative structure, and biogeochemical, hydrological, and habitat functions of high-quality natural reference wetlands. Aiming for equivalence with pristine wetlands would not only help ensure that the goal of NNL is actually being achieved in practice, but would help restore the historically lost wetlands, which is crucial in our fight against climate change. If WMB fails to effectively offset wetland function and biodiversity loss continues, then there needs to be greater emphasis placed on avoidance within the mitigation hierarchy. Overall, with this qualitative review, I aim to offer valuable insights for professionals engaged in the execution and oversight of CWM projects, helping to guide future policy development, defining project goals, and managing projects effectively.

# 2.0 Methodology

## 2.1 Sample Selection

The time for completing this review is constrained to one academic semester (four months) and, for this reason, I could not collect primary data. Instead, I reviewed and analysed 30 peer-reviewed studies and grey literature that assess the ecological outcomes (e.g., developmental characteristics, functions, and structure) of wetlands within the USA and Canada’s WMB and PRM systems, comparing them to natural reference wetlands of varying anthropogenic disturbance levels. These reference wetlands will serve as a baseline for determining whether the ecological equivalence and the goal of NNL are achieved. I included studies that conducted comparative analyses using impacted reference wetlands in addition to studies that focus on pristine reference wetlands, given that development projects often damage some area of wetlands rather than completely destroying them. Additionally, I broadened the scope of this review to include wetlands under the PRM system due to limited WMB-specific research.

While this is not a truly comprehensive systematic literature review, I used a systematic sampling method for article selection. This approach prioritises replicability which will allow future researchers to follow the same selection process and potentially generate similar results. Keywords used in search engines such as Google Scholar, Omni, and Web of Science included:

- “Wetland mitigation banking” AND “ecological equivalence”
- “Wetland mitigation banking” AND “no net loss”
- “Wetland mitigation banking” AND “floristic quality”
- “Wetland mitigation banking” AND “soil functions”
- “Wetland mitigation banking” AND “microbial-mediated functions”
- “Wetland mitigation banking” AND “hydrological functions”
- Wetland mitigation banking” AND “carbon sequestration”
- Wetland mitigation banking” AND “habitat”
- “Wetland mitigation banking” AND “biodiversity”
- “Wetland mitigation banking” AND “functional loss”

The search was restricted to English-language articles in the following topic categories:

- Biodiversity outcome variables for assessing losses and gains.
- Landscape context, climate, fragmentation, proximity to land use, and wetland extent.
- Animal and plant abundance, biodiversity, nutrient cycling, hydrology, climatic and biological regulation.

An initial scan of titles and abstracts assessed article relevance, and a subsequent second scan involved a full-text review based on inclusion/exclusion criteria (Refer to Table 1 in Appendix). I sought additional studies fitting the inclusion criteria from the reference lists of articles that fit the criteria during the second scan.

## 2.2 Potential Sources of Bias

Based on the established inclusion/exclusion criteria, sampling biases may arise. Limiting the sample to English-language articles introduces a language bias and reinforces language barriers, limiting opportunities for readers from non-English speaking countries. Ultimately, this hinders the use of scientific information between environmental field practitioners and policy makers between communities, presenting a major barrier to global science (Amano *et al.*, 2016). Future research can address this by providing abstracts written in multiple languages. Moreover, time constraints may present an information bias. In particular, the extensive documentation of the USA's CWM system (OECD, 2016) may cause an overrepresentation of USA-focused studies, and the emphasis on vegetation-based metrics for wetland evaluation (Cole & Shafer, 2002) suggests a potential overrepresentation of articles focusing on vegetative structure and function. These information biases may potentially be addressed with future primary data collection when more time and resources are available.

## 2.3 Ethical Considerations

This research that I will be undertaking under the jurisdiction, or auspices of the University of Ottawa does not involve conducting directed studies that require participation of humans or any form of human biological material. There are no ethical concerns that I need to address, and therefore, I will not need to receive an ethics approval from a University Research Ethics Board.

# 3.0 Results

My review examined 30 studies published between 2003 and 2023. The geographical focus of North America provided 23 studies conducted in the USA and seven in Canada. The included studies encompassed a variety of publication types: one thesis, four state government reports, and the remaining 25 being peer-reviewed scientific papers.

## 3.1 Vegetative Indices and Metrics

Assessing only vegetative structure may not provide a comprehensive representation of wetland function. In Ohio, USA, comparisons between CWM wetlands and natural reference wetlands by Mack & Micacchion (2006), Micacchion *et al.* (2010), and Micacchion (2012) revealed challenges in achieving the state's required "good" ecological quality standards, which are deemed essential for maintaining future environmental resilience. In Mack & Micacchion (2006), only 10% of bank wetlands reached the "good" quality standard. Bank wetlands had lower plant community quality than high-quality reference wetlands, but better than degraded ones. Similar findings were observed in PRM wetlands by Micacchion *et al.* (2010), which generally had lower plant community quality than reference wetlands, with most equivalent to "poor" or "fair" ecological condition. In Lake Erie, Micacchion (2012) found that 78% of natural wetlands were in "good" to "excellent" condition, compared to only 30% of bank wetlands and 13% of PRM wetlands reaching these conditions. Stefanik & Mitsch (2012) found that Ohio bank wetlands had lower plant productivity and diversity than in natural reference wetlands. Similar observations were made in the Chicago District, USA, by Tillman *et al.* (2022) who found that, while the plant communities in bank wetlands did not reach equivalence with the high-quality plant communities of undisturbed natural wetlands, they surpassed the ecological quality of the most degraded natural wetlands that they assessed in the region.

Assessing floristic quality over extended monitoring periods, beyond the typically required 5-years, can provide valuable insights into the successional trajectory of CWM practices, and reveal more nuanced information than simple short-term vegetation metrics. In the Central Parkland ecoregion of Alberta, Salaria *et al.* (2018) found that restored wetlands did not reach equivalence with natural wetlands in terms of plant species richness and community composition. Even after 20 years post-restoration, these wetlands did not reach their full potential. In Ohio, Spieles *et al.* (2006) found that restored and created bank wetlands had lower floristic quality than natural reference wetlands by their fifth year of monitoring post-construction. By the tenth year of monitoring, the floristic quality of created bank wetlands significantly declined, dropping below that of reference wetlands. There were, however, no differences found in species richness, hydrophytic vegetation prevalence, or non-native species presence between bank and reference wetlands at the 10-year mark. Conversely, a study in Illinois, USA, conducted by Van den Bosch & Matthews (2016) found that PRM wetlands had greater floristic quality than reference wetlands in the long-term, but had lower abundance of native perennial species. Collectively, these studies really highlight the need to move beyond using simple vegetation metrics.

The California Rapid Assessment Method (CRAM) assesses the ecological condition of wetlands. Using CRAM, Ambrose *et al.* (2007) compared wetlands in WMB, PRM, and ILF systems in California, USA, to high-quality natural reference sites. The researchers established a CRAM score of 70% or higher as the cutoff criteria for “optimal” wetland condition, with nearly all (89%) high-quality natural reference wetlands having met this optimal condition. On the contrary, CRAM scores for CWM wetlands revealed that only 19% of CWM wetlands achieved this condition. An additional observation of the researchers was an overall net loss of wetland habitat across the state, with wetter areas being replaced by drier riparian and upland habitats.

Beta diversity is a measure that can be used to assess the variation in species composition among different wetland habitats, revealing potential biotic homogenization. In a study conducted in Illinois, Price *et al.* (2019) found that the overall beta diversity between PRM wetlands and both high- and low-quality natural reference sites was similar. However, PRM wetlands showed spatial variation particularly related to hydrology. In drier areas of PRM wetlands, dominant wetland species found in reference wetlands were absent, whereas the wetter zones more closely resembled the reference wetlands. High-quality reference sites had greater species richness and more unique species. PRM sites shared more community traits with the low-quality reference sites, likely due to regional proximity, with plants of low conservation value predominating, such as weedy upland species. The only species that differentiated PRM sites from both reference wetland types was false nettle.

## 3.2 Hydrological Functions

Observations of the extremes of inundation in wetlands indicates that natural hydrological functions are not effectively being restored through CWM practices. Micacchion *et al.* (2010) found that more than half (54%) of PRM wetlands in their study had permanent inundation with deeper water levels compared to their natural referents. Some of the PRM wetlands resembled ponds rather than typical wetlands. Similarly, Austin & Schreiver (2013) found that the created PRM wetlands were permanently flooded, and Hoeltje & Cole (2007), Hossler *et al.,* (2011) and Anderson & Rooney (2019) found hydrological tendencies of created depressional wetlands to be exposed to longer periods of greater inundation than all reference types included in their studies. On the contrary, Swartz *et al.* (2019) and Swartz *et al.* (2019b) observed complete or partial drying of most of the created wetlands that they assessed. Highly concerning was the timing of this drying, having taken place prior to pond-breeding amphibian larvae metamorphosis, making the created wetlands act as ecological traps. Petranka *et al.* (2003) showed how an ecological trap outcome can potentially be avoided by creating wetlands that are deep enough to prevent early drying during warm spells. In their study, bank wetlands were larger and deeper than their referents, maintaining water for a long enough duration for amphibian metamorphosis to take place. Thus, the bank wetlands functioned more as population sinks.

Restoration of the natural hydrological regime is an important goal for CWM projects as it provides the necessary conditions for other functions to be restored; however, it may not be adequate for reaching full ecological equivalence. This was revealed by De Steven *et al.* (2010), who assessed depressional wetlands that were experimentally restored through ditch-plugging in South Carolina, USA. Water depths and durations in the depressional wetlands, which would potentially be used for future bank crediting, resembled natural basins. Despite having established diverse flora, only half of the depressions met the required vegetation performance standards, and natural colonisation by the typical perennial species was limited.

## 3.3 Biogeochemical Functions

The nos*Z* gene can be used as an indicator of wetland microbial-mediated functions, specifically denitrification. Peralta *et al.* (2010) assessed plant community composition and amplified the nos*Z* gene to assess the diversity of microorganisms in restored wetlands within a bank in Illinois and compared such to natural reference floodplain wetlands. The microbial communities in the bank wetlands were significantly different than those observed in the reference wetlands, where the latter exhibited higher levels of soil moisture, total organic matter, soil carbon/nitrogen ratio, and available nitrate compared to bank wetlands. The denitrification potential of the soil microbial community was thus greater in reference wetlands than in restored wetlands.

A reduction in Carbon (C), Nitrogen (N), and Phosphorus (P) cycling within wetlands indicates diminished plant and microbial functions. Mack & Micacchion (2006) found that only three of the bank wetlands that they assessed had N values comparable to or higher than the bottom 25% of N values found within natural reference wetlands. Additionally, most bank wetlands had substantially lower percentages of C and soil organic matter compared to reference wetlands, but reference wetlands had higher concentrations of calcium, potassium, and magnesium compared to bank wetlands. Hossler *et al.* (2011) made C, N, and P amendments to examine potential nutrient limitations influencing plant growth in bank and PRM wetlands in Ohio. The researchers revealed significantly lower stocks of C, N, and P compared to natural wetlands, an observation that occurred despite similarities in hydrology, biotic structure, and nutrient availability between sites. More specifically, the CWM wetlands had 90% less C within litter, 80% less C within soil, 80% less N within litter and soil, 80% less P within litter, and a 40% lower annual cycling through decomposition. Consequently, CWM wetlands demonstrated substantially reduced microbial-mediated functions, including basal respiration, methane production, denitrification, and C and N net mineralization. Finally, Fennessy *et al.* (2008) observed faster rates of decomposition, higher concentrations of soil organic nutrients (C%, %N, plant available P, and soil ammonia), increased levels of plant tissue nutrients, and greater plant biomass production in natural reference wetlands compared to created PRM wetlands in Ohio. PRM wetlands had greater variability in biomass production, were deficient of C and N, and had greater homogeneity when considering nutrient pools.

Analysis of soil organic carbon (SOC) stocks offers valuable insights into the potential rates of methane and nitrous oxide emissions from wetlands. In a study in Alberta, Badiou *et al.* (2011) estimated carbon sequestration rates based on changes in Prairie Pothole wetlands based on changes in SOC stocks between newly restored, long-term restored, and reference wetlands (i.e., wetlands that had never been drained for agriculture). Methane and nitrous oxide emissions, as well as SOC levels, fell within the natural range of variability for both restored wetland types. However, reference wetlands still had higher SOC and greater carbon sequestration, highlighting the long-term impacts of wetland drainage and degradation. Despite these differences, both restored wetland types functioned as carbon sinks rather than sources, reflecting the potential of wetland restoration in this region to mitigate greenhouse gas emissions.

## 3.4 Species-Specific Responses

### 3.4.1 Invertebrate Habitat and Communities

Invertebrate community structure can provide insights into the ecological function of wetlands. In North Carolina, USA, Gianopulos *et al.* (2021) found that re-established and enhanced PRM wetlands had comparable or even greater macroinvertebrate community structure, including density, taxonomic richness, diversity, and evenness than in the reference wetlands. Using Ohio's Rapid Assessment Method showed that PRM wetlands were generally equivalent with reference sites, where the former received "moderate" to "superior" ratings and the latter generally received "superior." In the Greater Yellowstone Ecosystem, Wyoming, USA, Swartz *et al.* (2019), found that created PRM sites had lower taxonomic richness compared to natural and impacted reference wetlands, and had distinct community compositions. A notable finding was that impacted wetlands (i.e., wetlands that only had <25% of their perimeter impacted by construction) resembled natural reference wetlands to a greater extent, with more overlap in terms of physical habitat characteristics, invertebrate richness, and community composition, suggesting that wetland invertebrate communities may have some level of resilience to anthropogenic disturbance.

Differences in invertebrate guild dominance between CWM wetlands and natural reference wetlands indicates differences in community quality. Spieles *et al.* (2006) found no differences in macroinvertebrate taxa richness and diversity between Ohio bank wetlands of 10 years of age and high-quality natural reference wetlands. However, created bank wetlands had significantly smaller herbivore biomass and significantly greater detrivore biomass than reference wetlands, and restored bank wetlands had significantly smaller detrivore biomass than reference wetlands. In West Virginia, USA, Balcombe *et al*., (2005) found overall similarities in familial richness, diversity, biomass, and density in the macroinvertebrate communities in created PRM wetlands compared to natural reference wetlands. However, several differences in abundance and biomass were revealed. Slugs, snails (Gastropoda), and true bugs (Hemiptera) were the two dominant benthic aquatic taxa in PRM wetlands, whereas crustaceans (Isopoda) and flies (Diptera) were predominant in reference wetlands. When assessing only the emergent areas of the wetlands, the PRM wetlands had higher density of bladder snails (Physidae) whereas reference wetlands had higher density of benthic ramshorn snails (Planorbidae). When assessing the entire wetland complexes, there was higher crustacean (Asellidae) density and biomass in the reference wetlands compared to the PRM wetlands.

### 3.4.2 Avian Habitat and Communities

Avian communities can serve as important indicators of wetland ecosystem health. In Southern New Hampshire, USA, McKown *et al.,* (2021) conducted three floristic surveys over 28 years to determine the long-term successional trajectory and suitability as wetland bird habitat of a created PRM wetland of 35 years of age. Increasing wetland habitat heterogeneity was observed, which supported specialised flora (Tussocks Sedge) that acted as biodiversity hotspots. The plants were 90% native at the 28-year mark. Found hiding in the cattail marsh, which had expanded substantially since the initial floristic survey, were the secretive species Virginia Rail and the state-listed species of Special Concern, Carolina Rail. Overall, the habitat diversity supported a highly varied avian community with waterfowl making up 45% of the observed species. The reintroduction of beavers allowed sedge meadow marsh and red maple swamp to emerge 20 years post-construction, which actually increased the wetland complex's area by 0.09 ha. Although the original 4 ha of scrub-shrub wetland was not fully replaced by the construction of this PRM wetland, McKown *et al.* (2021) concluded that the mitigation project nevertheless successfully established a self-sustaining, biodiverse wetland.

The avian community composition in wetlands restored outside of CWM systems can still offer insights into the potential efficacy of restoration as a strategy within these systems. In the Parkland Region of Alberta, Anderson & Rooney (2019) determined that natural marshes had higher total bird abundance and more bird species than wetlands restored through ditch-plugging methods. Although restored wetlands supported a significantly higher average bird abundance for wetland-associated species compared to less disturbance reference wetlands, both wetland types showed similar overall species diversity for wetland-associated birds. Additionally, the avian community composition in restored wetlands differed from all natural reference types, but the community of wetland-associated species fell within the range of natural wetland variability within the region. Overall, restored wetlands lacked the complexity of the least disturbed natural reference wetlands, with diminished beta diversity and certain species being entirely absent (e.g., no tree-associated bird species were present). Thus, while waterfowl habitat was created through ditch-plugging, this method was not adequate for compensating for the historical loss of habitat for various avian species in the area. Another study in Alberta by Begley *et al.* (2012) found that the number of species per wetland differed significantly between restored Prairie Pothole wetlands and natural reference wetlands, with the former having lower avian species richness and diversity. Species composition also differed, with restored wetlands being more characterised by open-grassland birds and shorebirds, whereas reference wetlands had more woodland-associated species and diving birds. Despite the restored wetlands and reference wetlands being similar size and vegetation type, the researchers concluded that equivalent habitat for avian species was not created. In Prince Edward Island, Canada, Stevens *et al.* (2003) assessed the ability of wetland restoration for revitalising the waterfowl populations to natural levels. The abundance of waterfowl pairs and broods was compared between restored and natural reference wetlands. Six of the eight recorded species had significantly more pairs in restored wetlands, and four of these species also had more broods in restored wetlands compared to the natural reference wetlands.

### 3.4.3 Amphibian Habitat and Communities

Amphibians are good indicators of wetland health. Swartz *et al*. (2019b) assessed the occurrence of four pond-breeding amphibian species' larvae in created PRM wetlands within the Greater Yellowstone Ecosystem, comparing it to impacted and unimpacted natural reference wetlands. Tiger salamanders and chorus frogs colonised PRM and reference wetlands at similar rates, but Columbia spotted frogs larvae and egg masses were more commonly observed in reference wetlands. PRM wetlands were shallower, smaller, and had less aquatic vegetation compared to reference wetlands. Petranka *et al.* (2003) found in North Carolina that wood frogs had similar juvenile production in created bank wetlands and natural and semi-natural reference wetlands, while spotted salamanders had higher juvenile production in the bank wetlands. These results were attributed to successful creation of larger breeding habitats with longer hydroperiods in the bank wetlands, and the tendency of reference wetlands to experience premature drying prior to metamorphosis.

Other studies using amphibians as indicators of wetland health observed fewer positive findings, suggesting challenges in effectively restoring amphibian communities in CWM wetlands. In Ohio, Mack & Micacchion (2006) found that bank wetlands had inferior amphibian habitat quality compared to natural reference wetlands dominated by emergent vegetation. There were different amphibian communities between bank and natural wetlands. Key species such as spotted salamanders and wood frogs, indicative of high-quality sites in Ohio, were absent from banks but found in natural forest and shrub areas of reference wetlands. Similarly, Micacchion *et al.* (2010) observed inferior quality of amphibian communities in Ohio PRM wetlands compared to natural reference wetlands, with 21 of the 24 PRM wetlands monitored classified as 'Limited Quality Wetland Habitat,' indicating poor-quality pond-breeding amphibian habitat. In Canada, Ward & Hossie (2020) found that mole salamander larvae abundance in 1-to-15-year-old created wetlands on Pelee Island was lower than in surrounding 50-year-old natural wetlands, indicating insufficient breeding habitat despite ecosystem-centred remediation efforts.

## 3.5 Whole-Community Approach

Taking a whole-community approach can provide a more holistic understanding of wetland ecological communities. In Michigan, USA, Austin & Schriever (2013) conducted a study comparing plant and animal communities in 2-to-25-year-old created PRM wetlands to established reference wetlands with unknown anthropogenic disturbance. Both wetland types showed similar levels of low to moderate ecological and biological integrity. Biodiversity in the

PRM wetlands resembled that in established wetlands, showing no significant differences in evenness and diversity. However, both types of wetlands were characterised by high proportions of non-native and generalist species. Higher fish diversity was observed in established wetlands, whereas higher fish abundance was observed in the PRM wetlands. The design of the created wetlands, having incorporated creeks and agricultural runoff, resulted in elevated average chlorophyll-a concentrations and dissolved oxygen compared to the reference wetlands.

## 3.6 Landscape-Scale Functional Loss

Combining Hydrogeomorphic (HGM) classification with landscape profiles offers a valuable approach to quantifying the cumulative functional and structural changes within a watershed. In Pennsylvania, USA, Hoeltje & Cole (2007) used regional HGM classification to assess potential functional shifts occurring from CWM practices. PRM wetlands excelled in specific ecological functions, such as retention of inorganic particles, but fell short in supporting vertebrate community structure, detrital biomass maintenance, landscape-scale biodiversity, and the maintenance of natural conditions compared to adjacent reference floodplain wetlands. These discrepancies may be attributed to PRM wetlands experiencing prolonged inundation periods with minimal recession, whereas the reference wetlands had shorter inundation periods that quickly receded.

A shift in HGM types can indicate a potential cumulative functional loss at the landscape level. In Pennsylvania, Gebo & Brooks (2012) found a shift in HGM types across the landscape. Mainstream floodplain CWM sites were less effective than reference sites in several ecological functions, including short-term surface water storage, retention of inorganic particles, export of inorganic carbon, maintenance of characteristic detrital biomass and support for vertebrate communities. In a study in Southern Louisiana, USA’s Liberty Bayou-Tchefuncta Basin, Tyrna

(2008) used HGM to assess whether bank wetlands achieved the goal of NNL. Despite 99% of permitted wetlands functioning similarly to their mitigated counterparts based on HGM attributes (soil, geology, vegetation, water sources and water flow information), an average mitigation ratio of 1.1:1 (ha mitigated to ha developed) resulted in a 1,014-ha functional loss across all banks. This indicated a cumulative impact from incremental permit decisions affecting wetland structure and function. Banks were located in two watersheds, but 46% of permits were not being mitigated within the same watershed. The Money Hill WMB in the Pearl River Basin accounted for 83% of mitigation occurring outside watershed boundaries, leading to the complete removal of 849 ha of wetland functions in the Liberty Bayou-Tchefuncta Basin.

Wetland area is not a good measure for wetland function, but it can be inferred that a substantial reduction in wetland area across an entire ecoregion corresponds to a decline in wetland function. Poulin *et al.* (2016) assessed Québec, Canada’s wetland conservation legislation by reviewing PRM permits from 2006 to 2010. Their analysis revealed a significant 99% loss of wetland area within the St. Lawrence Lowlands ecoregion, with only 15 ha created to compensate for a 2,870-ha loss. This substantial decline in wetland area likely suggests a corresponding deterioration in wetland function.

# 4.0 Discussion

My review revealed limitations in the current North American CWM systems, particularly regarding the long-term ecological viability of wetlands created or restored under both WMB and PRM mechanisms. CWM wetlands often lacked the natural variability found in healthy ecosystems, resulting in potential declines in functionality and resilience across impacted regions over time. CWM wetlands often fall short of replicating the complex structures and functions of high-quality natural wetlands, despite achieving some level of equivalence with degraded wetlands. Furthermore, these constructed wetlands showed inconsistent biodiversity and functional losses across ecologically defined watersheds, suggesting a cumulative loss of function at the landscape level. These findings raise concerns that CWM practices may inadvertently lead to the expansion of degraded wetland areas or even complete wetland function and coverage loss.

Numerous studies attributed the low amphibian habitat quality in CWM wetlands to an inability to restore natural hydrological regimes. Even when explicitly prioritised, creating habitats suitable for conserving rarer, high-value species remains a significant challenge within CWM practices. Given the concerning decline in amphibian populations globally and across North America specifically (Houlahan *et al.,* 2000), the conservation of existing pond-breeding amphibian habitats is crucial. Hydrological extremes present substantial risks. On the drier spectrum, they can be detrimental to amphibian larvae prior to metamorphosis. For instance, Swartz *et al*., (2019b) observed that over 80% of created wetlands experienced partial or complete drying before metamorphosis. This raises concerns about the effectiveness of CWM projects in serving as suitable habitats, questioning whether they might act as population sinks or ecological traps. In contrast, many created wetlands experienced prolonged or permanent inundation, which resulted in diminished hydrological variability and potential biotic homogenization.

Excessive water depth or permanent inundation can also elevate predation risk for larvae of high-value species. In the study by Mack & Micacchion (2006), where banks had permanent hydroperiods, predatory fish were present. Micacchion *et al*.'s (2010) attributed the insufficient breeding habitat for amphibians to the presence of aggressive predatory species praying on both adults and larvae. Austin & Schriever (2013) also noted how incorporating creeks and agricultural runoff into wetland design improved water flow in created wetlands, resulting in higher dissolved oxygen and productivity. However, these aquatic corridors also facilitated colonisation of highly predatory fish species, such as bullhead catfish and centrarchid sunfishes. Similarly, Ward & Hossie (2020) found that ponds lacking sufficient breeding habitat for mole salamanders consistently lacked larvae in the presence of fish. In contrast, none of the natural ponds where salamanders were found contained fish.

Reduced CNP nutrients in the soils of CWM wetlands suggests that nutrient cycling is impaired. The reduced denitrification functions in CWM also suggests that nutrient loading in nearby surface waters may be occurring (Hossler *et al.* 2011). Eutrophication is an important driver of methane emissions which is expected to increase over the next century due to climate change. Nutrient loading in lentic waters could increase annual carbon dioxide emissions over the next century to levels equivalent to 18-33% of annual emissions from fossil fuel combustion (Beaulieu *et al.,* 2019). Therefore, the reduced CNP functions through CWM practices and the associated potential increase in eutrophication in surrounding waters is a major concern that policymakers must take into consideration, as it threatens our ability to mitigate the effects of climate change.

Incorporating variation in micro-topography, spatial layout, and water flow variation into CWM wetland design may help promote macroinvertebrate communities. Features like variation in water depth and elevation, along with a balanced mix of emergent vegetation and open water areas promote a wider range of macroinvertebrates (Balcombe *et al.* 2005). For instance, areas with extended inundation and a mix of open water and emergent vegetation support more snails and slugs, while zones with less open water and increased plant cover provide refugia for other macroinvertebrate groups like aquatic isopods (Asellidae). Furthermore, incorporating strategic canopy cover with areas of open tree canopy allows sunlight penetration into the water, enhancing dissolved oxygen levels and fostering the growth of filamentous algae which serves as a food source for many macroinvertebrate gazers (Gianopulos *et al.* 2021). By considering these factors and incorporating elements like diverse water depths, balanced vegetation zones, and strategic canopy cover, CWM wetland design can potentially promote healthy and diverse macroinvertebrate communities.

## 4.1 Research Gaps

Current CWM practices often rely solely on basic vegetation metrics, which limits our understanding and quantification of overall wetland function. Long-term monitoring that goes beyond just vegetation inventories to incorporate floristic quality is crucial to accurately assess the successional trajectory of CWM wetlands. Furthermore, research on beta diversity and the potential for biotic homogenization within CWM wetlands is scarce.

Achieving natural hydrological regimes in CWM wetlands remains a significant challenge. Current practices often fail to adequately replicate the depth and duration of inundation observed in natural wetlands. This inconsistency poses a major threat to pond-breeding amphibian populations. More research is needed for identifying the optimal wetland depths during creation. This ideal depth should avoid permanent inundation while ensuring depths are deep enough to withstand dry spells during hot weather events. It is also important to keep in mind that restoring natural hydrology is not always sufficient for achieving full ecological equivalence. Further investigation is needed to understand how other wetland functions might remain compromised even when hydrological regimes are successfully restored.

Significant differences were found in the microbial-mediated functions between CWM and natural wetlands, particularly regarding denitrification potential. There was a notable reduction in CNP-related cycling within CWM wetlands compared to the natural reference wetlands. This indicates potential nutrient limitations impacting plant growth, warranting further investigation. Moreover, most studies did not take a holistic approach for wetland evaluation, an approach that is needed for understanding the overall ecological and biological integrity of CWM wetlands. Notably, only one study reviewed employed a whole-community approach. Additionally, further research is needed to elucidate the relationship between wetland size and functional decline across ecologically defined watersheds.

Achieving the goal of NNL depends heavily on the future resilience of CWM wetlands. Studies by Balcombe *et al.* (2015) and Swartz *et al.* (2019b) offer some promising insights. Their findings suggest that impacted wetlands might retain some resilience to human disturbances, as long as the impact zone is limited to less that 25% of the wetland perimeter. To confirm these findings, however, further research is needed. In the meantime, it is crucial to prioritise avoidance and minimising impacts altogether.

Finally, while the Government of Canada and some provinces have implemented the goal of NNL for wetlands, my review has identified a critical gap in research directly evaluating Canadian CWM practices. Filling this knowledge gap should be a key priority.

## 4.2 Study Limitations

The overrepresentation of studies evaluating CWM practices in the USA poses a limitation. Consequently, the findings of my review may not be generalizable to all of North America, but rather are more specific to the USA. Furthermore, the majority of studies I reviewed primarily compared CWM wetlands to natural reference wetlands, rather than contrasting them with the functions of the impact site prior to development. This is a fundamental challenge in assessing whether the goal of NNL has been achieved, as direct comparisons of losses and gains are often absent. Additionally, the inability to monitor all wetland functions introduces another limitation to the conclusions drawn in my review.

# 5.0 Future Recommendations and Conclusion

Given that the trajectory of CWM wetland ecology appears to be complex and non-linear, the currently required 5-year monitoring period may be insufficient to capture critical changes. Capturing trends in the longer term, such as successional declines in plant or amphibian community quality, might require extending the monitoring time frame. Stefanik & Mitsch (2012) advocate for a monitoring period of at least 10-15 years after wetland creation. Extending the monitoring comes with logistical challenges, however, particularly economic costs for bank sponsors. If a longer monitoring period becomes financially prohibitive for bank sponsors, it raises the question as to whether CWM should even be implemented at all, as a rapidly changing climate amidst a biodiversity crisis may mean the risk is too high.

Establishing high-quality vegetation in CWM wetlands remains a challenge often attributed to invasive species. While some projects meet basic vegetation performance criteria, such as native species composition, it remains a hurdle to effectively managing invasive plants (Austin *et al.* 2013; Mack & Micacchion, 2006; Price *et al.* 2019; Tillman *et al.,* 2022; Van den Bosch & Matthews, 2016). Several studies offer strategies for tackling this issue. De Steven *et al.* (2010) suggests leveraging diverse seed banks and restoring hydroperiods appropriately in less degraded environments for promoting successful wetland restoration with minimal invasive species presence. Similarly, Van den Bosch & Matthews (2016) propose that situating wetlands near high-quality natural areas can enhance and maintain long-term floristic quality. Additionally, Micacchion (2012) recommends restoring suitable growing mediums and employing high seeding rates to discourage invasive species reinvasion. Beyond these approaches, intensive control measures during construction and ongoing management are crucial. Common reed canary grass (*Phalaris arundinacea*), narrowleaf cattail (*Typha angustifolia*), and common reed (*Phragmites australis*) pose significant challenges (Austin & Schriever, 2013; Mack & Miacacchion, 2006; Price *et al.,* 2019; Tillman *et al.,* 2022; Van den Bosch & Matthews, 2016). However, there are some encouraging observations. McKown *et al.* (2021) documented a decline in the dominance of common reed canary grass over a 28-year period, suggesting the possibility of long-term control. This study also noted an increase in non-native shrubs, however, highlighting the need for continued monitoring and management. Overall, controlling invasive species is critical for both WMB and PRM success in achieving not only regulatory goals but true ecological equivalence to natural wetlands. Tillman *et al.,* (2022) recommended stricter performance standards with a zero-tolerance approach for the invasive species listed above, alongside more intensive and prolonged control measures.

Plant and animal species that colonised the CWM wetlands often exhibited higher dispersal capabilities. In Swartz *et al*. (2019), specific invertebrates, such as the pea clam, were associated with hydrologic connectivity or were in areas where waterfowl are present. In Tillman *et al*. (2022), rarer species with higher conservation values had difficulty becoming established in WMBs due to dispersal limitations. De Steven *et al*.'s (2010) study also found that the typical native perennial species were dispersal-limited and failed to colonise the depressions. Future CWM wetland design should promote dispersal of plant and animal species, particularly for those with high conservation value or those that are dispersal limited. This could include connecting CWM wetlands to existing natural areas through corridors of suitable habitat, designing water flow patterns that facilitate the movement of amphibians, or including a mix of native plant species that are readily eaten by waterfowl that can help disperse other species.

Studies provided several other recommendations for improving the amphibian and avian communities in CWM wetlands. To help ensure wetlands are not designed as ecological traps for amphibian populations, Swartz *et al.* (2019b) advocated for designing CWM wetlands with structural complexity, particularly deeper water zones. First, these zones provide critical refugia for amphibian larvae during dry spells, preventing wetlands from becoming ecological traps. Second, the deeper water contributes to the overall resilience of the wetland against climate fluctuations by being part of a complex habitat structure. Therefore, incorporating these deeper areas alongside shallow littoral zones is a crucial design element. Building on the importance of structural complexity, Austin & Schriever (2013) recommend incorporating trait-based analyses into wetland planning. This approach considers the ecological needs of target amphibian and avian species. Similarly, Gianopulos *et al.* (2021) advocate for increased habitat heterogeneity within wetland design. This means creating more microhabitats, like varying water depths, plant zones and basking areas to benefit a diverse range of species with diverse requirements (Gianopulos *et al*., 2021). Furthermore, Balcombe *et al.* (2005) emphasise the importance of considering habitat heterogeneity in the broader landscape context. Designing wetlands in isolation might not be sufficient. A watershed-scale approach is currently merely a recommendation by the ACE, but should realistically be a requirement, considering networks or corridors of wetlands throughout the watershed to improve habitat connectivity.

Achieving successful ecological CWM wetland outcomes hinges on both strategic placement and data-driven design. Ideally, these wetlands should be situated within a mainstream floodplain to capitalise on the natural benefits of overbank flooding for restoring hydrological capacity (Gebo & Brooks, 2012). Beyond placement, design considerations should factor in avian species colonisation. Anderson & Rooney (2019) suggest two potential approaches: plant shrubs in deforested areas to enhance habitat suitability for tree-dependent birds, and/or target projects near existing forested areas to leverage readily available nesting and foraging habitat. However, effective site selection goes beyond just location and design. It is crucial to account for potential legacy effects, which can hinder a wetland's ability to achieve ecological goals. Peralta *et al.* (2010) and Micacchion (2012) highlighted the lasting impacts of past land use such as agriculture and urbanisation on soil properties, potentially limiting functions like denitrification (Peralta *et al.* 2010) or achieving high plant diversity (Micacchion, 2012). Similarly, Tyna (2008) emphasised the potential challenges of re-establishing natural hydrology and soil characteristics in areas impacted by past disturbances like hurricanes. Understanding these legacy effects is crucial for informed site selection. By carefully considering past land use, hydrology, and potential long-term impacts, developers and bank managers can choose sites with a higher likelihood of successful restoration and ecological equivalence.

To achieve the goal of NNL, prioritising avoidance in the mitigation hierarchy is crucial. When impacts to wetlands are absolutely unavoidable, however, both the impacts and compensation actions should ideally occur in areas already experiencing significant degradation. While CWM practices might achieve equivalence with nearby disturbed wetlands, a more ambitious approach is necessary. CWM efforts should strive to replicate the functionality of high-quality wetlands or even aim for net gain in severely degraded areas. This approach is crucial for restoring lost wetland functions, particularly in the context of mitigating and adapting to the effects of climate change. By prioritising degraded areas for wetland loss, we can significantly reduce the risk of failing to achieve ecological equivalence and increase the chances of achieving net gain of function, thus restoring lost ecological services.

# 7.0 Bibliography

Amano, T., González-Varo, J. P., & Sutherland, W. J. (2016). Languages are still a major barrier to global science. *PLOS Biology*, *14*(12). https://doi.org/10.1371/journal.pbio.2000933

Ambrose, R. F. (2000). Wetlands mitigation in the United States: Assessing the success of mitigation policies. *Wetlands Australia*, *19*(1), 1. https://doi.org/10.31646/wa.242

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O., Swartz, B., Quental, T. B., Marshall, C., McGuire, J. L., Lindsey, E. L., Maguire, K. C., Mersey, B., & Ferrer, E. A. (2011). Has the Earth's sixth mass extinction already arrived? *Nature*, *471*(7336), 51–57. https://doi.org/10.1038/nature09678

Beaulieu, J. J., DelSontro, T., & Downing, J. A. (2019). Eutrophication will increase methane emissions from lakes and impoundments during the 21st Century. *Nature Communications*, *10*(1). https://doi.org/10.1038/s41467-019-09100-5

Bennett, G., Gallant, M., & Kate, K. (2017). *State of Biodiversity Mitigation 2017. Markets and Compensation for Global Infrastructure Development.* Forest Trends' Ecosystem Marketplace. Retrieved from https://www.forest-trends.org/wp-content/uploads/2018/01/doc_5707.pdf

Bond, W. K., Cox, K. W., Heberlein, T., Manning, E. W., Witty D. R., & Young, D. A. (1992). *Wetland Evaluation Guide.* North American Wetlands Conservation Council. Retrieved from https://nawcc.wetlandnetwork.ca/Wetland%20Evaluation%201992.pdf

Burgin, S. (2009). 'mitigation banks' for wetland conservation: A major success or an unmitigated disaster? *Wetlands Ecology and Management*, *18*(1), 49–55. https://doi.org/10.1007/s11273-009-9147-5

Calvet, C., Napoléone, C., & Salles, J.-M. (2015). The biodiversity offsetting dilemma: Between economic rationales and ecological dynamics. *Sustainability*, *7*(6), 7357–7378. https://doi.org/10.3390/su7067357

Cole, C. A., & Shafer, D. (2002). Section 404 wetland mitigation and Permit Success Criteria in Pennsylvania, USA, 1986-1999. *Environmental Management*, *30*(4), 508–515. https://doi.org/10.1007/s00267-002-2717-4

Convention on Wetlands. (2021). Global Wetland Outlook: Special Edition 2021. Gland, Switzerland: Secretariat of the Convention on Wetlands.

Cox, K. W., & Grose, A. (2000). *Wetland mitigation in Canada: A framework for application*. North American Wetlands Conservation Council (Canada).

Dertien, J. S., Self, S., Ross, B. E., Barrett, K., & Baldwin, R. F. (2020). The relationship between biodiversity and wetland cover varies across regions of the conterminous United States. *PLOS ONE*, *15*(5). https://doi.org/10.1371/journal.pone.0232052

Environmental Protection Agency. (EPA). (2024). *Permit Program under CWA Section 404.* United States Environmental Protection Agency. Retrieved from https://www.epa.gov/cwa-404/permit-program-under-cwa-section-404

Environmental Protection Agency. (EPA). (2024b). *Why are Wetlands Important?* Environmental Protection Agency. Retrieved from https://www.epa.gov/wetlands/why-are-wetlands-important

Environmental Protection Agency. (EPA). (2023). *How do Wetlands Function and Why are they Valuable?* Environmental Protection Agency. Retrieved from https://www.epa.gov/wetlands/how-do-wetlands-function-and-why-are-they-valuable

Environmental Protection Agency (EPA). (2023b). *Mitigation Banks under CWA Section 404*. United States Environmental Protection Agency. Retrieved from https://www.epa.gov/cwa-404/mitigation-banks-under-cwa-section-404

Environment and Climate Change Canada. (ECCC). (2023). *Timeline: Canadian Wildlife Service and Canada's nature conservation milestones.* Government of Canada. Retrieved from https://www.canada.ca/en/environment-climate-change/campaigns/50-years-environmental-action/eccc-timeline/canadian-wildlife-service-nature-conservation.html

Ermgassen, S. O., Baker, J., Griffiths, R. A., Strange, N., Struebig, M. J., & Bull, J. W. (2019). The ecological outcomes of biodiversity offsets under "No net loss" policies: A global review. *Conservation Letters*, *12*(6). https://doi.org/10.1111/conl.12664

Hanski, I. (2011). Habitat loss, the dynamics of biodiversity, and a perspective on conservation. *AMBIO*, *40*(3), 248–255. https://doi.org/10.1007/s13280-011-0147-3

Horwitz, P., Finlayson, M. C., Weinstein, P. (2012). *Healthy wetlands, healthy people: A review of wetlands and human health interactions.* Ramsar Technical Report No. 6. Retrieved from https://www.ramsar.org/sites/default/files/documents/pdf/lib/rtr6-health.pdf

Houlahan, Jeff. E., Findlay, C. S., Schmidt, B. R., Meyer, A. H., & Kuzmin, S. L. (2000). Quantitative evidence for global amphibian population declines. *Nature*, *404*(6779), 752–755. https://doi.org/10.1038/35008052

Institute for Water Resources. (IWR). (2015). *The Mitigation Rule Retrospective: A Review of the 2008 Regulations Governing Compensatory Mitigation for Losses of Aquatic Resources.* Institute for Water Resources. Retrieved from https://www.epa.gov/sites/default/files/2015-11/documents/mitrule_report_october_2015.pdf

Kate, K., Bishop, J., Bayon, R.  (2004). Biodiversity Offsets: Views, experience, and the business case. IUCN. https://www.iucn.org/sites/default/files/import/downloads/bdoffsets.pdf

Keddy, P. A., Fraser, L. H., Solomeshch, A. I., Junk, W. J., Campbell, D. R., Arroyo, M. T., & Alho, C. J. (2009). Wet and wonderful: The world's largest wetlands are conservation priorities. *BioScience*, *59*(1), 39–51. https://doi.org/10.1525/bio.2009.59.1.8

Koh, L. P., Dunn, R. R., Sodhi, N. S., Colwell, R. K., Proctor, H. C., & Smith, V. S. (2004). Species Coextinctions and the Biodiversity Crisis. *Science*, *305*(5690), 1632–1634. https://doi.org/10.1126/science.1101101

Lele, S., Springate-Baginski, O., Lakerveld, R., Deb, D., & Dash, P. (2013). Ecosystem services: Origins, contributions, Pitfalls, and alternatives. *Conservation and Society*, *11*(4), 343. https://doi.org/10.4103/0972-4923.125752

Levrel, H., Scemama, P., & Vaissière, A.C. (2017). Should we be wary of mitigation banking? evidence regarding the risks associated with this wetland offset arrangement in Florida. *Ecological Economics*, *135*, 136–149. https://doi.org/10.1016/j.ecolecon.2016.12.025

Mitsch, W. J., & Hernandez, M. E. (2012). Landscape and climate change threats to wetlands of North and Central America. *Aquatic Sciences*, *75*(1), 133–149. https://doi.org/10.1007/s00027-012-0262-7

Nature Conservancy Canada. (NCC). (2023). *Wetlands 101*. NCC. https://www.natureconservancy.ca/en/what-we-do/resource-centre/conservation-101/wetlands-101.html

OECD. (2016). *Biodiversity Offsets: Effective Design and Implementation.* OECD Publishing. Paris. http://dx.doi.org/10.1787/9789264222519-en

Rubec, C. D., & Hanson, A. R. (2008). Wetland mitigation and compensation: Canadian experience. *Wetlands Ecology and Management*, *17*(1), 3–14. https://doi.org/10.1007/s11273-008-9078-6

Salimi, S., Almuktar, S. A. A. A. N., & Scholz, M. (2021). Impact of climate change on wetland ecosystems: A critical review of experimental wetlands. *Journal of Environmental Management*, *286*, 112160. https://doi.org/10.1016/j.jenvman.2021.112160

Tillman, S. C., & Matthews, J. W. (2023). Evaluating the ability of wetland mitigation banks to replace plant species lost from destroyed wetlands. *Journal of Applied Ecology*, *60*(6), 990–998. https://doi.org/10.1111/1365-2664.14391

United States Army Corps of Engineers (Corps) & Environmental Protection Agency (EPA). (2008). *Compensatory Mitigation for Losses of Aquatic Resources under CWA Section 404 (Final Rule).* United States Environmental Protection Agency. Retrieved from https://www.epa.gov/sites/default/files/2015-03/documents/2008_04_10_wetlands_wetlands_mitigation_final_rule_4_10_08.pdf

Vaissière, A.-C., Levrel, H., & Pioch, S. (2017). Wetland mitigation banking: Negotiations with stakeholders in a zone of ecological-economic viability. *Land Use Policy*, *69*, 512–518. https://doi.org/10.1016/j.landusepol.2017.09.049

Zedler, J. B., & Callaway, J. C. (1999). Tracking wetland restoration: Do mitigation sites follow desired trajectories? *Restoration Ecology*, *7*(1), 69–73. https://doi.org/10.1046/j.1526-100x.1999.07108.x

Zinn, J. (1997, September 12). *Wetland mitigation banking: Status and prospects*. EveryCRSReport.com. Retrieved from https://www.everycrsreport.com/reports/97-849.html

# 8.0 Articles Selected for Review

Ambrose, Callaway & Lee. (2007). *An Evaluation of Compensatory Mitigation Projects Permitted Under Clean Water Act Section 401 by the California State Water Resources Control Board, 1991-2002. Final Report.* California State Water Resources Control Board. Retrieved from https://www.waterboards.ca.gov/water_issues/programs/cwa401/docs/wetlandmitstudy_rpt.pdf

Anderson, D. L., & Rooney, R. C. (2019). Differences exist in bird communities using restored and natural wetlands in the Parkland Region, Alberta, Canada. *Restoration Ecology*, *27*(6), 1495–1507. https://doi.org/10.1111/rec.13015

Austin, A. R., & Schriever, T. A. (2023). Created wetlands support similar communities of low conservation value as established wetlands in Michigan. *Wetlands Ecology and Management*, *31*(4), 521–537. https://doi.org/10.1007/s11273-023-09931-6

Badiou, P., McDougal, R., Pennock, D., & Clark, B. (2011). Greenhouse gas emissions and carbon sequestration potential in restored wetlands of the Canadian Prairie Pothole Region. *Wetlands Ecology and Management*, *19*(3), 237–256. https://doi.org/10.1007/s11273-011-9214-6

Balcombe, C. K., Anderson, J. T., Fortney, R. H., & Kordek, W. S. (2005). Aquatic macroinvertebrate assemblages in mitigated and natural wetlands. *Hydrobiologia*, *541*(1), 175–188. https://doi.org/10.1007/s10750-004-5706-1

Begley, A. J. P., Gray, B. T., & Paszkowski, C. A. (2012). A comparison of restored and natural wetlands as habitat for birds in the Prairie Pothole Region of Saskatchewan, Canada. *The Raffles Bulletin of Zoology*, *60*, 173–187.

De Steven, D., Sharitz, R. R., & Barton, C. D. (2010). Ecological Outcomes and evaluation of success in passively restored southeastern depressional wetlands. *Wetlands*, *30*(6), 1129–1140. https://doi.org/10.1007/s13157-010-0100-4

Fennessy, M. S., Rokosch, A., & Mack, J. J. (2008). Patterns of plant decomposition and nutrient cycling in natural and created wetlands. *Wetlands*, *28*(2), 300–310. https://doi.org/10.1672/06-97.1

Gebo, N. A., & Brooks, R. P. (2012). Hydrogeomorphic (HGM) assessments of mitigation sites compared to natural reference wetlands in Pennsylvania. *Wetlands*, *32*(2), 321–331. https://doi.org/10.1007/s13157-011-0267-3

Gianopulos, K., Shiflett, S., Baker, V., & Rubino, G. (2021). Aquatic macroinvertebrate communities in relation to environmental variables in restoration and reference wetlands in North Carolina, USA. *Wetlands Ecology and Management*, *29*(6), 823–841. https://doi.org/10.1007/s11273-021-09812-w

Hoeltje, S. M., & Cole, C. A. (2007). Losing function through wetland mitigation in Central Pennsylvania, USA. *Environmental Management*, *39*(3), 385–402. https://doi.org/10.1007/s00267-006-0212-z

Hossler, K., Bouchard, V., Fennessy, M. S., Frey, S. D., Anemaet, E., & Herbert, E. (2011). No-net-loss not met for nutrient function in freshwater marshes: Recommendations for wetland mitigation policies. *Ecosphere*, *2*(7). https://doi.org/10.1890/es11-00009.1

Mack, J.J., & Micacchion, M. (2006). *An ecological assessment of Ohio mitigation banks: Vegetation, Amphibians, Hydrology, and Soils.* Ohio EPA Technical Report. Retrieved from https://epa.ohio.gov/static/Portals/35/wetlands/Bank_Report_Ohio_Final.pdf

McKown, J. G., Moore, G. E., Payne, A. R., White, N. A., & Gibson, J. L. (2021). Successional dynamics of a 35 year old freshwater mitigation wetland in southeastern New Hampshire. *PLOS ONE*, *16*(5). https://doi.org/10.1371/journal.pone.0251748

Micacchion, M., Gara, B., and Mack, J.J. (2010). *Assessment of wetland mitigation projects in Ohio. Volume 1: An Ecological Assessment of Ohio Individual Wetland Mitigation Projects.* Ohio EPA Technical Report. Retrieved from https://epa.ohio.gov/static/Portals/35/wetlands/M928_Final_Report_Vol_1.pdf

Micacchion, M. (2012). *Great Lakes Basin Evaluation of Compensation Sites Report*. Great Lakes Basin Evaluation of Compensation Sites Report. https://midwestbiodiversityinst.org/publications/reports/great-lakes-basin-evaluation-of-compensation-sites-report

Peralta, A. L., Matthews, J. W., & Kent, A. D. (2010). Microbial community structure and denitrification in a wetland mitigation bank. *Applied and Environmental Microbiology*, *76*(13), 4207–4215. https://doi.org/10.1128/aem.02977-09

Petranka, J. W., Murray, S. S., & Kennedy, C. A. (2003). Responses of amphibians to restoration of a southern appalachian wetland: Perturbations confound post-restoration assessment. *Wetlands*, *23*(2), 278–290. https://doi.org/10.1672/7-20

Poulin, M., Pellerin, S., Cimon-Morin, J., Lavallée, S., Courchesne, G., & Tendland, Y. (2016). Inefficacy of wetland legislation for conserving Quebec wetlands as revealed by mapping of recent disturbances. *Wetlands Ecology and Management*, *24*(6), 651–665. https://doi.org/10.1007/s11273-016-9494-y

Price, E. P., Spyreas, G., & Matthews, J. W. (2019). Wetland compensation and its impacts on β-diversity. *Ecological Applications*, *29*(1). https://doi.org/10.1002/eap.1827

Salaria, S., Howard, R., Clare, S., & Creed, I. F. (2018). Incomplete recovery of plant diversity in restored prairie wetlands on agricultural landscapes. *Restoration Ecology*, *27*(3), 520–530. https://doi.org/10.1111/rec.12890

Spieles, D. J., Coneybeer, M., & Horn, J. (2006). Community structure and quality after 10 years in two Central Ohio Mitigation Bank Wetlands. *Environmental Management*, *38*(5), 837–852. https://doi.org/10.1007/s00267-005-0294-z

Stefanik, K. C., & Mitsch, W. J. (2012). Structural and functional vegetation development in created and restored wetland mitigation banks of different ages. *Ecological Engineering*, *39*, 104–112. https://doi.org/10.1016/j.ecoleng.2011.11.016

Stevens, C. E., Gabor, T. S., & Diamond, A. W. (2003). Use of restored small wetlands by breeding waterfowl in Prince Edward Island, Canada. *Restoration Ecology*, *11*(1), 3–12. https://doi.org/10.1046/j.1526-100x.2003.00107.x

Swartz, L. K., Hossack, B. R., Muths, E., Newell, R. L., & Lowe, W. H. (2019). Aquatic macroinvertebrate community responses to wetland mitigation in the Greater Yellowstone Ecosystem. *Freshwater Biology*, *64*(5), 942–953. https://doi.org/10.1111/fwb.13276

Swartz, L. K., Lowe, W. H., Muths, E. L., & Hossack, B. R. (2019b). Species-specific responses to wetland mitigation among amphibians in the Greater Yellowstone Ecosystem. *Restoration Ecology*, *28*(1), 206–214. https://doi.org/10.1111/rec.13031

Tillman, S. C., Spyreas, G., Olnas, A., & Matthews, J. W. (2022). Plant communities in wetland mitigation banks surpass the quality of those in the most degraded, naturally occurring wetlands, but fall short of high-quality wetlands. *Ecological Engineering*, *176*, 106526. https://doi.org/10.1016/j.ecoleng.2021.106526

Tyrna, A. A. (2008). *Wetland mitigation banks and the no-net-loss requirement: An evaluation of the section 404 permit program in Southeast Louisiana*. LSU Scholarly Repository. https://repository.lsu.edu/gradschool_theses/3116/

Van den Bosch, K., & Matthews, J. W. (2016). An assessment of long-term compliance with performance standards in compensatory mitigation wetlands. *Environmental Management*, *59*(4), 546–556. https://doi.org/10.1007/s00267-016-0804-1

Ward, M., & Hossie, T. J. (2020). Do existing constructed ponds on Pelee Island, Ontario match the habitat requirements of endangered Ambystoma larvae? *Wetlands*, *40*(6), 2097–2108. https://doi.org/10.1007/s13157-020-01364-8

**APPENDIX.**

**Table 1.** The Inclusion and Exclusion Criteria for this Review.

| **Inclusion Criteria** | **Exclusion Criteria** |
|---|---|
| • Peer-reviewed journals from search engines and academic journals such as Web of Science and Omni<br>• Studies that directly measure or infer function of wetlands created, restored, or enhanced under WMB, ILF, or PRM systems<br>• Studies that compare structure and function of CWM wetlands to natural reference wetlands of a gradient of anthropogenic disturbance<br>• Studies from relevant topic categories such as the type of biodiversity outcome variable, magnitude of the outcome variable, context of the landscape, and specific wetland ecological functions (animal and plant abundance and diversity, soil formation, nutrient cycling, hydrology, climatic regulation, and biological regulation).<br>• Studies that reported observed, not simulated, ex-post outcomes of CWM practices<br>• Geographical scope: North America (Canada and USA) | • Non-English studies<br>• Studies outside of the geographical scope of North America (Canada and USA)<br>• Studies that are not focused on WMB, ILF, or PRM strategies<br>• Studies that do not provide outcome variables or do not measure the impact on biodiversity or ecosystem services/functions |