Offsetting approved harmful anthropogenic impacts in the 21st century – Insights into global offsetting practices, habitat banking as an alternative offsetting mechanism and application of habitat enhancement in northern boreal lake systems

by

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in
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Abstract

Land-use change via human development is a major driver of biodiversity and habitat area loss and ecosystem function impairment. To reduce these impacts, billions of dollars are spent on environmental offsets, aimed to compensate for authorized negative impacts. Studies evaluating offset project effectiveness and individual mechanisms, remain rare.

Chapter 2 aimed to address the persistent questions of whether high project compliance is synonymous with high functional success as well as how to address residual or chronic impacts in aquatic ecosystems which can occur after offset establishment, for example because of the ephemeral timescale of some projects through a systematic review process and meta-analysis. While compliance and function were related to each other, a high compliance score did not guarantee a higher degree of function. However, function did improve with larger projects, specifically when projects targeted productivity or specific habitat features, and when multiple complementary management targets were in place. Altogether these relationships highlight specific ecological processes that may help improve offsetting outcomes for the conservation of habitat and biodiversity. The meta-analysis for offsetting residual or chronic impacts yielded three main approaches (habitat creation; restoration and enhancement and biological manipulation). Habitat creation projects, mainly targeting salmonids, with a high pooled effect size (0.8) and biomass increase (x1.4), needs to be explored for other species. Habitat restoration projects targeted a wide range of species and communities with a pooled effect size of 0.66, and intermediate biomass increases (>1x). Biological manipulation had the lowest effect size (0.51) with effort outcomes being highly variable.

Conservation and mitigation banks are widely used alternative mechanisms to traditional offsetting to compensate for unavoidable negative environmental impacts from development. In Chapter 3 we utilized publicly available banking data from for the United States to test whether area ratio requirements were met as well as how well ecological equivalency was achieved and to model current and future bank reserves through a predictive modeling framework. We conclude that most bank transactions using Preservation, Enhancement, and Re-establishment targeting wetlands, species, or multiple Mitigation-Targets met No Net Loss requirements on a ratio base. Wetland transactions, making up most of all assessed transactions (n = 10628), still

missed matching appropriate impact to offset types in 25% of all cases, mainly due to Preservation not leading to any additional habitat area gain. While the Preservation of wetlands and the Rehabilitation of streams can provide a multitude of benefits, both practices need to be revised on an ecological level to bridge the gap between Not Net Loss based on credit and area yield ratios and ecological equivalence. Future predictions indicate a decrease in available reserves for banks targeting wetlands or multiple ecosystems, with potential bottlenecks relating to large reserves being limited to the Southeast and release schedules not catching up to the current and anticipated demand. Banks targeting species or streams are predicted to meet future demand, with species banks (conservation banks) following a different legislative and operational approach based on the listing of endangered species. Most current reserves for all four bank types are restricted to very few service areas with around one-third of all bank areas still awaiting release, limiting their availability on a broader scale.

Chapter 4 focussed on the introduction of coarse woody habitat in a northern boreal lake and responses of aquatic fish, invertebrate and macrophyte communities through a Bayesian modeling approach and the use of changes in beta diversity components over time. Catch data was collected over 2 years and posterior model predictions showed an increase in habitat use of the enhanced areas by resident fish (spottail shiner - *Notropis hudsonius*; northern pike - *Esox lucius*; white sucker - *Catostomus commersonii*; brook stickleback - *Culaea inconstans*), while no probable effect on overall fish health, measured in Relative Weight, was linked to the enhancements. Enhancement structures featured increased macrophyte and invertebrate richness and biomass compared to reference sites and pre-treatment assessments over the course of three years. Enhanced sites also retained improved richness (macrophytes), diversity (macroinvertebrates) and biomass (both), despite structural integrity loss of enhancements as early as 1 week post construction.

Preface

This thesis is an original work by Sebastian Theis. Ethics approval for this research project, for which this thesis is a part, was received from the University of Alberta Research Ethics Board, Animal Care and Use Committee "Oil Sands Compensation Lakes" AUP00001547 and provincial Research License 20-1802 RL issued by Alberta Parks and Environment (AEP). This thesis is written in a paper-based format and uses the collective 'we' / 'our' as to keep with the respective journal format. The three chapters consist of six individual papers which are either published or under review as peer-reviewed journal articles or scientific reports. All image resolutions have been reduced to 200dpi to reduce document size and prevent loading and display issues.

Co-authors helping with the publication process for the papers in each chapter were: Jonathan L.W. Ruppert from the Toronto and Region Conservation Authority, Karling N. Roberts from the University of Alberta, Charles K. Minns from the University of Toronto, Marten Koops from Fisheries and Oceans, Jesse Shirton from fRI research and Mark S. Poesch from the University of Alberta. Contact information can be found in each Chapter preceding the respective paper.

Chapter 2.1 has been published as of February 2020 in Conservation Biology: 10.1111/cobi.13343 as *Compliance with and ecosystem function of habitat offsets in North American and European freshwaters* and as part of a Canadian Science Advisory Secretariat (CSAS) meeting in January 2019. The full report can be accessed under: Science Advisory Report 2020/013. I was responsible for the data collection and analysis as well as the manuscript composition. Jonathan L.W. Ruppert, Karling N. Roberts, Charles K. Minns and Marten Koops contributed to manuscript edits. Mark S. Poesch served as the supervisory author and was involved with concept formation and manuscript composition.

Chapter 2.2 is in the process of being as part of a Canadian Science Advisory Secretariat (CSAS) meeting in April 2021. The full report can be accessed once it has undergone the review process. I was responsible for the data collection and analysis as well as the manuscript composition. Marten Koops contributed to manuscript edits. Mark S. Poesch served as the supervisory author and was involved with concept formation and manuscript composition.

Chapter 3.1 is currently under review and considered for publication in Conservation Science and Practice CSP2-21-0306.R1 after a first round of revisions (October 2021). I was responsible for the data collection and analysis as well as the manuscript composition. Mark S. Poesch served as the supervisory author and was involved with concept formation and manuscript composition.

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Chapter 4.1 is currently under review and considered for publication in Ecological Solutions and Evidence ESO-21-12-117 (December 2021). I was responsible for the data collection and analysis as well as the manuscript composition. Jonathan L. W. Ruppert contributed to manuscript edits. Mark S. Poesch served as the supervisory author and was involved with concept formation and manuscript composition.

Chapter 4.2 has been accepted for publication in Aquatic Ecology as of January 12th, 2022, 10.1007/s10452-022-09949-7 as *Measuring beta diversity components and beneficial effects of coarse woody habitat introduction on invertebrate and macrophyte communities in a shallow northern boreal lake; implications for offsetting*. I was responsible for the data collection and analysis as well as the manuscript composition. Jesse R. Shirton assisted with the data collection process in the field. Jonathan L. W. Ruppert contributed to manuscript edits. Mark S. Poesch served as the supervisory author and was involved with concept formation and manuscript composition.

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List of Abbreviations

AIC - Akaike Information Criterion

AICc - Akaike Information Criterion corrected for small sample size

ANOVA - ANalysis Of VAriance

ARIMA – Auto-Regressive Integrated Moving Average

AUR – Accumulated Unwithdrawn Reserve

BACI – Before-After-Control-Impact

BIC - Bayesian Information Criteria

CI – Confidence Interval

CPUA – Catch Per Unit Area

CPUE – Catch Per Unit Effort

CWA – Clean Water Act

CWH – Coarse Woody Habitat

DFO – Fisheries and Oceans Canada

EA – Age Equivalents

EPA/ USEPA – United States Environmental Protection Agency

ESA – Endangered Species Act

ESS – Effective Sample Size

FSA – Fish Stock Assessment data

FWS/ USFWS – United States Fish and Wildlife Service

GIS - Geographic Information System

GSS – General Success Score

HPI – Habitat Productivity Index

IBI – Index of Biotic Integrity

ILF - In-Lieu Fee program

MCMC – Markov Chain Monte Carlo simulation

NGO – Non-Governmental Organization

NNL – No Net Loss

PI(E)CO – Population; Intervention/ Exposure; Control; Outcome

RAM – Rapid Assessment Method

RIBITS - Regulatory In-lieu Fee and Bank Information Tracking System

UCA – Unreleased Capacity

UN – Un-avoided Losses

USACE - United States Army Corps of Engineers

Wr – relative Weight

YAC – Yearly Added Capacity

YOY – Young Of Year

ΔB – Biomass change

Chapter 1 - General Introduction: Compensatory mitigation aiming for No Net Loss – *Offsetting approved harmful anthropogenic impacts in the 21st century through the mitigation hierarchy*

Habitat, ecosystem service and biodiversity loss linked to anthropogenic influences

Habitat loss because of negative anthropogenic impacts has been attributed as a main driver for global biodiversity loss and impairment of ecosystem function (e.g., Ferreira et al. 2017; Kareiva & Kareiva 2017; Reid et al. 2019; Sutton et al. 2016). Other major drivers affecting terrestrial and aquatic species are competition and displacement through invasive species, pollution, and overharvest (e.g., Doherty et al. 2016; Janse et al. 2015; Reid et al. 2019; Schipper et al. 2019).

To slow current negative impacts and prevent future losses, conservation policies have become part of legislation in many countries (e.g., Vaccaro et al. 2013; Williams et al. 2012). Since ecosystem services are essential for society and overall human well-being, developing a sustainable and responsible approach on resource and land-use has become a priority (e.g., Bennett et al. 2015; Bringezu & Bleischwitz 2017; Haines-Young & Potschin 2010).

Mitigating negative impacts

Negative impacts exerted through land-use and development projects can be of permanent or temporal nature as well as potentially being avoided altogether through appropriate project planning and approval processes (e.g., Kiesecker et al. 2010; Phalan et al. 2017). This has led to the establishment of the mitigation hierarchy (e.g., Arlidge et al. 2018; Kiesecker et al. 2011; Phalan et al. 2017; Figure 1.1). The mitigation hierarchy requires proponents of development projects that would otherwise affect ecosystem services, physical habitat area or biodiversity in a negative way, to explore alternative options to avoid, minimize or reverse negative impacts first (e.g., Arlidge et al. 2018; Kiesecker et al. 2011; Phalan et al. 2017; Figure 1.1A). Authorized unavoidable negative impacts that remain after the previous three steps are required to be compensated for through an environmental offset to reach Not Net Loss (NNL) of ecosystem service, habitat area or biodiversity (Gibson 2016; Figure 1.1A). This offset can be at the same location of the original impact (onsite) or at a different location (offsite) and can provide similar ecosystem aspects and gains that were lost in the development project (in-kind) or aspects that target different ecosystem aspects and gains (out-of-kind; Bull et al. 2014; McKenney &

Kiesecker 2011; Quétier & Lavorel 2011). An example here is from Uzbekistan where offsetting was used to compensate for negative environmental impacts through gas extraction. The study considered different approaches to calculate required gains for in-kind and out-of-kind offsets. In-kind efforts consisted of vegetation restoration and out of kind efforts were comprised out of protecting fauna from poaching (Bull et al. 2014). These very different types of offsets showcase how in, and out-of-kind measures are possible, and could provide very different gains while still aiming at NNL. Being proponent-led, alternative offsetting mechanisms like habitat banking exist, transferring responsibility for offset establishment and management to a third party through a credit system, responding to habitat or biodiversity values determined by the regulator (e.g., Burgin 2008; Fleischer & Fox 2021; Wende et al. 2005).

Issues and limitations of offsetting

While widely adopted and increasing popularity both from a scientific as well as regulator and proponent standpoint (Figure 1.1B) offsetting still struggles with a wide variety of issues and potential challenges that not only concern the on the ground implementation and provided gains by the physical offset but span every aspect of the offsetting process (e.g., Apostolopoulou & Adams 2015; Bull et al. 2013; Gonçalves et al. 2015; Maron et al. 2016; McKenney & Kiesecker 2009; Figure 1.2A). Issues concerning offsets can be divided into two groups. The first group relates to legislative issues like policies not being approved due to economic and political stakeholder conflicts or permits using insufficient guidance and complicated language, leading to non-compliance by the proponent (McKenney & Kiesecker 2009; Keane et al. 2008; Theis et al. 2019). The second group of current and potential issues concerns the physical offset. Lack of knowledge or improper methods can lead to benefit loss over time or complete failure of the offset (Apostolopoulou & Adams 2015; Josefsson et al. 2021; Maron et al. 2012; Quétier & Lavorel 2011). Monitoring timeframes and methods are often inadequate and recorded and reported data of poor quality making it difficult to evaluate project success or determine if offsets meet ecological equivalency (Hill et al. 2013; Maron et al. 2012; Morris et al. 2006). These legislative, conceptual, and practical issues and concerns need to be better addressed to ensure that offsetting becomes a more sustainable practice, achieving NNL, providing long-term gains in perpetuity while incorporating changing global drivers relating to climate as well as society (Bull et al. 2013; McKenney & Kiesecker 2009; Norton & Warburton 2014).

Thesis structure and objectives

In this this thesis we address issues from both sides of the offsetting cycle (Figure 1.2B).

Chapter 2.1 specifically addresses the question of whether strong regulatory compliance by offsetting project proponents leads to high ecosystem function in said projects through a systematic review spanning multiple countries. It is commonly accepted that compliance ensures effective conservation but thus far has been evaluated independently and could potentially lead to biased evaluations on the current effectiveness of offsetting (e.g., Arias 2015; Keane et al. 2008; Solomon et al. 2015).

Chapter 2.2 aims to provide advice on how to address the issue of chronic and residual harm in aquatic ecosystems that can occur after a harmful impact has been approved and was offset for. We used a meta-analysis approach to identify commonly used methods and approaches and determine their benefits as well as shortcomings and possible application scenarios, monitoring timeframes, costs, and unintended impacts. Making residual and chronic harm part of current legislation and providing guidance on best practice is an important step to allow proponents to incorporate adequate measures and steps during the planning process and into long-term management plans. Proactive adaptive management built on the provided advice could reduce future harm fostered by the current reactive approach (e.g., Barnthouse et al. 2019; Mamo et al. 2019; Rood et al. 2005).

Chapter 3.1 addresses habitat banks (conservation and mitigation) in the United States and their effectiveness in reaching NNL both in terms of required area ratios and ecological equivalency. Studies on a regional or local scale suggest high compliance rates for area ratios but question the true ecological value provided by banked areas compared to habitat area or ecosystem function lost in the approved development impact (e.g., Reiss et al. 2009; Zu Ermgassen et al 2019). We base our analysis on a centralized database used for banking data collection to bridge the gap between ecological and ratio-based metrics.

Chapter 3.2 focusses on the supply and demand side of habitat banking in the United States. While widely established, no comprehensive supply and demand evaluation for both mitigation and conservation banks has been done for the United States, with regional and local studies introducing unaddressed bias. The current knowledge on banking data leaves overall trends for banked area and bank numbers out of context of potential regional and method specific bottlenecks (e.g., Poudel et al. 2018; 2019). We aim to address this issue in Chapter 3.2. by using forecast modelling to predict future supply and demand trends across different bank types while using spatial mapping to identify current reserves and bottlenecks within the United States

. **Chapter 4.1** is testing for effects of Coarse Woody Habitat introduction on fish communities in a northern boreal lake. Habitat enhancement has been proven to be an effective approach for offsetting (Roni et al. 2008; Theis et al. 2019). The northern boreal is exposed to a multitude of development impacts in many countries, mainly natural resource extraction, which require large scale offsets (Audet et al. 2015; Leppänen et al. 2017). Newly created aquatic systems in these areas like streams and lakes could benefit from structural enrichment through Coarse Woody Habitat introduction (Nagayama & Nakamura 2010; Sass et al. 2019). We investigate enhancement effects through multi-year fish surveys as well as recording structural integrity of constructed habitat while making use of Bayesian models to address common temporal and spatial bias of monitoring studies in remote systems (Bilby et al. 1999; Hooten & Hobbs 2015).

Chapter 4.2 builds on the same enhancement efforts as Chapter 4.2, measuring aquatic macroinvertebrate and macrophyte changes in response to the mentioned habitat enhancements. We use Beta-diversity components to capture community changes on a higher resolution compared to Alpha diversity. Identifying drivers for community changes is becoming more and more important to accurately evaluate restoration and enhancement projects, often missing community changes due to the use of Alpha diversity measures, skewing project results and success ratios (Bennett & Gilbert 2015; Boreo & Bevilacqua 2014). Beta-diversity in combination with considerations like wood regime and fetch can provide in depth information for current and future proponents of offsetting projects in the northern boreal on what benefits the easy to implement and readily available Coarse Woody Habitat can provide and how to best monitor these benefits over time (Marburg et al 2006; Paillex et al. 2013).

Chapter 1: Figures and Tables

Chapter 1: Figures

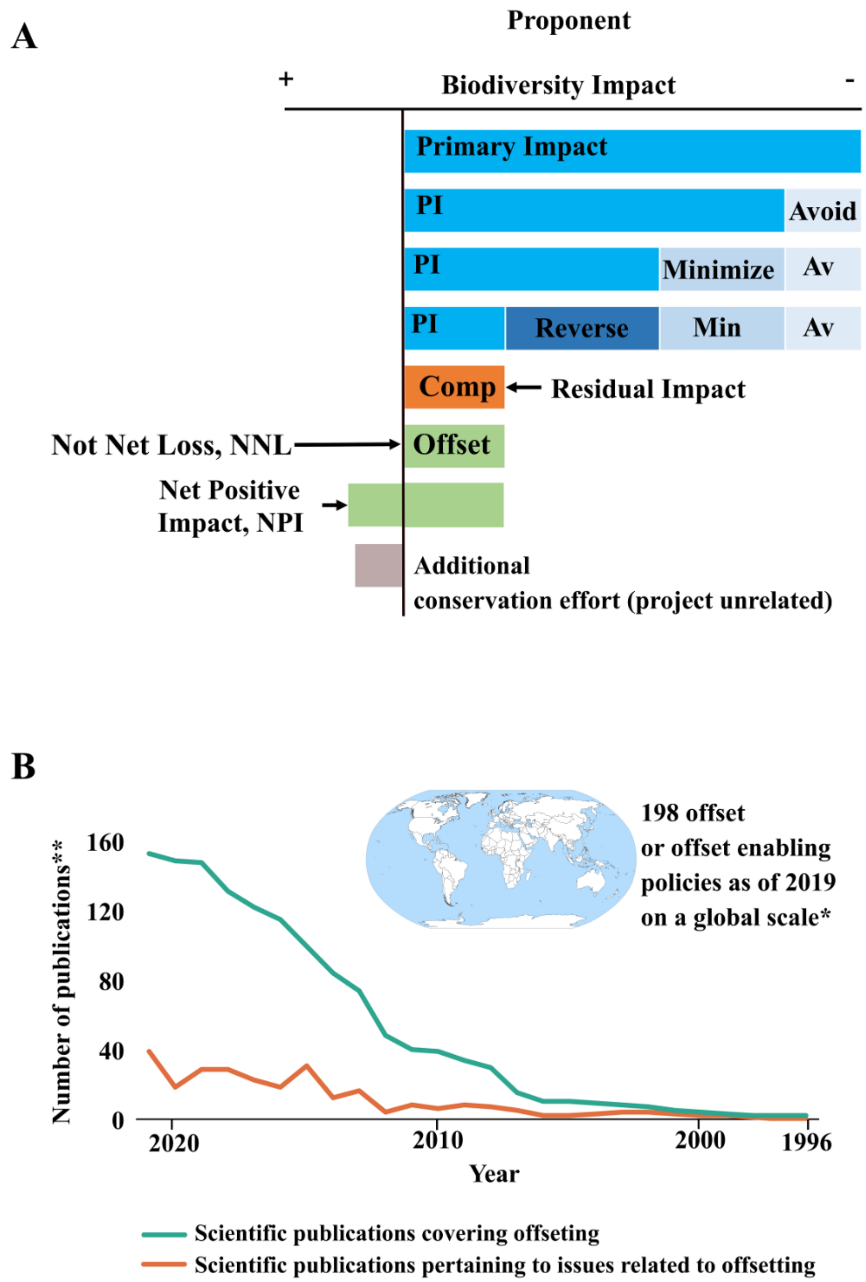
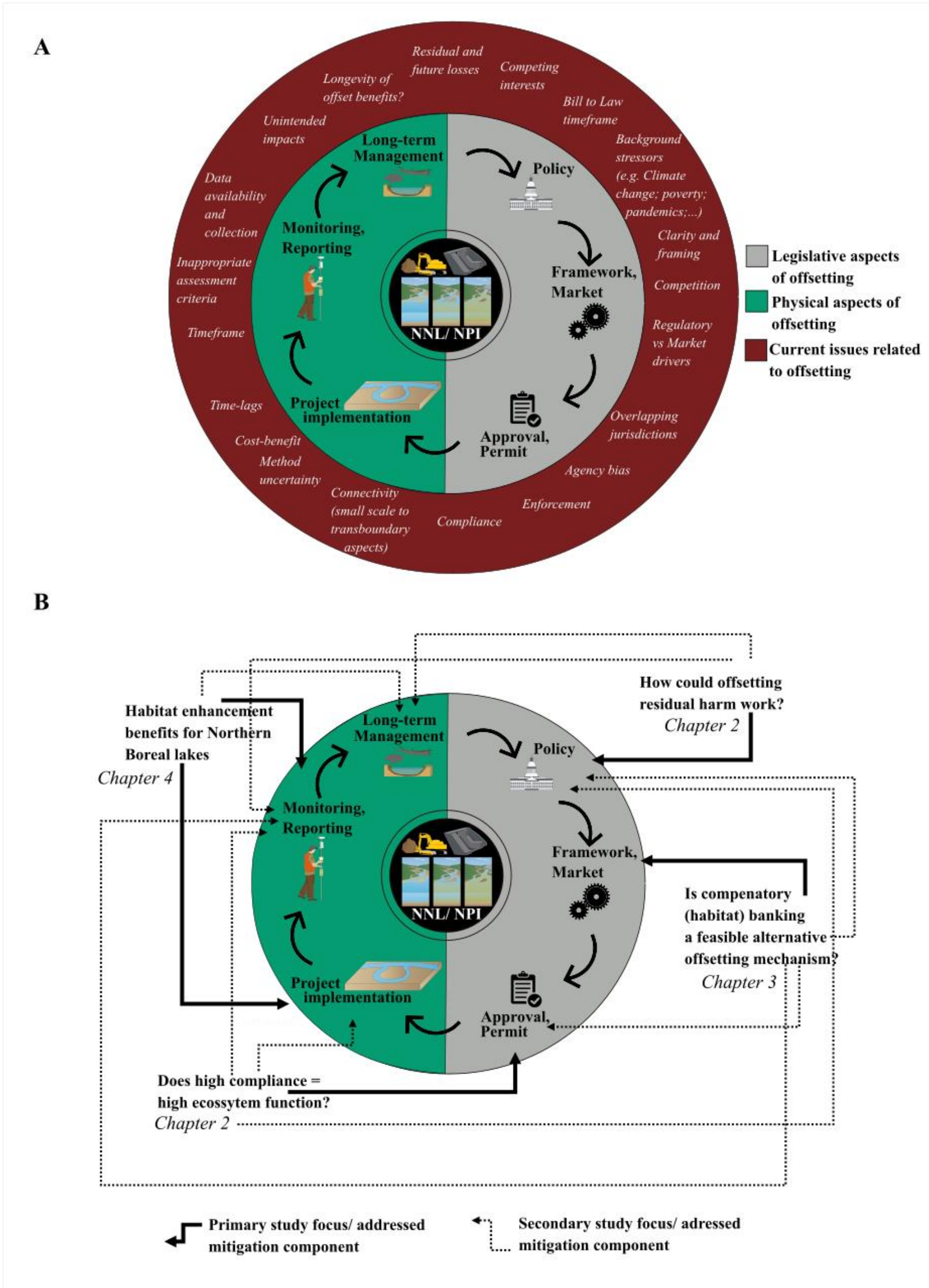


Figure 3.1. *(A) Mitigation hierarchy allowing for approval of harmful environmental impacts because of anthropogenic development projects after following the previous steps of impact avoidance, reduction, and reversal (adapted from Kiesecker et al. 2011; Ten Kate et al. 2018). (B) Peer reviewed scientific publications pertaining to compensatory offsetting practices (green) and issues associated with offsetting (red) from 1996 to 2022 (based on data from Web of Science and Global Inventory of Biodiversity Offset Policies (GIBOP; IUCN).*



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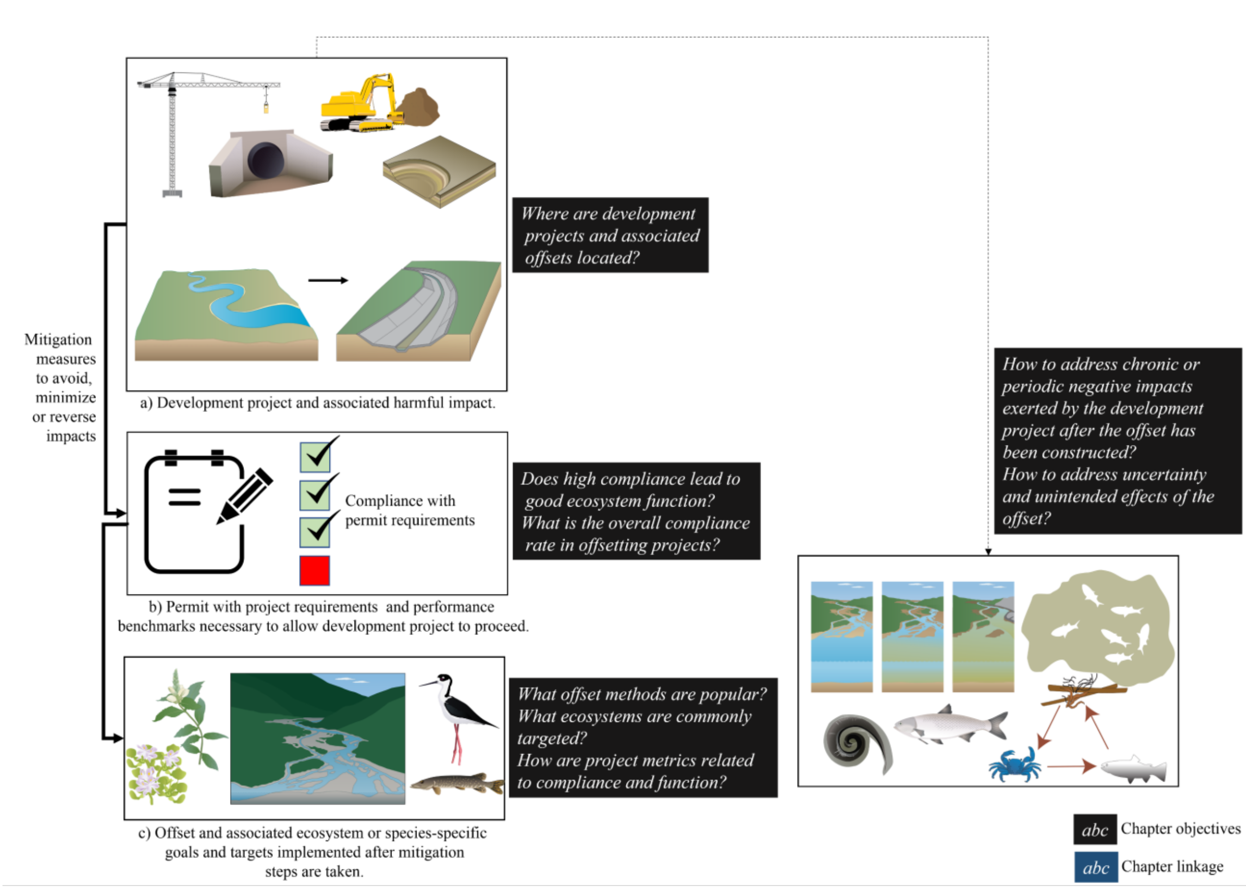
Chapter 2: The offsetting framework - *Systematic review on offsetting compliance and success and implications for post-offsetting issues and project implementation*

Graphical synopsis of key background concepts, terms, and Chapter goals



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Chapter 2.1: Compliance with and ecosystem function of habitat offsets in North American and European freshwaters

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Summary

Land-use change via human development is a major driver of biodiversity loss. To reduce these impacts, billions of dollars are spent on habitat and biodiversity offsets. However, studies evaluating offset project effectiveness, looking at components such as the overall compliance and function of projects, remain rare. Here, we conduct a scientific synthesis assessing offsetting projects in freshwater ecosystems. We reviewed 577 offsetting projects, which included four basic metrics: project size, type of aquatic system (e.g., wetland, creek), offsetting measure (e.g., enhancement, restoration, creation) and an assessment of the projects' compliance and functional success. We found that despite considerable investment in offsetting practices, there are still crucial key issues that persist. While compliance and function were related to each other, a high compliance score did not guarantee a higher degree of function. However, function did improve with larger projects, specifically when projects targeted productivity or specific habitat features, and when multiple complementary management targets were in place. Lastly, restorative measures were more likely to achieve high function scores than creating entirely new ecosystems. Altogether these relationships highlight specific ecological processes that may help improve offsetting outcomes for the conservation of habitat and biodiversity.

2.1.1 Introduction

The worldwide loss of habitat, especially over the last century, has led to a steady decline in biodiversity (Mccauley et al. 2015; Sala et al. 2000). One of the major drivers of habitat loss and biodiversity declines is land-use change, whether urbanization or resource extraction, due to economic development (e.g., Sala et al. 2000; Vörösmarty et al. 2010). When impacts to biodiversity or habitat cannot be avoided or mitigated, environmental offsets can be used to preserve ecosystem function and services. The offsetting principle is founded in the No Net Loss (NNL) framework, which aims to counterbalance biodiversity and ecosystem service loss linked to economic development (Maron et al. 2018). Offsetting is the last step in the mitigation hierarchy, such that it is utilized only after avoidance, minimization of harmful impacts or rehabilitation of the affected ecosystem following exposure, have been ruled out or deemed impossible to achieve (Figure 2.1.1) (McKenney & Kiesecker 2009).

Freshwater ecosystems are not immune to anthropogenic impacts and can be sensitive to broad spatio-temporal scale developments (Dextrase & Mandrak 2006; Dudgeon et al. 2006; Sala et al. 2000; Vörösmarty et al. 2010). Freshwater fishes have the highest extinction rate among vertebrates in the 20th century (Burkhead 2012). Besides overexploitation, losses to habitat or habitat alteration represent a major threat faced by aquatic species (e.g., Dextrase & Mandrak 2006, Dudgeon et al. 2006). Habitat loss can occur through destruction or degradation of habitat and habitat features, or alteration of physical properties like flow regime or pollution (Dudgeon et al., 2006). Offsetting can provide one solution to these threats and is an often-utilized conservation tool for aquatic ecosystems, most commonly addressing one of four major project targets: species productivity, basic function, habitat, or preservation (banking), while using methods like creation, restoration, or enhancement of habitat (e.g., Cahill et al. 2015; Reiss et al. 2009).

Offsetting is not a new concept, as it has been used for decades in some regions (e.g., United States since ca. 1972), but in many parts of the world it remains a novel and often experimental approach with high uncertainties regarding its effectiveness and feasibility (Curran et al. 2014; Moilanen et al. 2009). Generally, offsetting is implemented in mandated frameworks such as the Water Framework Directive (European Union) and Habitats Directive (European Union), Canada's Fisheries Act, Wetland Mitigation and Banking Policy (United States), Australian Offset Policies, Brazilian Industrial and Forest Offsets, and linked to regulatory requirements imposed on proponents where development impacts ecosystems (Goodchild 2004; McKenney & Kiesecker 2009). In the absence of a global policy, each country has its own approach to conserving habitat and biodiversity; several offsetting approaches exist in different parts of the world (e.g., Ambrose 2000, Bull et al. 2013; Maron et al. 2015). However, evaluations of projects and regular monitoring programs beyond mandatory requirements are rarely conducted, making it difficult to evaluate actual offsetting success (Gonçalves et al. 2015; Horack & Olsen 1980; Roni et al. 2008). Offsetting projects also rely on the proponents' compliance with required measures. Previous work has highlighted that proponent compliance in offsetting projects is generally poor and monitoring data can often be superficial and seldom span an adequate time to conduct scientifically rigorous quantitative assessments (Harper & Quigley 2005a; Quigley & Harper 2006a; 2006b; Tischew et al. 2008). Additionally, long-term success rates and efficacy of aquatic offsetting projects remain largely unevaluated or are misjudged, which makes it difficult to further develop and adapt the planning process for future or ongoing projects (e.g., Tischew et al. 2008; Zedler & Callaway 1999).

Given the importance and popularity of offsetting coupled with high uncertainty about offsetting projects, we conducted a global synthesis to determine if aspects of aquatic offsetting projects are related to project compliance and ecosystem function. Specifically, we aimed to (1) determine if there is a relationship between compliance and function for offsetting projects and

(2) assess whether there were any trends across regions, scale of the project, project targets, methods used for offsetting, and ecosystem types in terms of compliance and function. Evaluating the compliance and function relationship will help determine if high compliance in projects consequently leads to good ecosystems function or vice versa and how permits might need to be adapted in the future. Identifying potential trends can highlight deviations between permit related (policy) and ecosystem function related (condition-based) assessments (Kozich & Halvorsen 2012).

2.1.2 Methods

Projects included in this synthesis were collected through a literature search in peer-reviewed and grey literature. The search was done using the PICO (Population, Intervention, Control, and Outcome) principle (Davies 2011). Categories were:

- **P**opulation (or ecosystem type): Offsetting projects in aquatic freshwater systems.
- **I**ntervention: Offsetting through creation, enhancement, or restoration.
- **C**ontrol: Presence of clearly stated goals or requirements for the respective project.
- **O**utcome: Monitoring or Evaluation of project success regarding official permit or stated goals.

Projects were screened using a set of defined screening criteria. An in-detail synthesis protocol can be found in the Supplementary synthesis protocol covering all major steps used to collect literature and synthesize relevant data, like Boolean operators and accessed databases. Projects included in the study had to be associated with freshwater projects. Marine projects were not considered due to high uncertainties and difficulties in evaluating their success (Bayraktarov et al. 2016; Powers et al. 2003). Furthermore, projects had to include a clearly defined target and evaluation process like checking stated targets though validated assessment methods, where overall offsetting success can be determined. Projects also needed to have offsetting as the main project goal by replacing, enhancing, or restoring impaired ecosystems or ecosystem aspects. Here project effectiveness and success had to be determined by a) meeting regulatory and legally binding requirements (compliance) and/ or b) ecosystem function (function) compared to reference systems or pre-construction assessments. Checking for inclusion and exclusion criteria yielded 51 usable records, which produced 637 single offsetting projects (see supplementary material references and protocol for critical appraisal). Some records produced more individual project files than others (see Supplementary Material). Projects were distributed across 27 countries and 5 continents (Figure S2.1.2), though most (98.4%) were in the United States (65.1%), Canada (13.5%) and Europe (19.8%; Figure S2.1.2).

Evaluation of project compliance and function

After compiling project characteristics such as location, project size, project targets, and implementation methods, we excluded projects outside the three main geographic areas (low sample size; n = 10 projects). We then investigated compliance and/or function metrics for each individual study. For compliance, this was generally based on legally binding permit requirements. Most of compliance criteria fell in one of the following categories: size of offset (area), species biomass/productivity, special habitat, or biological requirements (e.g., invasive species), and/or preservation (e.g., compensatory banking), and sometimes implementation of monitoring programs. With habitat and biodiversity banking, proponents offset the expected ecosystem impact by purchasing a certain number of credits from a habitat bank (Burgin 2008). While habitat banking is often mentioned alongside traditional offsetting schemes, it does not follow the same principle since most banks preserve already existing habitat. In policy and practice, this is not the same as enhancement, restoration, or creation, which presents a different philosophy in the initial goal of achieving not net loss. Due to these differences and a smaller sample size of banking offsets (n = 50; all in the United States), we excluded banking projects from the analyses (final n = 577).

There were a multitude of compliance related metrics and targets used to stipulate desired offsetting outcomes. Firstly, offsetting area requirements (commonly North America) entails the physical area that must be replaced to adequately offset the expected habitat loss. Offsetting area was generally specified in the permit and evaluation files, obtained from government agencies, which allowed for comparison (See synthesis protocol and records list for data origin). Secondly, many permit requirements explicitly state the replacement target of lost species biomass and productivity (e.g., No Net Loss/ NNL; Harper & Quigley 2005b). Calculations for biomass and productivity are commonly done under the premise of maximum ecosystem natural capability (Langton et al. 1996; Minns 1997; Stebbing 1992). Thirdly, special requirements can often be found in offsetting permits, where conditions ranged from the construction of a specific habitat to the reduction of invasive species. Lastly, compliance for some projects was also linked to monitoring programs. In those cases, a post-construction monitoring program was to be set up and with data collection issued, through subcontractors.

Like compliance, function is a broad term, defined by an overall assessment (i.e., Rapid assessment methods, RAM), single factors (e.g., water quality), Not-net-loss (i.e., NNL of species diversity), and/or enhanced habitat features (e.g., shelter construction in spawning area). Function was assessed by government agencies, researchers associated with the project, or it was evaluated as part of an independent scientific study. Rapid assessment methods (RAMs) were often used to quickly assess a broad array of ecosystem functions for wetlands, providing a summary measure of an ecosystems' state (Carletti et al. 2004; Fennessy et al. 2007). Secondly, function was also assessed on single factors like hydro-geomorphological aspects (e.g., flow velocity of a river after rip-rap construction) (e.g., Brinson 1998), or specific chemical processes (e.g., nitrogen retention in restored wetlands) (e.g., Craig et al. 2008). Thirdly, species-dependent ecosystem functionality was mostly measured through biomass and productivity replacement. Finally, ecosystem function was also evaluated by improving habitat to enhance the ecosystem. In this category, constructed habitat features were assessed regarding their integrity and benefit provision.

A common metric for compliance and function

To allow for objective comparisons between projects, we converted project characteristics into common metrics. Characteristics such as project size, management target (e.g., Habitat-based, Productivity-based), utilized methods (e.g., restoration, enhancement), location and ecosystem type (e.g., wetland; lake) were compiled. Projects that had a two-dimensional footprint were all converted into hectares. In contrast, most of the riverine project information (streams and rivers separated by stream order (stream $\geq$ 6 river < 6; see synthesis protocol) were provided on a one-dimensional scale (e.g., enhancing 500m of river stretch) and converted to km. Project types were then classified into small, medium, or large projects (small: < 0.5 ha/km; medium: 0.5-5 ha/km; large: >5 ha/km). Furthermore, each project had one or multiple project targets assigned to them (Habitat-based, Productivity-based and Basic-Function-based), in correspondence to the source material.

Compliance and function scores were converted into integer scales. Compliance scores ranged from 0 to 3, with 0 indicating non-compliance (0-25% of requirements met), 1 indicating partial compliance (25-90% of requirements met), 2 indicating full compliance (90%-110% of requirements met) and 3 indicating over-compliance when compliance exceeded the stated goals (>110%). A similar scoring system was developed for function, ranging from 0 to 2. 'Over-functionality' is never considered in project assessments because maximum ecosystem function unlike compliance cannot be surpassed. A score of 0 indicates non-functionality with little to none (0-25%) of the ecosystem properties functioning as desired. Partial functionality was labelled with a 1 (25-90%), and a score of 2 was given for full functionality, in terms of meeting declared targets (>90%). The large margin for partial compliance and function (25-90%) was chosen due to high project uncertainty and variation in requirements and assessments across projects (see synthesis protocol and appendix literature list). Additionally, an error margin of 10% was included which is applied in most permits (e.g., meeting 90% of the permit requirements or assessment criteria was accepted as full compliance/ full functionality).

The following example from an official evaluation report (Ambrose et al. 2007) provides an example of how the available project information was translated into compliance and function scores (Figure S2.1.3). The extension of the Newport Coast Drive in California, led to the functional loss of 0.58 hectare of wetlands. The official permit required the creation of 2.30 hectare of new wetlands and revegetation with native plants. Compliance results showed that 2.42 hectare of new wetlands were created, and the revegetation of native plant species was successful. The newly created wetlands were 105% of the required offsetting project size and fell into the 90-110% margin. This project received a score of 2, meaning 'full compliance'. Revegetation, being stated as successful, received a full compliance score as well. The mean compliance score amounted to a total of 2, deeming the project as fully compliant for the analysis. An independently conducted scientific evaluation of the same project was assessed for ecosystem function using a RAM. The official RAM score was 63.19 out of 100. This score indicated that not all assessed ecosystem aspects were functioning as required to reach full function (>90). Accordingly, the project achieved 'partial functionality' and a score of 1. This scheme was applied to all 577 projects. We further provide two sample assessments in Figure S2.1.3.

Data analyses

First, we determined if ecosystem function scores were dependent on the assessed level of compliance (Table S2; Appendix) with a permuted ANalysis of VAriance (perANOVA) from the *Car* package for R software (Fox et al. 2017) and pairwise *t*-tests to identify individual significant effects. perANOVA was chosen because it has a non-parametric design and the data for compliance had a non-normal distribution (Andersen 2001). Adjustments for multiple testing were completed following Holm (1979). The Holm (or Holm-Bonferroni) correction counters the possibility of under-claiming significant pairs and groups (Aickin & Gensler 1996; Holm 1979). Results were presented as a two-way frequency bar graph underlining the integer ratings of the common metric for compliance and function.

Second, we determined if compliance and function differed based on scale of the project and the three possible project targets (Habitat, Basic-Function and Productivity; see Table 1 for descriptive statistics; mean ± SD). For this we used a permuted linear model and perANOVA. The permuted linear model was first completed for possible significant influences of project location (country), system (river, stream, lake, and wetland), project size (small, medium, large), number of project targets, and number of offsetting methods used, on the response variables compliance and function. The perANOVA was completed for the linear permuted model testing for all five factors and possible interactions (i.e., Compliance (Function) ~ Location * System * Scale * Method-Number * Target-Number) ('Car' package; Fox et al. 2017). The permuted design was chosen and applied to both the compliance and function model to generate comparable means (Anderson 2001). A Scheffé-Test for compliance and function was conducted *post-hoc* to determine significant differences for pairwise comparisons (Scheffé 1960). Results were presented in cumulative percentage bar graphs to stay truer to the original integer ratings.

Finally, a second permuted linear model and perANOVA was used to investigate the effect of specific project targets (Habitat; Productivity; Basic-Function) in conjunction with type of offsetting method (Creation; Restoration; Enhancement) had on project outcome (positive or negative) and visualized using stacked bar graphs. Additionally, proportional project distribution across aquatic systems (Rivers; Streams; Lakes; Wetlands) was calculated for Canada, Europe, and the United States. Analyses were completed using R statistical software (R core team 2020, Packages used: *Car, lsmeans, multcomp, ggplot2, tidyr, gridExtra, dplyr*). For a cutoff level for significance the null hypothesis was rejected when $p < 0.05$ and not rejected when $p > 0.05$ (Results reported as $p < 0.05$ = between 0.001 and 0.05) and results lower than 0.001 stated as < 0.001).

2.1.3 Results

Cross country relationships for compliance and function

Function increased gradually with higher compliance scores ($p < 0.001$) until leveling out when reaching over-compliance (score 3; $p = 0.53$; Table A2.1.2; Figure 2.1.2). Overall, function scores were lower than compliance scores (Table 2.1.2). Function increased with project size ($df = 2$; $p < 0.05$; Figure 2.1.2; Table 2.1.2). Location, system, scale, and number of project targets, but no interaction terms, had significant influence on the function scores ($df = 2$; $p < 0.05$) for offsetting projects (Table 2). Location and system, but no interaction terms, were also significant factors for compliance ($df = 2$; $p < 0.05$).

Compliance analysis

Among ecosystem types, compliance was the highest for river and lake projects (2/3 of projects had full or over-compliance, 1.81 ± 0.78; 2 ± 0) and were significantly higher compared to wetland projects (1.38 ± 0.98; Figure 2.1.3A). Approximately 50% of wetland projects only achieved compliance levels of 0 or 1 (Figure 2.1.3A). Projects incorporating streams did not differ significantly from the other systems (Figure 2.1.3A). Mean compliance was not significantly different among project scales and targets. Regionally, Canada had the highest compliance scores (1.78 ± 1.18; 25% of projects were over-compliant), which were significantly higher than projects in the United States (1.41 ± 0.91; Figure 2.1.3A). Europe did not differ significantly in its compliance from either Canada or the United States, though having higher compliance than US projects and the fewest projects with a compliance score of 0 (1.71± 0.62; Figure 2.1.3A).

Function analysis

Riverine projects had the highest function scores (1.55 ± 0.6; ~60% achieved full function score) and were significantly higher than stream and wetland projects (1.28 ± 0.6; 1.05 ± 0.65; Figure 2.1.3B). Projects in Canada and Europe had significantly higher functionality (1.59 ± 0.51; 1.55 ± 0.62) than projects located in the United States (1.07 ± 0.44; Figure 2.1.3B). Lakes had high function scores associated with offsetting projects as well (1.5 ± 0.55), but a low sample size (n = 6). Small projects had significantly lower function scores (1.09 ± 0.63; 20% with ecosystem function 0) than medium or large projects (1.28 ± 0.62; 1.4 ± 0.69; Figure 2.1.3B). Mean function scores increased significantly with the number of management targets (Figure 2.1.3B).

Function was lowest for projects with a single target (1.10 ± 0.65) and increased in projects with two (1.35 ± 0.61; Figure 2.1.2B) or three (1.79 ± 0.41; Figure 2.1.3D) different targets.

Specific management targets and methods

Both compliance and function were significantly higher (1.75 ± 0.69; 1.53 ± 0.64; $p < 0.05$) in projects that took a productivity approach compared to those without (1.41 ± 0.95; 1.13 ± 0.64; Figure 2.1.4A; B). Habitat-based project approaches also had a positive influence on function ($p < 0.001$), with over 60% of these projects achieving full ecosystem functionality (Figure 2.1.4A). Projects focusing on basic function replacement had a lower proportion of projects in the higher compliance and function levels and consequently lower overall mean compliance (1.43 ± 0.02 vs 1.77 ± 0.75) and function scores (1.2 ± 0.65 vs 1.39 ± 0.69; Figure 2.1.4A; B). Including restoration measures in a project increased the mean function score (1.38 ± 0.64 vs 1.06 ± 0.64) of projects compared to projects without restoration ($p < 0.05$). Projects using any form of habitat enhancement were less likely to be non-compliant (Level 0; 12% vs 22%; $p < 0.05$). Lastly, function (1.05 ± 0.61 vs 1.37 ± 0.66) and compliance (1.37 ± 0.98 vs 1.64 ± 0.77) were significantly lower ($p < 0.05$) for offsets that created entirely new habitat or whole ecosystems (Figure 2.1.4A; B) than ones restoring or enhancing existing habitat or ecosystem features.

Canada, the United States and Europe differed in their project management targets and implemented methods. Enhancing existing ecosystems was equally used in the US and Europe (Figure 2.1.5B1; C1). Habitat creation was used mostly in Canadian and US projects (57.8%; 62.8%; Figure 2.1.5A1; B1), whereas habitat creation represented a minority of European offsetting projects (14.3%; Figure 2.1.5C1). We also found that 33.3% of US and 41.9% of Canadian projects implemented habitat restoration measures (Figure 2.1.5A1; B1), while restoration efforts were widespread in Europe (92.1% of projects, Figure 2.1.5C1). Function-focused approaches were present in 95.6% of the US projects, 72.2% of Canadian projects and 51.1% of European projects. Productivity and habitat were equally considered in around half of the Canadian and European projects, but in less than 10% of US projects (Figure 2.1.5A1; B1; C1).

Offsetting ecosystem

Canadian projects were most associated with running waters (33% streams; 22 % rivers), followed by wetlands (43%) (Figure 2.1.5A2). The vast majority of assessed projects in the

United States were wetland related (79%), with rivers (8%) and streams (12%) making up the remainder (Figure 2.1.5B2). European projects were predominately located in rivers (58%) and streams (28%), with only 13% wetland related (Figure 2.1.5C2). Lakes, (no usable case study data found for reservoirs), were underrepresented in all three regions (<2%).

2.1.4 Discussion

Many projects in this study were officially labelled a success because compliance linked to legislative requirements was high. This also hints at a probable bias in the published literature towards projects that were considered successful. We tried to reduce this bias by including all available literature, ranging from official reports to later conducted evaluations. While potentially fostering increased ecosystem functionality, compliance and function are not equivalent due to different criteria and motivations in achieving ecosystem function (Kentula 2000; Kozich & Halvorsen 2012). These findings demonstrate the advantages of incorporating more ecosystem related aspects into legislative and regulatory tools to ensure proper implementation, while acknowledging apparent ecological constraints and ecosystem limitations. Our results suggest that: 1) compliance cannot be treated as equivalent to ecosystem function, 2) offsetting projects benefit from increased ecosystem function when several, complementary management targets are in place, 3) offsetting projects benefit from increased ecosystem function in larger projects, and 4) creating novel ecosystems have underestimated challenges and uncertainties, leading to a higher risk of failure.

Compliance

Offsetting compliance was affected by project system type and location (or geographic position). Lower compliance in wetlands appears to be directly related to permit goals and requirements. For example, many assessed studies in this synthesis and other literature, wetland permits often include criteria that may be difficult to achieve or underestimated dynamics, which leads to reduced compliance and increased failure (e.g., Allen & Feddema 1996; Bendor 2008; Quétier & Lavorel 2011). A similar effect was observed for criteria which did not provide the proponent with clear guidance (Brown & Veneman 2001; Matthews & Endress 2008). This may be related to a lack of knowledge and misunderstanding on the proponent's part or ambiguity within the permit and shortcomings or loopholes in the legislation or framework (Brehn & Hamilton 1996). Finally, low compliance in wetland projects may be directly linked to functionality issues with creation of new ecosystems and wetlands in general (underestimated system), and proponents being able to meet requirements (Brown & Veneman 2001; McKenney & Kiesecker 2009; Matthews & Endress 2008). Geographic dependent differences in compliance are partially a consequence of different offsetting frameworks, and partially due to regional differences in the ecosystem types used for offsetting projects. The United States had a high proportion of wetland projects assessed in this synthesis (Figure 2.1.5), which were less compliant than projects in other systems (Figure 2.1.3).

In addition to project related factors, external influences are important for compliance. Of note, many European offsetting projects are embedded in the Natura 2000 framework (Ostermann 1998: Weber & Christophersen 2002). This regulatory framework encourages restoration approaches and perpetual project duration (McKenney & Kiesecker 2009), which could ensure higher and long-lasting compliance, which aligns with goal framing theory (Etienne 2011: Sunstein 1996). While the mitigation system in the United States has a strong basis under Section 404 of the Clean Water Act, backed by the 1990 Memorandum (Hough and Robertson 2008: McKenney & Kiesecker 2009), the theory and practices differ. The equivalence, location, timing, duration, and offset ratio factors are comparable to other mitigation systems, but many states have developed their own offsetting systems and ratios, with a focus on area replacement. These between state differences (Brown & Venenman 2001; Matthews & Endress 2008) combined with administrative shortcomings (Matthews & Endress 2008; Turner et al. 2001) potentially lead to reduced compliance in offsetting projects in the United States.

Function

One of the main aspects often considered in ecosystem-based offsets is size and scale (Palmer et al. 2010; Peterson et al. 1998). Larger projects had significantly higher functionality than smaller scale projects (Figure 2.1.3). One reason for this is the inability of small systems to become resilient. For instance, if a project in a small system fails, it often fails completely, while larger systems may have greater capacity and resiliency to offset for partial loss of function (Jähnig et al. 2010; Mant et al. 2016). Also, function in small projects may be impaired by catchment related degradation and unaddressed broad-scale pressures like water quality or connectivity beyond the scope of the offsetting project (e.g., Bernhardt & Palmer 2011; Jähnig et al. 2010). Also, more detailed, and careful planning processes are often evident in larger projects (Brown &

Veneman 2001). This could explain the lower functionality of stream versus river-based projects. Offsetting functionality is also system dependent, with higher offset function in river projects and the lowest function found in wetland projects (Figure 2.1.3B). There are two possible explanations for low functionality in offsetting wetlands. First, wetland restoration or creation is an underestimated endeavor as it draws on complex interactions of landscapes, different aquatic and terrestrial microhabitats, hydrological and soil properties, a vast array of chemical processes and rarely follows the general principles of succession (Brown & Bedford 1997; D'Avanzo 1989). Further, wetland projects assessed here were often newly created ecosystems (64.3%) while riverine projects mostly relied on restoration and enhancement of existing systems. Enhancement and restoration had strongly positive effects on function, while creation led to lower functionality compared to the other two methods.

Unsurprisingly, creating a new ecosystem has a greater uncertainty than restoring an existing one. Ecosystem processes that involve nutrient cycling and food webs must be established and there is also a higher risk of introduction of invasive species during the assembly process (D'Avanzo 1989). This may explain the significant differences in success in ecosystem function between the United States and Canada/Europe, since 79% of the US projects involved wetlands with the main method being creating new ecosystems. In Europe, most projects were completed on riverine systems (86%) and relied heavily on restoration and enhancement, which resulted in good overall functionality. While Canadian offsetting projects featured significantly higher functionality than projects in the United States, they still contained a large amount of wetland related projects (43%) and the creation of new systems (62.8%). Half of the Canadian projects were focused on habitat specifications, productivity, or both. Only a minority (<10 %) of the US projects considered those approaches, while most projects were focused on basic function replacement (95.6%). Basic function replacement can be difficult because it often leaves out many species related factors, habitat features and physical interactions on an ecosystem and landscape scale (Whigham 1999). In these cases, long-term establishment of an ecosystem is still likely but will differ from natural healthy systems (Scatolini & Zedler 1996).

Lastly, having multiple management objectives increased ecosystem function. Focusing on a single target approach, such as bolstering productivity through re-established connectivity but disregarding habitat features, flow regime and other factors is unlikely to achieve full functionality for many species (Minns et al. 1996; Palmer et al. 2010). A multi-target approach would aim to reach "the least degraded and most ecologically dynamic state possible, given the regional context" (Palmer et al. 2005). A multi-target approach also holds the potential to reduce possible distortion of ecosystem productivity. For instance, high biomass by itself would not consider fish community composition and habitat suitability. A fish community with high biomass comprised mostly of low trophic level species would most likely not be sustainable over the long-term and would not include commonly desired higher trophic level target species (Carpenter et al. 2001; Gascuel et al. 2005; Ruppert et al. 2018). Thus, habitat offsetting projects appear to benefit from increased ecosystem function scores when several, complementary management targets are in place. This is consistent with recent calls for offsetting projects to include multiple management targets which may improve long-term ecosystem function (Ruppert et al. 2018).

Relationship between Function and Compliance dependency

Overall, a higher compliance score generally yielded higher ecosystem functionality. This weak relationship suggests that compliance is important, but not sufficient to achieve good ecosystem function. There also seems to be a threshold to increasing function, as over-compliant projects did not substantially increase functionality. This result is likely due to several reasons. First, there is likely to be an ecological threshold (e.g., carrying capacity) for each ecosystem, limiting the overall effect of the utilized offsetting method. For instance, a constructed spawning area for Salmonids can only raise productivity up to a certain degree depending on the dimension pf measure itself but also the ecosystem it is embedded in. Second, there may be time lags which were not considered (i.e., it might take longer for full functionality to be realized in projects; Minns 2006; Moilanen et al. 2009; Scrimgeour et al. 2014). This underlines on the one hand the need for proper long-term monitoring programs and on the other hand inclusion of the most recent scientific advancements to estimate ecosystem limitations and dynamics and develop realistic timelines (Calvet et al. 2015). Finally, many assessed projects were over-compliant in targets like project size and biomass, which do not necessarily lead to increased function. This could also underline the fact that necessary components for enhancing habitat functionality are still poorly understood (Courtice et al 2014). Higher levels of compliance could be also motivated by non-ecological drivers not assessed in this study. Those drivers are founded in strategic behavior theory, where over-compliance is often driven by competitive advantages, public image or linked to values and beliefs of upper management (e.g., Karpoff et al. 2005;

Maxwell et al. 2000; Wu 2009).

2.1.5 Study limitations

Conducting a scientific synthesis on such a large scale holds its limitations. First, pooling both peer -reviewed, and grey literature might lead to uncertainty in data quality despite critical appraisal strategies as well as to a bias depending on which government agencies did provide data. However, including grey literature offers the option to reduce the bias of peer reviewed literature mostly covering 'successful' projects. Overall using a common metric lowers information value in a tradeoff for harmonized and comparable data. Finally, there were language restrictions since offsetting projects in large parts of the world were inaccessible due to it and might hold different results than North American and European projects.

2.1.6 Conclusions

Compliance seems to be a rather well-defined measurement in the form of permit requirements, which can be potentially influenced by administrative shortcomings rather than actual project specifications. Though often including criteria linked to ecosystem function, permits rarely encompassed a holistic ecosystem assessment which made compliance a poor measurement for overall project function. Project planning and official permits should aim to encompass more ecosystem function requirements. This approach, especially when done in a more holistic manner, covering different ecosystem aspects would further strengthen the relationship between compliance and function when properly enforced and implemented. This in turn requires an increased consideration of scientific studies in advisory reports to be able to give proper advice for offsetting policies or the refinement of newer approaches like banking schemes and possible commoditization of conservation efforts (e.g., Mann 2015; Reid 2011). Ecosystem functionality can be harder to assess and evaluate since no clear guidelines exist on what should be included, and which method should be used. This uncertainty is emphasized by higher function levels being harder to achieve via creation of new ecosystems, especially wetlands, compared to restoring or enhancing existing systems. Additionally, bigger projects often hold more potential to achieve higher ecosystem function than smaller ones. Furthermore, the inclusion of multiple management targets improved ecosystem function, underlining the need for more ecosystem-based approaches in offsetting projects to ensure long-term stability and resilience. Lakes as means to offset environmental losses were highly underutilized and hold potential for future offsetting projects, especially considering the global abundance of reservoirs and abandoned mining/gravel pits (e.g., Gammons et al. 2009; McCullough & Lund 2006; Ruppert et al. 2018). Considering the variability in offsetting projects, it remains vital to increase knowledge and develop management plans on a project-by-project basis, to help develop a broader, general framework that can aid in providing guidance and support. While it is encouraging that compliance and function are positively related, policy and practice should strive to strengthen this relationship to realize long-term goals of offsetting projects, such as healthy and sustainable ecosystems.

Chapter 2.2: A meta-analysis on the effectiveness of offsetting strategies for Residual or Chronic Harm in freshwater ecosystems

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Summary

Offsetting aims to compensate for negative impacts due to authorized anthropogenic impacts. While anchored into legislation through extensive frameworks across many countries, residual or chronic impacts can occur after offset establishment for example because of the ephemeral timescale of some projects. Advice and best practice on how to approach these impacts is rare. To address this, we reviewed 30 projects based on a systematic review and meta-analysis in freshwater ecosystems dealing with residual or long-term negative impacts to provide application advice for the three main identified approaches of: habitat creation, habitat restoration, and biological and chemical manipulation. Project information was obtained from scientific databases and grey literature through Boolean search terms and web-scraping. Habitat creation projects, mainly targeting salmonids, had a pooled effect size of 0.8 and offsetting ratios of 1:5 with high biomass increases of over 1.4x compared to pre-establishment, associated with them.

Habitat restoration projects targeted a wide range of species and communities with a pooled effect size of 0.66, offset ratios ranging from 1:1.2 to 1:4.6, and biomass increases generally > 1x compared to pre-restoration. Biological manipulation had the lowest effect size (0.51) with stocking efforts being highly variable both in terms of biomass benefits and project outcomes pointing towards stocking being mostly applicable in cases of direct fish harm not related to environmental degradation or habitat loss. Many projects targeted salmonid species and application for a wider range of species needs to be further assessed. We conclude that 1) all three assessed approaches have a potential application use for offsetting Residual or Chronic Harm with approach specific caveats. 2) time to record first benefits required one to two years with time lags needing to be accounted for in the implementation and monitoring process, 3) monitoring timeframes of more than four years and conducting pre-assessments increased projects success significantly.

2.2.1 Introduction

Countering the global loss of habitat, biodiversity and ecosystem services related to anthropogenic influences has been a priority for conservation practitioners, scientists, and managers over the last few decades (Bull & Strange 2018; Evans et al. 2014). Expedited by rapid urbanization, population growth and climate change, offsetting has been developed as a mechanism to accommodate development projects while compensating for any negative impacts (Bull & Strange 2018; Damiens et al. 2021).

Offsetting is the last step in the mitigation hierarchy following the steps of impact avoidance, minimization and rehabilitation or restoration (Figure 2.2.1). The goal of offsetting is to compensate for the authorized loss of species, habitat, or ecosystem services, with the goal of achieving the objective of No Net Loss (NNL), which means providing equal or greater benefits (net positive gains) through the compensatory mitigation measures than the initial negative impact (Figure 2.2.1). Offsetting has become part of many authorization processes and policies on a global scale (over 50 countries, e.g., Fisheries Act – Canada; Natura 2000 – Europe; Clean Water Act – United States; Biodiversity Offsets Policy for Major Projects – Australia; Bull & Strange 2018; Damiens et al. 2021; Evans et al. 2014). While the mitigation hierarchy and associated offsetting frameworks are well established, outcomes largely rely on the assumption that authorized negative impacts are offset with the goal of meeting NNL requirements and provide long-term benefits (Bull & Strange 2018; Gardner et al. 2013; Figure 2.2.1). However, offsets can lose functionality over time due to non-compliance, impaired function or size of the project (e.g., Theis et al. 2019). These then un-avoided losses (UN; Figure 2.2.1) need to be offset as well and can be targeted through adaptive management approaches, long-term monitoring and stewardship programs and planning strategies (e.g., DFO 2009; Duplisea et al. 2020; Moilanen et al 2009). Another type of un-avoided losses, Residual or Chronic Harm can further impact a population or ecosystem being exposed to an authorized harmful impact.

Residual or Chronic Harm are negative impacts affecting aquatic communities after the initial negative impact has been authorized and been compensated for through an appropriate offset as for example defined in Canadian policy and science advice (DFO 2019a; 2019b; Figure 2.2.1). Residual or Chronic Harm affecting fish populations can be a one-time event (e.g., spill event; Lemly 2010) or can occur over a longer period occasionally, periodically, seasonally or continuous (e.g., Horne et al 2004; Williams et al. 2005). Examples of residual or chronic harm include impingement and entrainment of fish at water intakes; displacement and stranding of fish due to flow ramping (e.g., Auer et al. 2017; Brownscombe & Smokorowski 2021; Figure 2.2.1). Harmful impacts for fish can range from direct injury like physical trauma, rupture, and damage to swim bladders or internal organs and lacerations, to indirect harm like habitat isolation, food web disruption, long-term health and reproductive effects and reduced survival for juvenile life-stages (Brownscombe & Smokorowski 2021; Horne et al. 2004; May & Lee 2004; Song et al. 2019; Williams et al. 2005, Figure 2.2.1). Residual or Chronic Harm, as per definition and legislation, needs to be compensated for (DFO 2019a; 2019b; Figure 2.2.1). Current best practice and advice on how to best quantify impacts linked to Residual or Chronic Harm, their consequences for fish populations and feasible options on how to best offset these new impacts, is sparse or still in development. Up to this point very few cases of Residual or Chronic Harm have been offset, so there are limited examples available upon which stakeholders, managers and policy makers can rely.

Objectives

The objective of this study is to develop guidance and best practices for offsetting Residual or Chronic Harm. To do this, we conducted a peer review of existing studies and a meta-analysis to help determine effective offsetting strategies. Our goal was to assess peer reviewed studies and grey literature across multiple databases and websites to gather information regarding, or potentially applicable, to Residual or Chronic Harm and extract studies and projects adhering to the Before-After-Control-Impact design (BACI). Our stated objectives are:

(1) Assess current offsetting strategies used to compensate for Residual or Chronic Harm in freshwater ecosystems.

(2) Provide concrete benefits in terms of biomass or productivity increase and effect size for different offsetting methods (BACI).

(3) Provide information on potential application scenarios for the different methods and commonly used offsetting ratios and costs as well as highlight potential dangers and pitfalls of current practices.

Ultimately, we aim to identify feasible offsetting approaches that potentially can be applied for future development projects experiencing Residual or Chronic Harm and develop guidelines for proponents and managers, as well as providing information that could help in developing policies, regulations and agency instruments accommodating potential post-project harm to fisheries and fish populations.

2.2.2 Methods & Analyses

Synthesis & literature search

Studies and projects included in this systematic review and meta-analysis were collected through a literature search both covering scientific and peer-reviewed databases as well as grey literature using Boolean operators and a web scraper. Web-scraping improves coverage of a topic since it can access a wider array of sources in a shorter amount of time as well as efficiently extract information and data. Web-scraping consequently can help to reduce evidence bias (fail to identify available data on a topic; human selection errors; R-extension; rakeR; slowraker; R-Core team 2020; Jones 2017).The search was based on the PI(E)CO (Population, Intervention/ Exposure, Control, Outcome) principle, preceded by defining of inclusion criteria and study scope and followed by the actual search, result screening, data extraction, critical appraisal and analysis and synthesis of information (James et al. 2016; Protocol in Table S2.2.1-2.2.3). Search categories were projects covering fish populations (Populations), exposed to serious or lethal harm post establishment of a required offset to compensate for negative impacts (Intervention/

Exposure). Projects and studies needed to contain information on pre-assessments (before after) or appropriate reference sites (Control) as well as information on offset impacts through recorded methods (e.g., stocking, restoration) and outcome in a tangible, fisheries related metric (productivity, abundance, presence, condition, diversity, or biomass; Outcome). Detailed screening and search criteria can be found in the supplemental material (e.g., Boolean operators and parsed sites and databases; Table S2.2.1-2.2.3). We covered 206,245 individual pages with 568 hits meeting search criteria, as well as 381 search results from scientific databases. The screening process yielded a total of 98 documents assessed for validity (Table S2.2.1; S2.2.4) and used as the main body of literature for informational value and 30 with sufficient data (in paper; in figures; in supplements, through direct contact with author or organization) to be used for further analysis (Table S2.2.2).

Recorded Metrics

Recorded project metrics were offset method (habitat creation, habitat restoration, biological and chemical manipulation) and their respective subcategories (Table A2.2.1), monitoring time (years), location (onsite, offsite), pre-assessment (yes, no), time until first benefit from offset detected (years), cost in relation to offset size (USD/m for linear offsets or USD/m^2), targeted fish species and commonly applied offset ratio (impact to offset). We further calculated three metrics: effect size (Hedge's g), Biomass increase (ΔB, unitless) and General Success Score (GSS) in R (R-Core team 2020).

Hedge's g, heterogeneity, and bias

To determine differences in offsetting strategies, we conducted a meta-analysis on the studies identified above using Hedge's g metric. Hedge's g, based on a fixed (Jackson Method) or random effects model depending on heterogeneity estimates (Hartung-Knapp adjustment for random effects model, accepted Alpha < 0.05 for significant pooled effect sizes), reflects the cumulative effect size for each offset method regarding before and after treatment effect and its respective difference (e.g., biomass for target species before and after offset; Harrer et al. 2021). Hedge's g is based on a standardized difference with pooled standard deviation and an estimator for population or treatment size. Thus, Hedge's g and its confidence intervals (CI) in our synthesis will be used as a measure for offset effectiveness in terms of provided benefits. Hedge's g was chosen over Cohen's d to account for small samples sizes ($n < 20$; Durlak 2009;

Hedges & Olkin 2014). Commonly used categories dividing effect size into 'small', 'medium' and 'large' are highly situational and investigator dependent, we therefore will discuss effect sizes in their respective contexts and meaning towards offsetting Residual or Chronic Harm (Durlak 2009). Hedge's g measures were pooled for overall method comparison and weighed through inverse variance. Heterogeneity, variance of the effect size parameters, was controlled for through tau^2 (τ^2) calculations and an Alpha of < 0.05 ('meta' – 4.19-1p; Harrer et al. 2021). Tau^2 values with confidence intervals not including 0, being significantly greater than 0 indicate heterogeneity between studies, supporting the choice for a random effect model (Harrer et al. 2021). Between studies variance was estimated through a 'restricted maximum likelihood' estimator for offset methods with high heterogeneity based on newer recommendations on precision and through 'DerSiminian-laird' estimator for cases with low heterogeneity (Veroniki et al. 2016). Funnel plots, measuring effect size against standard error (SE), were used to examine bias of the study results (asymmetry of funnel can indicate bias), estimate study power (study size) as well as to confirm heterogeneity measures (funnel plot contains 95% of the studies = homogeneity). Funnel plots rely on the assumption that treatment effect precision increases with increasing sample size (Sterne et al. 2011).

Biomass increase

Biomass increase (ΔB) is an estimate of how many kilograms of fish an offsetting project will produce annually and expressed as unit-less relative change between final biomass and initial biomass over initial biomass (Barnthouse et al. 2019). Biomass increase will be used in the context of this review as a productivity benefit estimate in addition to the effect size provided by hedge's g to allow a better comparison across methods and not just within treatments.

General Success Score

General Success Score (GSS) was used to investigate project success in terms of monitoring timeframes, pre-assessments and on or off-site location of offset. Projects with the above information available were divided into three categories corresponding to fully meeting project goals and targets (Score = 2), partially meeting stated goals and targets (Score = 1) or not meeting offset goals and targets (Score = 0). These basic numeric metrics were used to test for generalized monitoring effects on project outcomes.

Offset Method

Measures to offset fish mortality and harm according to the assessed literature fall into the three mentioned primary categories; habitat creation, habitat restoration and enhancement, and biological and chemical manipulation with several subcategories containing more detailed information on the utilized method (Table A2.2.1).

Habitat creation

Habitat creation in the assessed studies refers to the practice of creating entirely new habitat to offset fish mortality by increasing productivity, abundance, density, and fish survival. Projects applying created habitat to offset for fish mortality used off channel habitat construction to provide essential life-history components, mostly for salmonid species. Off-channel habitat can take the form of side channels, sloughs, ponds, floodplains, and wetlands (Rosenfeld et al. 2008; Table A2.2.1). Constructed side channels are normally excavated in a current or former floodplain near the main channel and can receive further enhancements through gravel addition, bank stabilization and the provision of cover. Side channels are primarily fed through groundwater sources (Roni et al 2006). For example, projects in the Pacific Northwest of North America (e.g., British Columbia, Oregon, and Washington State) often utilize groundwater fed side channels to create new spawning and rearing habitat for various salmonids to offset lost productivity and increase juvenile survival (Giannico & Hinch 2003). Overwintering pond creation often accompanies side channel construction. Off channel pond habitat is also associated with wetlands and is used as overwintering and rearing habitat for many fish species Ponds can be newly excavated or re-purposed from logging and mining activity, e.g., gravel pits and mill ponds. Off channel ponds can also be created through impoundment or re-connection of formerly isolated habitats (Roni et al. 2006; Table A2.2.1).

Habitat restoration

Habitat enhancement and restoration for projects involving Residual or Chronic Harm can be divided into four categories: 1) targeting structure and cover, 2) connectivity, 3) substrate, and 4) riparian restoration (Table A2.2.1). Adding structure and cover to existing aquatic ecosystems can take many different forms, from creating riparian cover, constructing boulder weirs, adding pools and riffles, or introducing large woody debris. Structural enrichment and their beneficial effects for fish productivity have been supported by studies such as Roni et al. (2010) or Morley et al. (2005). Connectivity and associated habitat restoration are common offsetting measures associated with fish mortality due to impingement and entrainment of fish at cooling water intakes or hydropower facilities (Barnthouse et al. 2019; Table A2.2.1). Substrate changes have occurred in a wide range of aquatic systems due to development projects like flow regulation, forest harvesting, and logging operations. Adverse effects on substrate spawners can be severe and lead to fish mortality on multiple life-stages. Substrate addition and (or) removal can have a beneficial effect for many gravel spawning species and is well supported in the literature. For instance, a systematic review of 75 studies from 64 articles conducted by Taylor et al. found that a lack of spawning substrate combined with a lack of access to suitable spawning habitat can be the main drivers for population collapse, especially for salmonid species. Spawning habitat tailored towards species-specific niche requirements can be effective in offsetting negative impacts from human development (Taylor et al. 2019).

Biological and chemical manipulation

Biological and chemical manipulation of habitats and ecosystems has been commonly used to either enhance productivity of nutrient poor systems or to control nutrient inputs and eutrophication, e.g., algal blooms (Sierp et al. 2008). It also refers to the practice of increasing fish abundance through stocking, (re)introduction, and translocation. Biological and chemical manipulation cover a wide range of options ranging from the simple addition of fish through stocking to influencing specific trophic levels or whole food webs through nutrient addition (Table A2.2.1). Stocking, (re)introduction, and transfer of fish is regularly used to mitigate losses of both recreationally and commercially important species as well as offset negative anthropogenic impacts. Cases where stocking is used to offset direct mortality are rare and most often used when the main sources of harm were from entrainment and impingement in hydropower facilities, flow regulation, and stranding events (Brown et al. 2013; Holmes 2018; Unwin & Gabrielson 2018).

Monitoring, pre-assessments and site

Monitoring timeframes and pre-assessments are a vital aspect of offsetting and mitigation projects and allow for adaptive management and evaluation of long-term success (Bell et al. 2008; Louhi et al. 2016). We related our GSS scores to monitoring timeframes (< 4 years; 4 to 6 years; > 6 years) as well as if a pre-impact assessment was done or previous monitoring data was available (yes; no). We further compared GSS scores to offset location (onsite; offsite). Differences were tested through a Kruskal-Wallis test accounting for non-normal data, following an assessment for normality and sample variance (Shapiro Wilk test; Levene test; Table A2.2.5; A2.2.6). Significant differences between groups (Alpha < 0.05), were identified through pairwise comparison (Wilcox pairwise comparison) and discussed through mean GSS scores and their standard deviations ('Car' – 3.0-11; Church & Whike 1976; Ostertagová et al. 2015).

2.2.3 Results

Habitat creation

Habitat creation, divided into side channel creation (*n* = 6, pooled hedge's g = 0.75; 95% CI [0.26; 1.25]; Table 2.2.1; Figure 2.2.2A) and pond and floodplain creation (*n* = 3, pooled hedge's g 0.92; 95% CI [-0.17; 2.01]; Table 2.2.1; Figure 2.2.2A) had a large effect size as mitigation and offset treatments related to Residual or Chronic Harm (pooled hedge's g = 0.8; 95% CI [0.47; 1.12]; $p < 0.001$; Table A2.2.2; Figure 2.2.2A). Side channel creation had monitoring timeframes of 5.7 ± 1.6 years, with mean time for first benefit recorded after 1.1 ± 0.9 years, while pond and floodplain creation had mean monitoring times of 4.9 ± 1.5 years with first benefits becoming apparent after 1.2 ± 0.6 years (Table 2.2.1). Side channel and pond and floodplain creation targeted mainly salmonid species, (Coho Salmon *(O. kisutch);* Chum Salmon *(O. keta);* Chinook Salmon *(O. tshawytscha);* Steelhead *(O. m. irideus);* Brook Trout *(S. fontinalis)),* with side channel creation costing on average 150 ± 46 USD/m^2 and pond and floodplain creation 85 ± 27 USD/m^2 for the assessed studies. Both types had similar offsetting ratios (side channel 1:5.7; pond and floodplain 1:4, Table 2.2.1) and estimated biomass increases (side channel 1.88 ± 1.01; pond and floodplain 1.47 ± 0.78, Table 2.2.1).

Habitat restoration

Habitat restoration, divided into riparian modification (*n* = 1, pooled hedge's g = 1.47; 95% CI [0.81; 2.13]; Table 2.2.1; Figure 2.2.2B), connectivity (*n* = 9, pooled hedge's g = 0.65; 95% CI [-0.47; 1.77]; Table 2.2.1; Figure 2.2.2B), structure addition (*n* = 3, pooled hedge's g = 0.51; 95% CI [-0.02; 1.04]; Table 2.2.1; Figure 2.2.2B) and substrate addition (*n* = 1, pooled hedge's g = 0.69; 95% CI [-0.16; 1.55]; Table 2.2.1; Figure 2.2.2B) had a large effect size as mitigation and offset treatments related to Residual or Chronic Harm (pooled hedge's g = 0.66; 95% CI [0.30; 1.02]; $p < 0.05$; Table A2.2.3; Figure 2.2.2B). Riparian modification had monitoring timeframes of 1.8 ± 1.1 years, with mean time for first benefit recorded after 0.8 ± 0.4 years and connectivity measures had a mean monitoring time of 4.1 ± 1.8 years with first benefits after 1.0 ± 0.6 years (Table 2.2.1). Structure addition had monitoring timeframes of 3.0 ± 0.7 years, with mean time for first benefit of 1.0 ± 0.7 years. Finally, substrate addition or removal had a mean monitoring time of 2.3 ± 1.1 years with first benefits after 1.1 ± 0.9 years (Table 2.2.1). Restoration measures targeted a wide range of species ranging from specific substrate spawners to whole community targets, (American eel *(A. rostrate;,* Coho Salmon *(O. kisutch);* Chum Salmon *(O. keta);* Chinook Salmon *(O. tshawytscha);* Steelhead *(O. m. irideus);* Brook Trout *(S. fontinalis);* Brown Trout *(S. trutta);* Yellow Perch *(P. flavescens);* White Sucker *(C. commersonii);* Lake Whitefish *(C. clupeaformis);* Walleye *(S. vitreus);* Arctic Grayling *(T. arcticus);* Eurasian minnow *(P. phoxinus);* River herring *(A. pseudoharengus)).* Riparian modification costs ranged across sites of the assessed study (68 ± 26 USD/ linear m, Table 2.2.1). Structure provision for aquatic species had mean costs of 188 ± 123 USD/ linear m (Table 2.2.1). Connectivity costs varied highly depending on the size of the connected or reconnected ecosystem (84 ± 77 USD/ m^2; Table 2.2.1). The assessed substrate addition study in relation to Residual or Chronic Harm had varying costs across sites and substrate (11 ± 7 USD/m^2; Table 2.2.1). The 4 restoration approaches differed in their mean offsetting ratios (riparian modification 1:1.2; structure addition 1:1.6; connectivity 1:4.6; substrate 1:2.1; Table 1) and estimated biomass benefits (riparian modification 0.21 ± 0; structure addition 1.62 ± 0.44; connectivity 1.24 ± 0.63; substrate 1.12 ± 0; Table 2.2.1).

Biological and chemical manipulation

Biological and chemical manipulation, divided into stocking (n = 5, pooled hedge's g = 0.28; 95% CI [-0.72; 1.27]; Table 2.2.1; Figure 2.2.2C) and nutrient addition (n = 2, pooled hedge's g 1.04; 95% CI [-0.11; 2.18]; Table 2.2.1; Figure 2.2.2C) had a smaller combined effect size as mitigation and offset treatments related to Residual or Chronic Harm as the other two categories (pooled hedge's g = 0.51; 95% CI [-0.18; 1.21]; p = 0.12; Table A2.2.4; Figure 2.2.2C). Stocking had long monitoring timeframes of 8.3 ± 9.0 years, with mean time for first benefit recorded after 1.2 ± 0.7 years, while nutrient addition had mean monitoring times of 4.0 ± 0.7 years with first benefits after 0.3 ± 0.47 years (Table 2.2.1). Biological and chemical manipulation projects targeted mostly salmonid species *(*Coho Salmon *(O. kisutch);* Chinook Salmon *(O. tshawytscha);* Sockeye Salmon *(O. nerka);* Dolly Varden *(S. malma);* Cutthroat Trout *(O. clarkii);* Alewife *(A.*

pseudoharengus); Rainbow smelt *(O. mordax);* Yellow perch *(P. flavescens)).* Stocking costs were species dependent and not readily discernible. Similarly nutrient addition costs were not available for any of the studies (Table 2.2.1). Stocking offsetting ratios were done on a 1:3.1 basis and not available for nutrient addition with estimated biomass increases of 0.84 ± 0.77 for stocking side and 2.01 ± 0.31 for nutrient addition (Table 2.2.1).

Monitoring, pre-assessment and site

Results from the review show that monitoring time is related to offsetting success when broken down into three basic numerical categories (Generalized Success Score (GSS); success = 2, partial success = 1, no success = 0) and major time increments in years (n = 30; <4, 4 to 6, >6; chi^2 = 7.59; df = 2; p-$value$ < 0.05; Table 2.2.2; Table A2.2.9), with projects having less than 4 years of monitoring showing significantly lesser GSS scores (0.86 ± 0.89) than projects having monitoring time frames ranging between 4 to 6 years (1.53 ± 0.70; p = 0.043) and lesser GSS scores than projects monitoring more than 6 years (1.30 ± 0.84; p = 0.059; Table 2.2.2; Table A2.2.9). Projects with pre-impact assessment studies showed higher success (GSS = 1.70 ± 0.57) than projects without pre-impact assessments (0.75 ± 0.89; chi^2 = 7.25; df = 1; p < 0.05; n = 27; Table A2.2.8; Table 2.2.2). The location of the offset, whether it is onsite or offsite, did not play a significant role in project outcomes measured through GSSs (chi^2 = 0.67; df = 2; p = 0.72; n = 29; Table A2.2.7; Table 2.2.2).

2.2.4 Discussion

Habitat creation application for Residual or Chronic Harm offsets

Overall, side channel creation can be an effective tool to increase fish productivity and increase juvenile survival by providing spawning and rearing habitat and thus is a suitable approach for offsetting fish mortality. Side channels are most often utilized in cases where harm stems from hydropower development projects leading to loss of connectivity as well as habitat degradation and juvenile mortality linked to a lack of rearing habitat (Scruton et al. 2005). Results from the case studies show the complementary nature of side channel and off channel pond creation. For example, in a case study from the Skagit River basin in the Pacific Northwest, populations of Coho Salmon (*Oncorhynchus kisutch*), Chum Salmon (*Oncorhynchus keta*), and other pacific salmon species declined significantly due to loss of habitat and increased juvenile mortality (Henning et al. 2006). Monitoring data for 13 years (3 to 7 years of data per basin) was evaluated with a focus on smolt density of Coho salmon, effect of project size, and offset morphology for 30 constructed and natural reference sites. Smolt density in constructed off-channel ponds approached natural reference values with 0.37 smolts/m^2 and an average abundance of 2,492 fish per site, indicating a successful offset in terms of natural productivity rates (Henning et al. 2006).

Both types of floodplain habitat construction (e.g., side channel and pond creation) increased productivity for salmonid species making newly created habitats that meet or exceed natural references, thus adhering to NNL criteria. The main drivers for productivity seem to be temperature, wetted area, and habitat heterogeneity. Greater depth and pond morphology produced larger smolts compared to channel-type habitat, with an average fork length difference of 13.3% (Roni et al. 2006). Cost benefit ratios are similar for both types. Offset size and costs increase rapidly with larger losses in biomass or productivity. Cost and space requirements may make habitat creation impractical for some mortality cases but could be used in conjunction with habitat restoration (floodplain reconnection, flow enhancement, gravel addition) or stocking.

Habitat creation as offsets for fish mortality focus mainly on salmonids with density as the primary assessment and success metric (e.g., Gibeau et al. 2020; Rosenfeld et al 2008; Scruton 2005). Harm and mortality mainly stem from habitat degradation and impingement affecting juvenile mortality, spawning success, and larval emergence (e.g., Rosenfeld et al 2008; Scruton 2005). Considering the large effect sizes in the assessed projects, habitat creation should be considered for cases of direct mortality, especially for salmonids. The main factor here was project size; 5,000 to 10,000 m^2 seems to be the optimum size given associated costs and target life stages (Rosenfeld et al. 2008). Commonly achieved offset to natural reference ratios were around 1:5 based on the assessed studies. Most ratios are applied to account for uncertainty in predicted gains or were derived from a 1:1 area replacement not accounting for higher productivity of the offset. Varying periodic harm should be handled through adequate monitoring timeframes which will also capture coinciding temporal population and ecosystem changes (Duplisea et al. 2020; Louhi et al. 2016; Moilanen et al 2009; Radlinger et al. 2019). Additional types of habitat creation and their application for offsetting mortality need to be explored for more species to inform better practices (Gammons et al. 2009). Monitoring averages are 4 to 5 years, including pre-impact assessments, with early benefits requiring at least 1 year post construction to manifest. Created off-channel habitat for salmonid species provided high biomass (ΔB) benefits (1.47 to 1.88) compared to natural reference systems. Onsite and like-for-like options are more common but offsite construction and out-of-kind offsets are possible as well through newly created habitat (e.g., Gibeau et al. 2020; Rosenfeld et al 2008; Scruton 2005). Overall habitat available information on habitat creation regarding its application towards Residual or Chronic Harm is sparse but points at the potential to utilize this approach outside of its usually salmonid focused application.

Habitat restoration application for Residual or Chronic Harm offsets

Habitat restoration, due to its demonstrated effect size, can be a highly effective measure to offset Residual or Chronic Harm. It can be applied in various scenarios and is often used by combining different restoration and enhancement measures (Barnthouse et al. 2019; Louhi et al. 2016; Roni et al. 2010). Like habitat creation, most past and recent studies were applied to cases that featured indirect harm due to habitat degradation, loss of connectivity, and juvenile mortality. However, studies like Barnthouse et al. (2019) show how habitat restoration can be utilized to offset Residual or Chronic Harm events by increasing overall habitat productivity and compensating for lost fish through quantification of equivalent biomass, habitat productivity index (HPI), or age equivalents (EA). A restoration case study from New Jersey, United States found that restoration and enhancement of a degraded salt marsh through reconnection was effective in offsetting losses of River Herring (*Alosa pseudoharengus*) due to entrainment and impingement at power generation stations in conjunction with commonly accepted impingement mitigation measures like deterrent systems, water intake regulation, and upgraded fish protection technology (Baletto & Teal 2011). The project set a 12-year monitoring timeframe to meet final success criteria which involved environmental variable thresholds like desired plant coverage, open water percentage and species abundance. Several other studies have shown the effectiveness of restoration measures linked to barrier removal and habitat re-connection, especially for migratory species (Hogg et al. 2015).

While salmonid species are the focus of most projects, habitat restoration can also provide benefits for other species. For instance, a 6-year dam removal study in the headwater streams of Shenandoah National Park, Virginia, showed that American Eel (*Anguilla rostrata*) abundance at 15 sites increased from 1.6 (± 0.825) to 3.75 (± 3.15), meeting numbers from unimpeded natural reference systems. Average length decreased in headwater locations, indicating successful passage of smaller size classes (<300 mm) (Hitt et al. 2012). These results demonstrate that the removal of a key bottleneck dam can offset chronic negative effects on American eel productivity and abundance for populations up to 150 km distance and on a landscape scale. In the case of fish mortality, it is important to be specific about like-for-like or out-of-kind replacement.

Offset ratios, benefits, and consequently sizes will differ significantly if the offset aims to meet lost biomass for a single species or for a community. Average offset ratios were often around 1:1.5 to account for uncertainties, though connectivity offsets often had higher offset ratios (1:4.6) since this type of offset, and its size, is more dependent on the environment and associated ecosystems than the measure itself (e.g., dam removal; Bradford et al. 2017; Braun et al. 2011; Scott et al. 2008). Costs varied greatly across offsets and hints at the scalable and flexible nature of restoration measures, making them feasible on a multitude of scale and appropriate in different environments (e.g., highly urbanized regions with spatial constraints; Scyphers et al. 2015; Simestad et al. 2005). Riparian restoration and structure addition is mainly assessed in restored or enhanced meters while connectivity and substrate offsets are measured in area (m^2). Substrate addition can be a cheap and effective measure when targeting species and spawning related aspects (McManamay et al. 2010). A study from British Columbia summarizing data from over 30 studies confirms the benefits of spawning gravel and linked spawning habitat for both anadromous salmonids (Coho; Chinook; Steelhead) and non-anadromous salmonids (Brook Trout; Brown Trout; Cutthroat Trout; Rainbow Trout). An 8-fold increase in gravel area led to an 88% increase in production per m^2 for anadromous species and 25 to 73% increase for non-anadromous resident species (Keeley et al. 1996). Like habitat creation, early benefits are normally measurable 1 year post construction. Overall monitoring times for restoration projects range from 2 to 4 years, including pre-assessments. Mean expected biomass benefits (ΔB) are generally greater than one, except for riparian restoration measures which usually do not target productivity directly. Variability of the described measures shows that compatible joint measures can complement each other (Barnthouse et al 2019; Bradford et al. 2017; Roni et al. 2010). Evaluated case studies show the potential of habitat restoration to offset fish mortality when losses can be translated into habitat metrics. Most monitoring assessments focus on densities and rarely biomass; this means that monitoring requirements need to be adjusted accordingly to ensure that restoration activities are effective. Offsets thus require pre-assessments and regular post monitoring to evaluate the full benefits properly. Early estimates can be derived from abundant literature and restoration studies from similar systems and species.

Biological and chemical manipulation application for Residual or Chronic Harm offsets

Nutrient enrichment can have significant short-term productivity increases in treated aquatic systems. However, most studies only suggest nutrient enrichment as an interim tool to offset nutrient deficits until natural pathways can be restored (Sierp et al. 2008; Wipfli et al. 2003). It could be suitable for situations where nutrient pathways are blocked or disrupted, or where reliant fish populations are extirpated or significantly reduced (Jarvie et al. 2013; Wipfli et al. 2003). Nutrient enrichment could be a suitable method to offset fish mortality given its mean effect size based on literature and fast response time for first benefits (immediate to 3 months) by increasing overall system productivity in systems that allow for the treatment while also being easily monitored and controlled. Complex systems and communities, both in size and species richness, are rarely suitable due to the magnitude of potential interactions. The main target species for nutrient enrichment are diadromous salmonid species (Kohler et al. 2012). Many enrichment programs are already in place which should allow for an easier implementation of nutrient enrichment for mortality offsets and falling back to recorded and established benchmarks from these studies. Nutrient enrichment can be adjusted and tailored to target life stages or important time frames during the year. Cost per area as well as benefits are highly variable and depend on the target species and enrichment intensity (Stoichiometry) but in most cases, carcasses can be readily acquired from hatcheries (Wipfli et al. 2003). Nutrient enrichment is often jointly conducted with stocking efforts. In these cases, its main aim is to increase nursery habitat productivity and increasing fry abundance through stocking (Koenings et al. 2000).

Nutrient enrichment requires extensive data on prior nutrient levels as well as system productivity. Evaluations rely on several important benchmarks to capture trophic responses and benefits of the ecosystem (primary production, secondary production, fish response). To evaluate and monitor benefits for target fish species linked to nutrient enrichment, the target level and lower trophic levels need to be monitored (Koenings et al. 2000). Enrichment effects are mostly assessed in fish growth parameters and primary and secondary production levels with monitoring time frames around 4 years to adequately capture long-term effects and seasonal variation and population dynamics. Lost fish should be translated into production foregone (biomass) that can be matched with enrichment monitoring metrics and expected productivity benefits. Potential significant community changes and situational benefits need to be considered (Jarvie et al. 2013).

Fish stocking differs from other approaches when considered as a mortality offset. Stocking does not meet the self-sustaining nature of an offset (DFO 2019b). Study results underline the inherent difficulty of using stocking as an effective offset or restoration measure. Large stocking efforts aiming to offset anthropogenic factors and mortality for diadromous species along the United States Atlantic coast have shown that stocking by itself is not sufficient due to low connectivity. Only 3% of the fish were able to complete vital passages (Brown et al. 2013). Studies and reviews from New Zealand show that stocking increased population numbers for diadromous salmonids, however, not to the anticipated degree due to significantly lower survival rates than initially predicted (Holmes 2018, Unwin and Gabrielson 2018). While survival differences between wild and hatchery fish can be considered by incorporating offset ratios (1.5 to 3), bottlenecks are often overlooked (Antonio Agostinho et al. 2010; Figure 2.2.3). For instance, stocking can rarely compensate for a lack of connectivity or degraded rearing and spawning habitat (Michaud 2000; Figure 2.2.3). Thus, stocking mainly seems to be an appropriate mortality offset when the mortality is direct and not linked to indirect sources or habitat loss or degradation (Barnthouse et al. 2019). Examples of these types of cases include one-time fish kills, losses due to entrainment and impingement, or stranding events through flow regulation (Young et al. 2011; Figure 2.2.3). In these cases, lost fish can be translated into age equivalents or production foregone and stocked according to these numbers or biomass. Like-for-like and out-of-kind scenarios are possible depending on the species. Both timeframe and hatchery fish survival need to be considered. A one-time fish mortality event requires only a one-time stocking offset while periodically or regularly occurring losses, e.g., power plant water intake, need to be adjusted accordingly, e.g., on a yearly basis. Furthermore, hatchery fish could exhibit lower survival rates (Margenau 1992) which need to be included in the accounting. Some studies also suggest a higher impingement and entrainment rate for hatchery fish due to behavioral differences compared to wild individuals (Michaud 2000). These uncertainties and potential long term stocking requirements translate to common offset ratios around 1:3 and monitoring timeframes of 8 years and more. Ratios coincide with commonly accepted uncertainty and time-lag related considerations (Bradford et al. 2017). Overall, stocking could be suitable for direct mortality events that do not include a habitat component. For scenarios with harm stemming from indirect mortality, offsets based on habitat restoration and creation should be preferred (Figure 2.2.3). Re-introductions are only advised after removal of the harm source (e.g., post clean-up after a spill event) and restoration of the affected habitat (Dunham & Gallo 2008). Most stocking studies measure success in survival rates for both stocked fish as well as reference populations and population impacts, e.g., stocking was implemented to offset a reduction in juvenile survival or juvenile to adult survival. Survival rates for both should be translated either into surviving equivalent adults (or other equivalent age class) or in production foregone (biomass).

Temporal considerations

Benefits for Residual or Chronic Harm related offsets described in the previous section can be summarized into three main categories which relate to their temporal and target specific benefit nature (long-term, short-term, one time). These three temporal categories should be consequently related to either a habitat/ ecosystem, population or habitat/ ecosystem and population effect. For instance, habitat creation provides long-term benefits on both a habitat as well as population level and thus can be suitable to offset mortality events that either happen on a longer temporal scale or also relate back to a detrimental habitat effect besides the Residual or Chronic Harm (e.g., larval mortality through flow reduction and sediment accumulation during spawning season, e.g., Gammons et al. 2009; Scruton 2005). Restoration and enhancement measures can fall into all categories. For instance, restored connectivity will likely benefit a whole fish community long-term, while spawning gravel addition often targets a single salmonid species and deteriorates over time without maintenance (Barnthouse et al 2019; McManamay et al. 2010). Stocking and nutrient addition on the other hand, have short term benefits with stocking having a sole population effect and nutrient enrichment targeting biochemical ecosystem processes (Jarvie et al 2013; Sierp et al. 2008; Wipfli et al. 2003). Both require long-term management to meet consistent benefits. Overall, besides the stated generalized benefits like biomass increase, temporal and target related benefits need to be considered in the strategic planning process for Residual or Chronic Harm offsets (e.g., Duplisea et al. 2020; Song et al. 2019).

Monitoring, unintended impacts and uncertainty

While benefits are often evaluated and can be derived from literature, risks and unintentional effects are far less considered (e.g., Kemp 2016; Pastorino 2019; Schirmer et al. 2014; Figure 2.2.4). Stocking for instance, generally poses risks due to interaction of hatchery fish with wild stocks and subsequent effects like introduction of genetic material and food web as well as community shifts (Pastorino 2019). To counter these risks in the cases of stocking as an offset for Residual or Chronic Harm, clear objectives and a sound strategic approach are necessary (Figure 2.2.3). Stocking strategies in cases of Residual or Chronic Harm should include factors such as source of stocked fish (hatchery information), stocking timeframe and intervals, stocking density in relation to density dependent effects and carrying capacity as well as potential genetic, pathogen, community, and behavioral related impacts (Cowx 1994). Following a clear pathway as outlined in the example of Figure 2.2.3 will help determine if stocking could be an appropriate mortality offset and how to ensure tangible benefits while minimizing risks. Long-term monitoring will further reduce the potential bias of annual fluctuations and aid the decision-making process as well as help adjust stocking amounts (Bell et al. 2008; Louhi et al. 2016).

Overall, all offsets that can be utilized for Residual or Chronic Harm offsets hold the potential for unintended and/ or adverse effects on an ecosystem or aquatic community (Figure 2.2.4). Creation as well as restoration and enhancement measures can impact physical processes and structural properties of an aquatic ecosystem as well as biogeochemical characteristics (nutrient turnover) or biodiversity and community related aspects (Schirmer et al. 2014). Main concerns are the spread of invasive species through restored connectivity of waterways as well as shifts in community and food web structure through nutrient addition as well as enhancements for a specific target species. Other challenges for restored or created habitat include density dependent effects. Created habitat for salmonids for instance can lead to an increase in fish density but at a certain point affects fish condition and ultimately reduces the biomass per fish gain, especially in cases where the created habitat type was never utilized intensely by native species in the first place (Bond et al 2019). Planning strategies for Residual or Chronic Harm offsets, should incorporate an assessment of potential unintended and adverse effects (Figure 2.2.4). The self-sustaining nature of habitat offsets also needs to be considered in the planning process. Almost all major offsets require maintenance to adhere to the in-perpetuity requirement of their benefits. Maintenance and long-term adaptive management relate, as shown in the results, directly to adverse and unintended effects which then can be compensated and adjusted for as well as a potential reduction in offset benefits (Bell et al. 2008; Louhi et al. 2016). For instance, a river restoration project on the Thur River led to the gradual formation of a point bar over the course of 5 years which in turn led to large scale bank erosion and subsequent removal of riparian forest area (Schirmer et al. 2014). This example shows how long-term maintenance and project adjustment is often necessary to balance benefits and unintended impacts.

2.2.5 Bias, validity, and study limitations

The funnel plots (Figure S2.2.1-S2.2.3) indicate bias for habitat creation studies (Figure S2.2.1) with effect sizes biased towards successful studies with large effect sizes. Positive results are commonly over-reported in publications and could potentially restrict applicability of found evidence especially in reviews that are restricted to narrow fields or lack available information like the case of Residual or Chronic Harm (Nissen et al. 2016). Though trying to address this bias as good as possible through inclusion of grey literature data and reports and testing for bias, utilizing model selection and weighted effect sizes, our results underline the need for reviews to test and account more for publication bias as well as to actively seek out sources that will cover failure and negative treatment effects (Huntington 2011). Furthermore, data availability was generally low and required extensive efforts to obtain or extract data through additional software or direct contact with responsible parties. This fact shows how difficult it is to form sound scientific advice on new or recently emerging topics and might delay policy development or implementation process. It further supports the need for better collaboration between stakeholders, agencies and on the ground personnel since benefiting all parties in the long run. Overall, due to low data availability and potential study bias, results from this review should be seen as potential application scenarios for different offsetting methods in cases of Residual or Chronic Harm, like the special case of stocking, rather than definite recommendations or absolute benefit estimates. Residual or Chronic Harm is a proven concept with little to no advice on best practice. It remains vital to increase knowledge on this topic and develop management and policy frameworks that can aid in providing guidance and support on Residual or Chronic Harm offsets.

2.2.6 Conclusions

The presented review of the literature and meta-analysis on offsets for Residual or Chronic Harm in aquatic ecosystems demonstrated that habitat creation, habitat restoration and enhancement, and biological and chemical manipulation can all be feasible options for offsetting Residual or Chronic Harm given caveats and general monitoring timeframes.

Habitat creation to offset Residual or Chronic Harm is mostly studied for Salmonid species and requires further study and application to other species and communities. Offset costs and size can increase rapidly in habitat creation projects, with a potential size threshold beyond which benefits become difficult to achieve. Habitat creation provides the most benefits for larval and juvenile live stages. Based on the assessed literature, applied offsetting ratios were around 1:5.

Restoration and enhancement are the most used offsets in cases of Residual or Chronic Harm. Reconnection can be an easily implemented measure to provide benefits on a large scale. Restoration measures often target whole communities and need to be carefully considered when targeting a specific species. Enhancement measures, such as spawning substrate introduction, may be more likely to ensure species-specific benefits. Habitat enhancement and restoration provides the most benefits for larval and juvenile life stages. Based on the assessed literature, applied offsetting ratios were around 1:2.5.

Stocking can be an effective replacement for lost or harmed fish, given a stable and unimpaired ecosystem and no significant bottlenecks. Hatchery fish tend to have slightly lower survival rates than wild fish and are more vulnerable to harm and mortality sources, e.g., impingement. Offset ratios (commonly between 1:1.5 and 1:3) can be applied to compensate for this uncertainty about survival. Stocks need to be monitored frequently to ensure benefits. Stocking needs to be conducted in frequent intervals when fish mortality stems from a regular occurring harm source. Based on the reviewed literature, applied offsetting ratios were around 1:3.

All three offsetting types can be potentially detrimental when an out-of-kind replacement or a species versus community effect takes place on a magnitude that disrupts or alters community structure and food web composition. Pre-impact assessments tend to increase offsetting success significantly and should be conducted for cases involving Residual or Chronic Harm if possible. Time to achieve first benefits in most offsets required one or more years. This time lag needs to be accounted for in both implementation and monitoring. Cases using habitat productivity metrics to quantify creation or restoration offsets should use unimpaired reference systems. References should reflect the regional average and the appropriate target species or community. Single reference sites, systems, or unsuitable literature reference values can easily distort the value of offsets. Monitoring timeframes with a minimum of four years tend to be associated with higher chances of success in projects offsetting Residual or Chronic Harm.

Chapter 2: Figures and Tables

Chapter 2.1 Figures

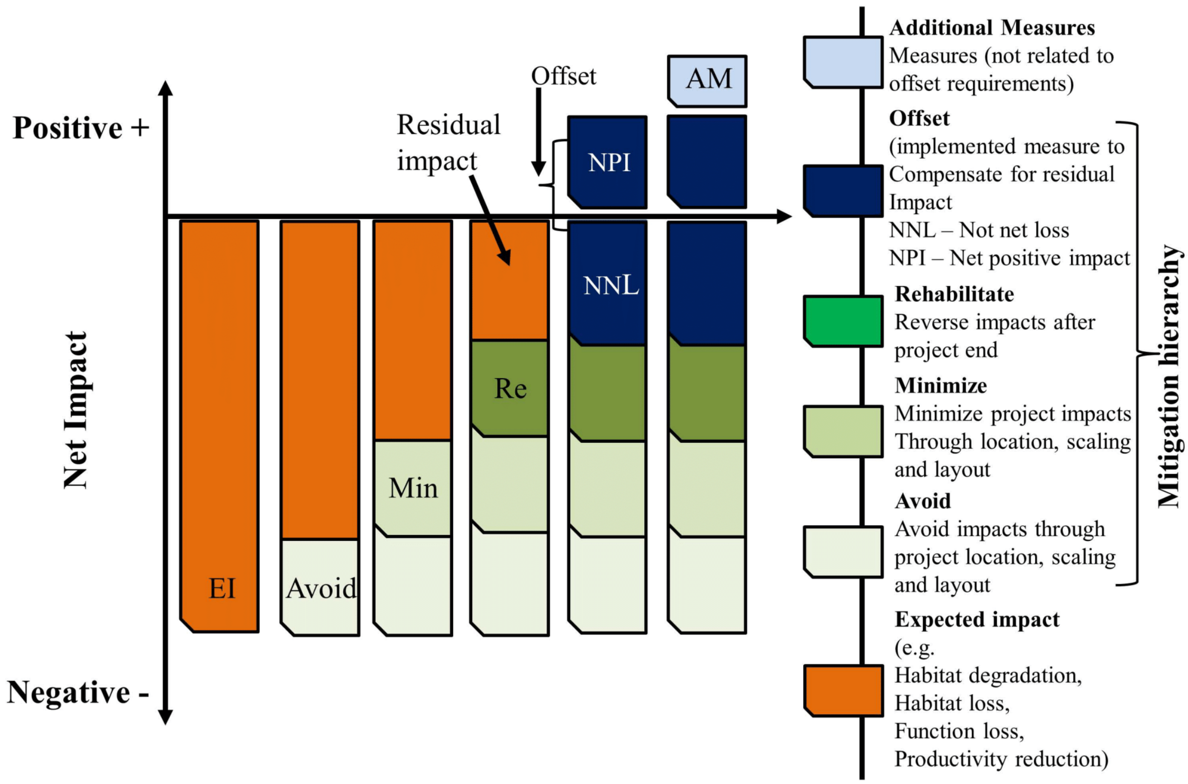


Figure 2.1.1. *Offsetting diagram, showing the offsetting principle and its place in the mitigation hierarchy in reducing residual impacts and achieving No Net Loss or Net Positive Impact (adapted from Kiesecker et al. 2011).*

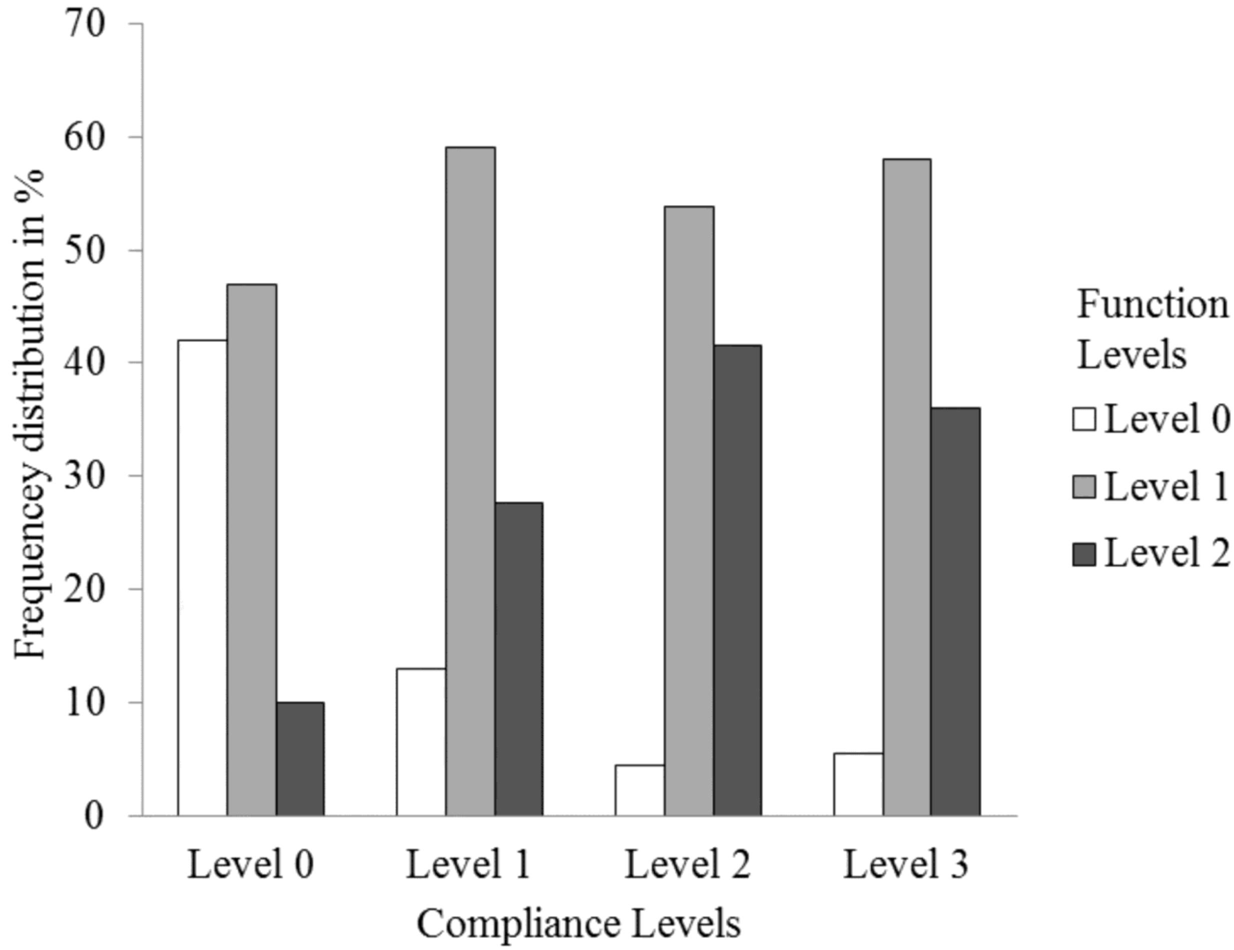


Figure 2.1.2. *Frequency distribution of function (0 = non-functional, 1 = partially functional, 2 = fully functional) and compliance (0 = non-compliant, 1 = partially compliant, 2 = fully compliant, 3 = over-compliant) levels.*

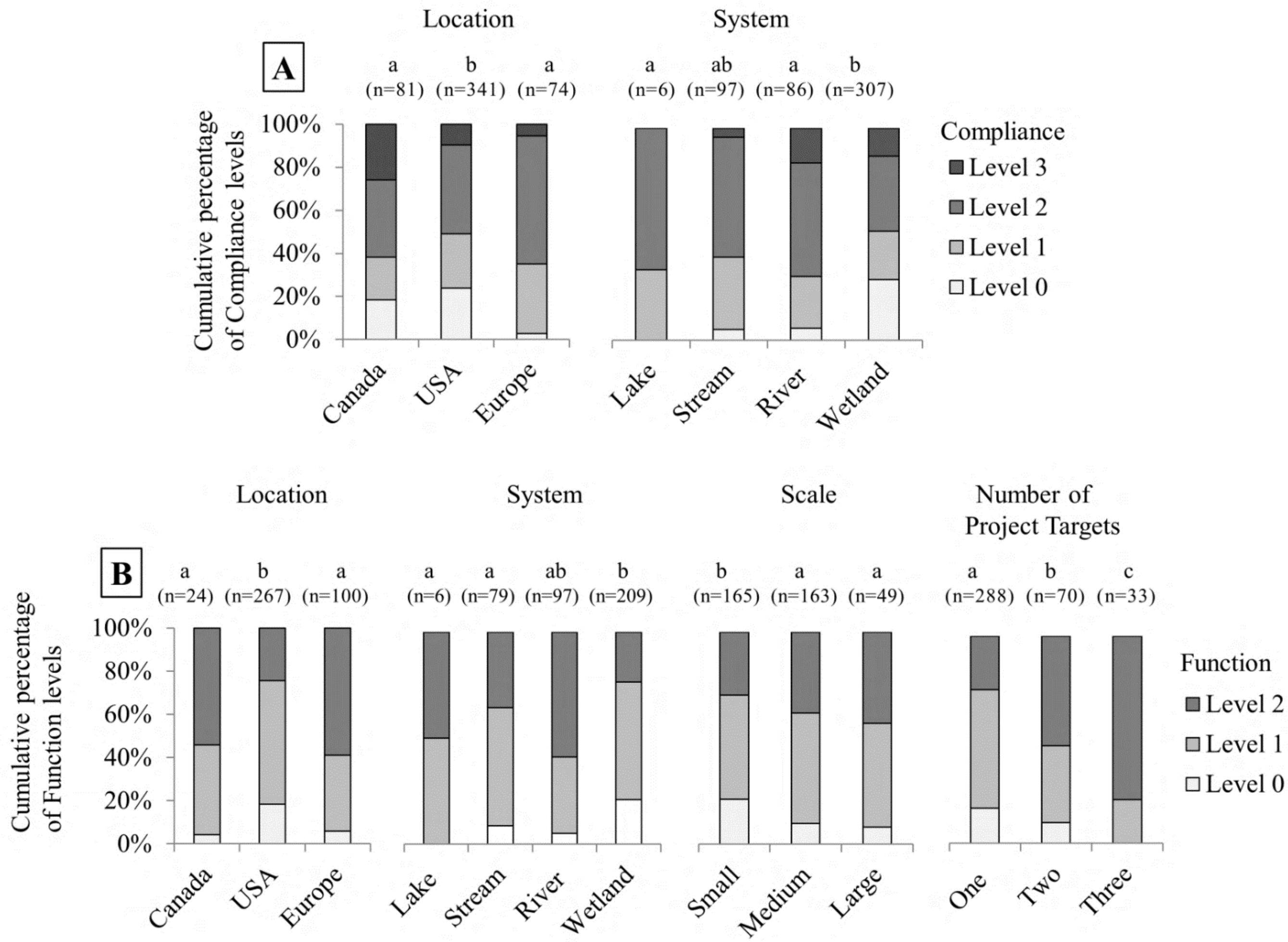


Figure 2.1.3. *Cumulative percentage bar graphs showing influential factors on Compliance (A) and Function (B) levels. Factors include project system (river, lake, stream, and wetland), region (Canada, US, and Europe), project scale (small, medium, large) and number of project targets (1, 2, 3, Productivity, Habitat, Function). Significant differences in mean compliance and function levels (Scheffé-Test) are identified by different letters in each group (a, b, c).*

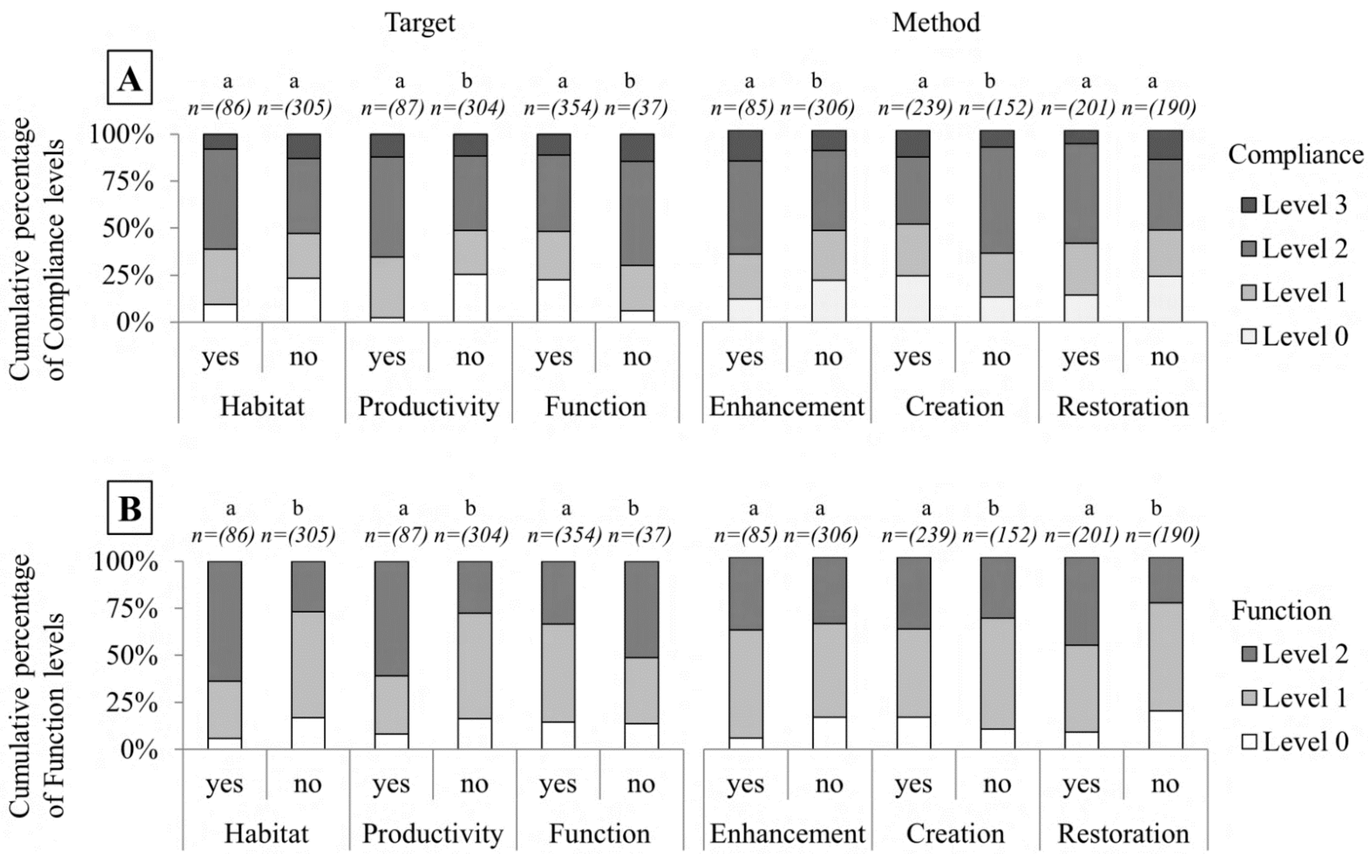


Figure 2.1.4. *Cumulative percentage bar graphs showing influential project target or used method on Compliance (A) and Function (B) levels. Targets include Productivity, Habitat and Function. Methods include Creation, Restoration and Enhancement. Significant differences in mean compliance and function levels (Scheffé-Test) are identified by different letters in each group (a, b).*

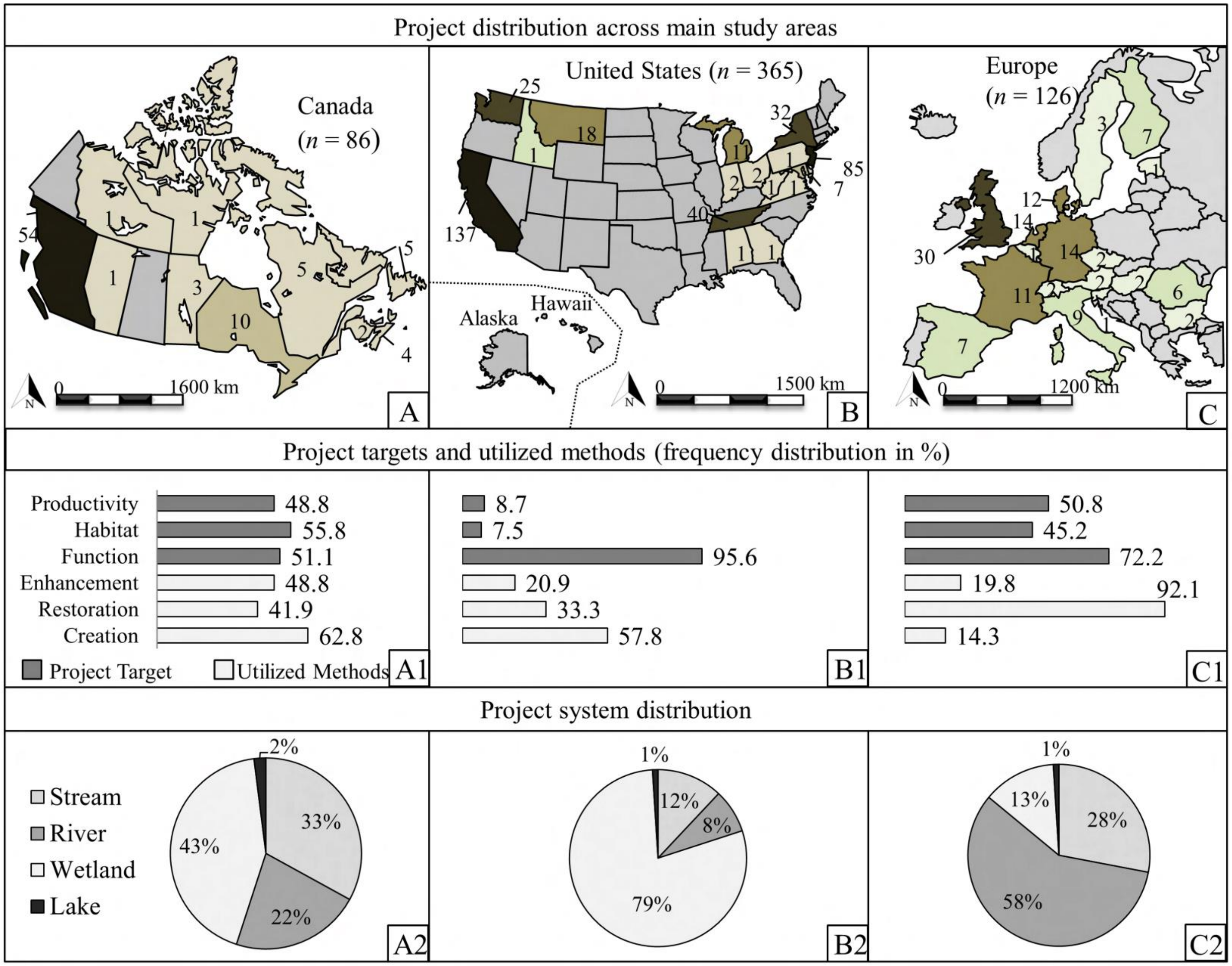


Figure 2.1.5. *Regional project density distribution for Canada (A), the United States (B) and Europe (C). Offsetting measures and project target for Canada (A1), United States (B1) and Europe (C1). Individual projects can contain several targets and measures (proportional bar graphs). Aquatic ecosystems in which offsetting projects were implemented: Canada (A2), US (B2) and Europe (C2).*

Chapter 2.1 Tables

Table 2.1.1. *Mean compliance (C) and function (F) scores and their standard deviations (± SD) for the three project targets: Productivity, habitat and basic-function on a global level (n = 577), arranged by project scale (small, medium, large). Multiple targets can be present in a single project.*

	Productivity		Habitat		Basic-Function	
Scale	C	F	C	F	C	F
Small	1.77 ± 0.59 (n = 36)	1.33 ± 0.73 (n = 33)	1.78 ± 0.75 (n = 41)	1.53 ± 0.62 (n = 30)	1.45 ± 0.99 (n = 209)	1.07 ± 0.68 (n = 166)
Medium	1.78 ± 0.72 (n = 47)	1.65 ± 0.53 (n = 35)	1.49 ± 0.79 (n = 55)	1.55 ± 0.61 (n = 47)	1.40 ± 0.86 (n = 208)	1.25 ± 0.59 (n = 137)
Large	2.18 ± 0.75 (n = 16)	2.00 ± 0 (n = 13)	1.66 ± 0.88 (n = 12)	1.81 ± 0.40 (n = 11)	1.55 ± 0.90 (n = 40)	1.38 ± 0.63 (n = 50)

Table 2.1.2. *ANOVA model statistics, their degrees of freedom, and levels of significance for function and compliance regarding location, system, scale, offsetting methods and project target. Non-significant factors and interactions were removed from the model (Initial model: Function (Compliance ~ Location*System*Scale*Target#*Method#). Non-significant factors as part of a significant interaction were kept in the model. Linear permutated model design used for non-linear distribution of data.*

	Df	SumSq	MeanSq	F-value	Pr(>F)
Function					
Location	3	22.27	7.423	20.752	<0.001
System	3	4.38	1.461	4.084	0.007
Scale	2	2.95	1.477	4.130	0.0168
Target #	2	6.05	3.027	8.462	<0.001
Residuals	405	144.87	0.358		
Compliance					
Location	3	8.4	2.8144	3.498	0.0154
System	3	8.4	2.8150	3.499	0.0154
Residuals	546	439.3	0.8046		

Chapter 2.2 Figures

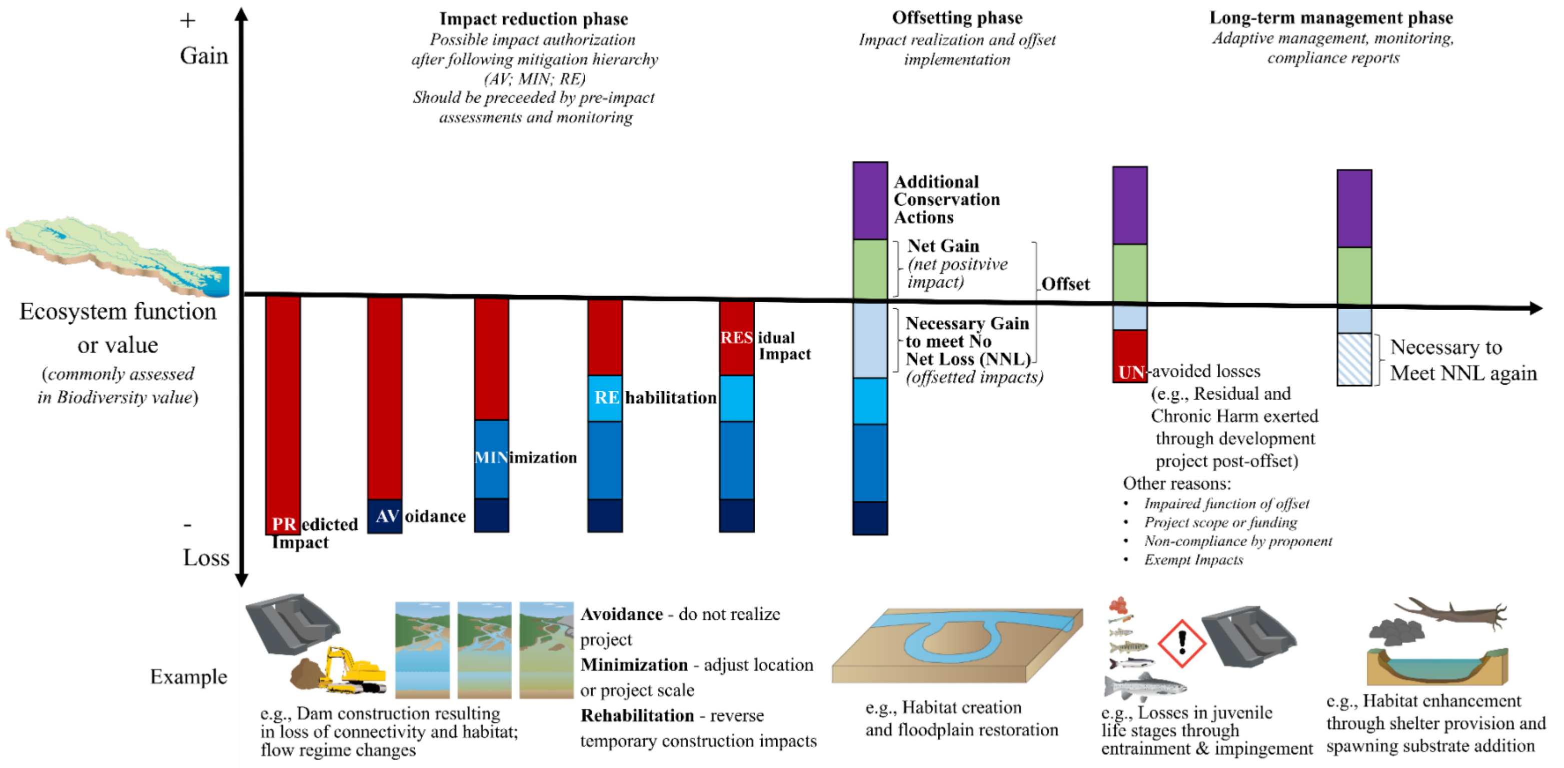


Figure 2.2.1. *Incorporation of Residual or Chronic Harm into the mitigation hierarchy (A). RH encompasses negative impacts linked to the initial authorized negative (anthropogenic) impact becoming apparent post offset implementation (B). Main Residual or Chronic Harm sources and types listed as found in the literature (C). (Based on Arlidge et al. 2018; Horne et al. 2004; Keeley 1996; Lemly 2010; Song et al. 2019; Williams et al. 2005). (Image attribution to: Claire Sbardella; Jane Hawkey; Kim Kraeer; Lucy Van Essen-Fishman; Sally Bell; Tracey Saxby; Integration and Application Network (ian.umces.edu/media-library).*

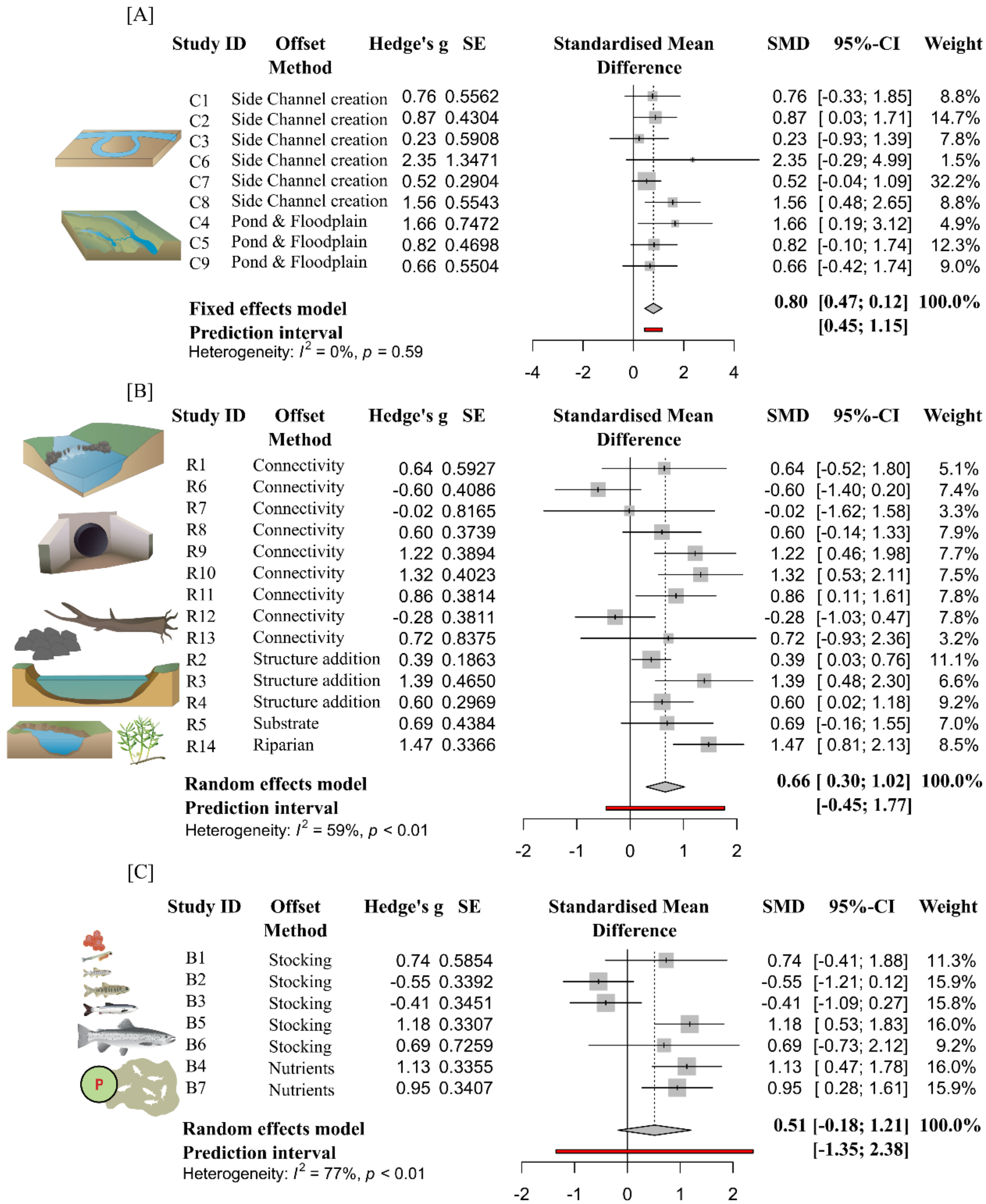


Figure 2.2.2. *Meta-analysis forest plots showing standardized mean difference (SMD) and SE for the assessed studies divided into the three main categories (habitat creation A; habitat restoration B; biological and chemical manipulation C). 95% Confidence intervals and weight based listed for each SMD. Results derived from fixed effects model (habitat creation; homogenous variance of effects); Jackson method; DerSimonian-Laird estimator for tau*^{2}*) and random effects models (habitat restoration; biological manipulation; heterogeneous variance of effects; Hartung-Knapp adjustment; restricted maximum likelihood estimator for tau*^{2}*). Full model output and tau calculations in Table A1 – 3 and Meta.xlsx. (Image attribution to: Claire Sbardella; Jane Hawkey; Kim Kraeer; Lucy Van Essen-Fishman; Sally Bell; Tracey Saxby; Integration and Application Network (ian.umces.edu/media-library).*

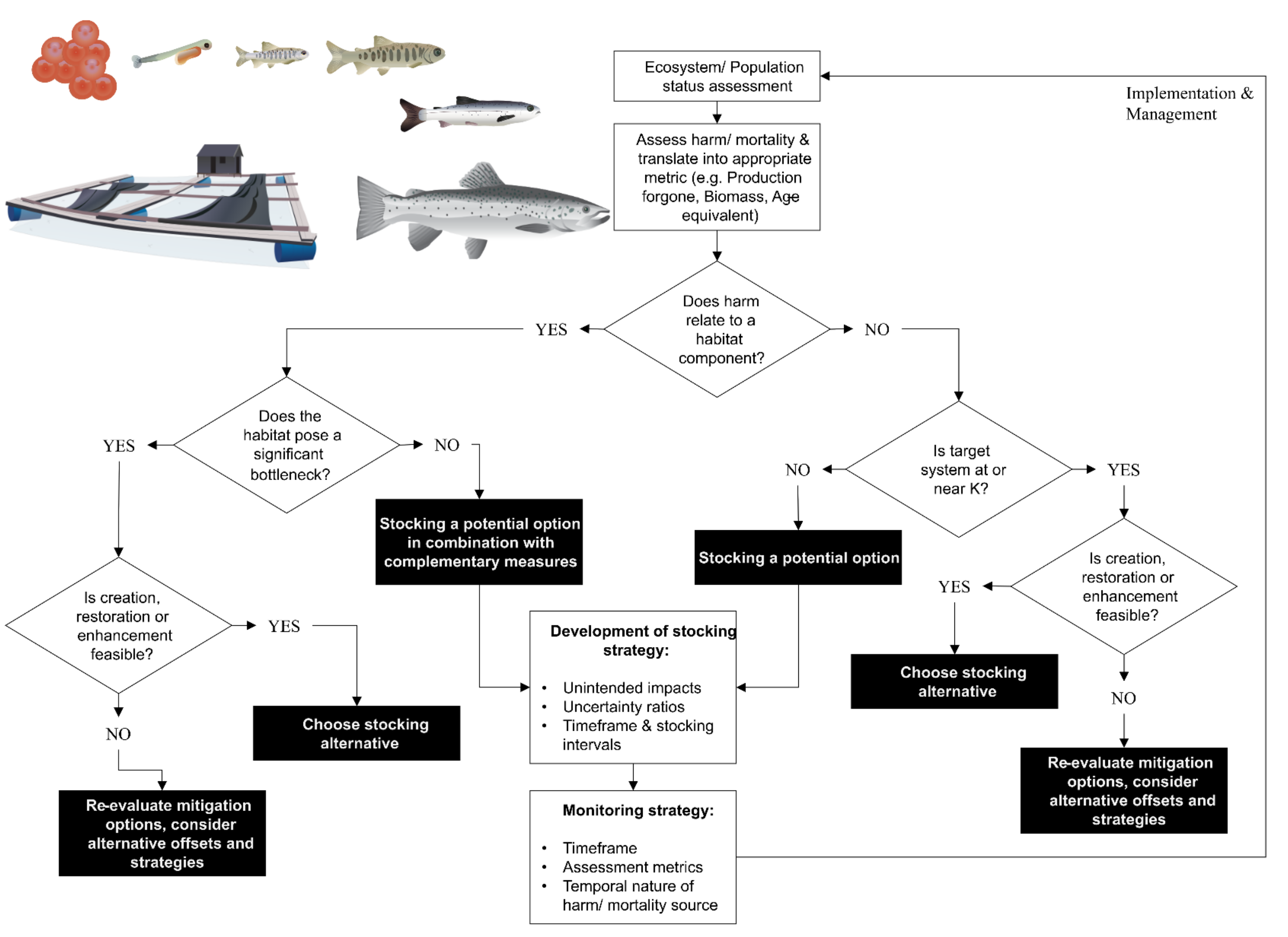


Figure 2.2.3. *Flowchart of potential stocking application in the context of Residual or Chronic Harm following the implementation of an offset linked to an authorized negative impact on aquatic resources and considerations on whether stocking could be viable and appropriate. (Image attribution to: Claire Sbardella; Jane Hawkey; Integration and Application Network (ian.umces.edu/media-library).*

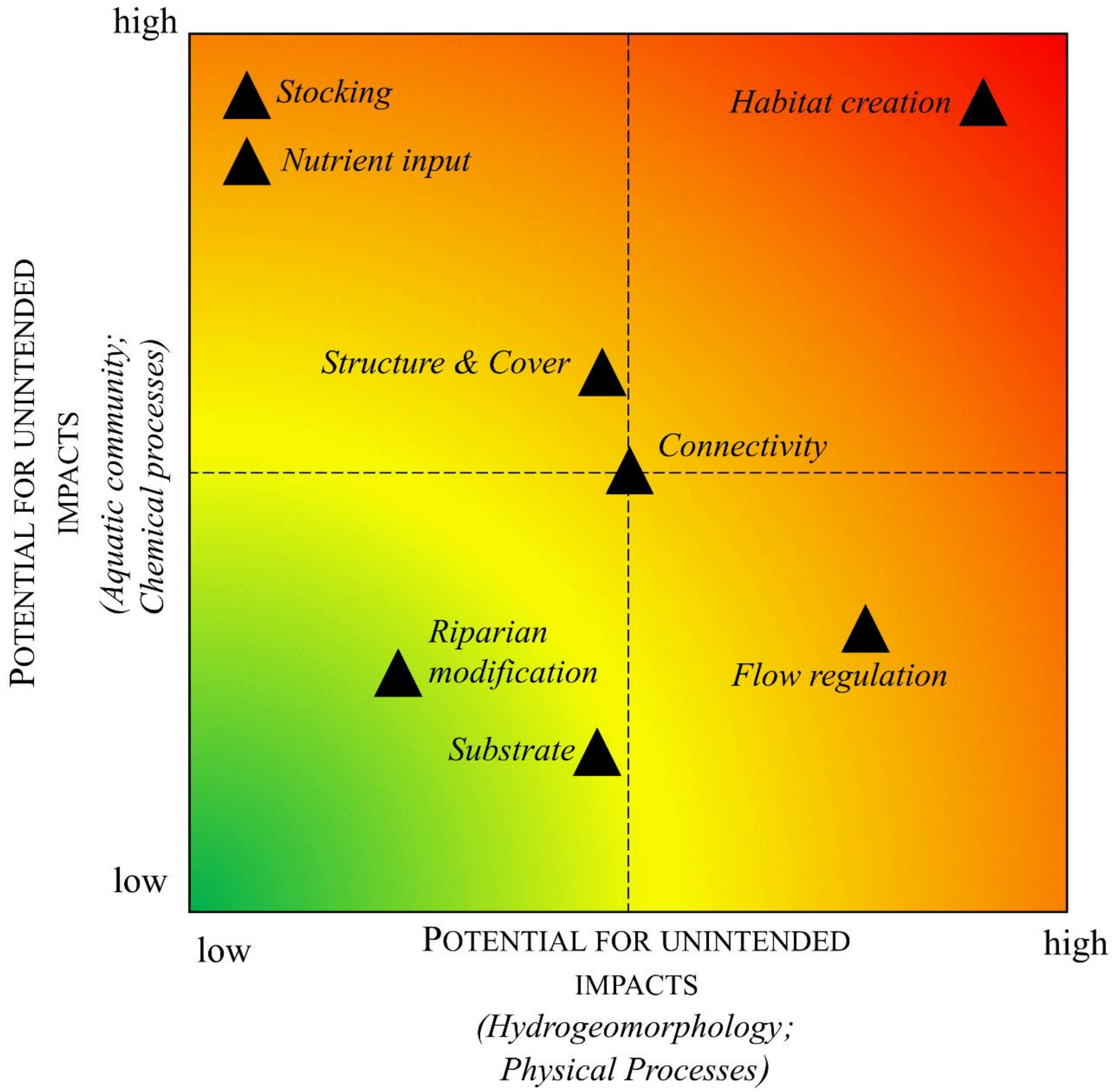


Figure 2.2.4. *Potential for unintended impacts exerted through the main types of assessed offsets, considered for residual chronic harm offsetting, as identified in the literature regarding community aspects (e.g., food web; nutrient cycling; completion) or physical ecosystem and habitat aspects (e.g., flow rate; erosion; temperature; based on: Cowx 1994; Kemp 2016; McLaughlin et al. 2012; Pastorino 2019; Schirmer et al. 2014).*

Chapter 2.2 Tables

Table 2.2.1. *Summary of habitat creation project (n = 9), habitat restoration projects (n = 14), and biological manipulation (n = 7) metrics and benefits for offsets associated with residual or chronic harm (RH) in terms of effect size (pooled and weighted effect sizes; CI) as well as biomass increase (*ΔB*). Average monitoring times in years (SD), average time for first benefits (years; SD), mean recorded costs per area (USD/m, m*^{2}*), target species, and commonly applied offset ratios between impact and offset.*

Offset Method	Monitoring average (years)	Time for first benefit (years)	Cost area /m/m^2	Preferred species*	Commonly applied offset ratio	Pooled Effect size (hedge's g, 95% CI)	ΔB
Habitat creation							
Side-channel	5.7 ± 1.6	1.1 ± 0.9	150 ± 46	Salmonid	1:5.7	0.75 [0.26; 1.25]	1.88 ± 1.01
Off channel pond or floodplain	4.9 ± 1.5	1.2 ± 0.6	85 ± 27	Salmonid	1:4	0.92 [-0.17; 2.01]	1.47 ± 0.78
Habitat restoration							
Riparian Restoration	1.8 ± 1.1	0.8 ± 0.4	68(m) ± 26	Community	1:1.2	1.47 [0.81; 2.13]	0.21 ± 0
Structure addition	3 ± 0.7	1 ± 0.7	188(m) ± 123	Salmonid, Community	1:1.6	0.65 [-0.47; 1.77]	1.62 ± 0.44
Connectivity	4.1 ± 1.8	1 ± 0.6	84(m^2) ± 77**	Diadromous, Potamodromous, Rheophilic	1:4.6	0.51 [-0.02; 1.04]	1.24 ± 0.63
Substrate	2.3 ± 1.1	1.1 ± 0.9	11(m^2) ± 7	Salmonid, substrate spawner/ Lithophilic	1:2.1	0.69 [-0.16; 1.55]	1.12 ± 0
Biological manipulation							
Stocking	8.3 ± 9	1.2 ± 0.7	Species dependent	Salmonid, Community	1:3.1	0.28 [-0.72; 1.27]	0.84 ± 0.77
Nutrients	4 ± 0.7	0.3 ± 0.47	-	Different trophic levels	-	1.04 [-0.11; 2.18]	2.01 ± 0.31

**full species list in supplements*
***highly variable and depends on the size of connected or reconnected habitat*

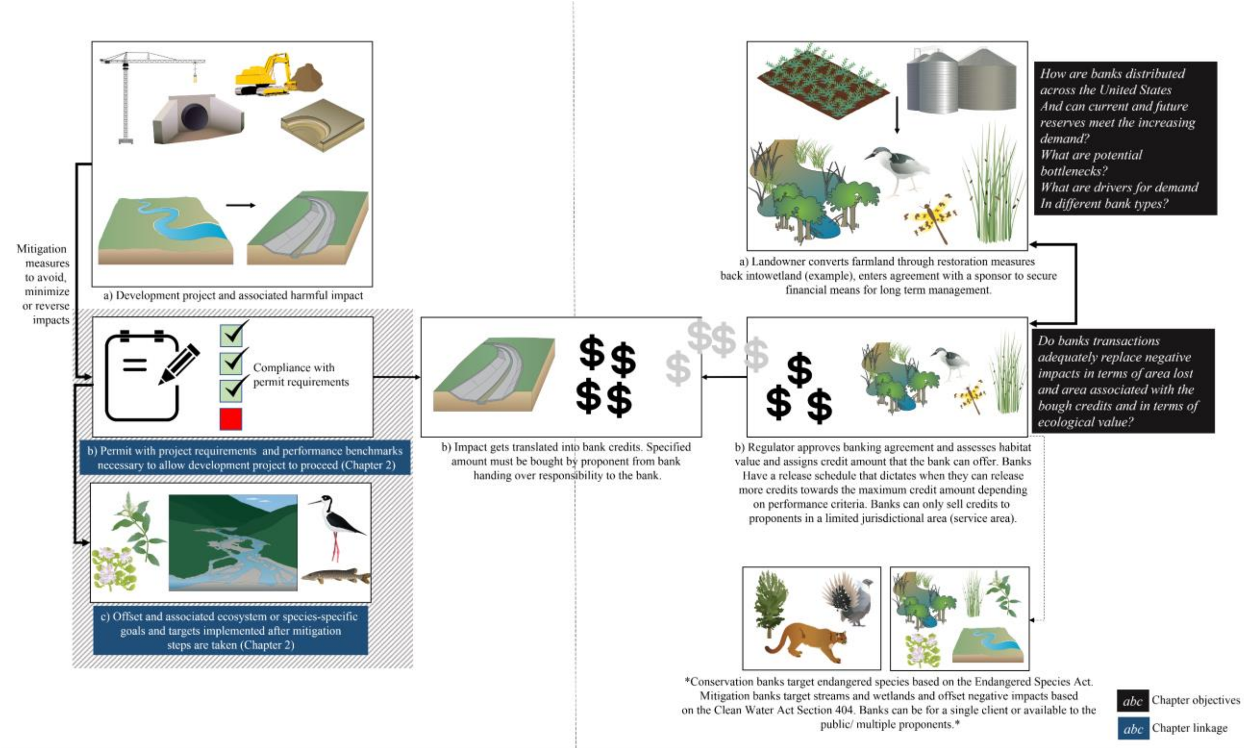
Table 2.2.2. *Effect of monitoring timeframes (years), location (onsite, offsite), and collection of pre-assessment data (yes; no) on general offset success based on the general success score (GSS; SD; no; partial; full). Different letters [abc] indicated significant differences (Kruskal-Wallis test & pairwise Wilcox test, Bonferroni correction for p-values).*

Monitoring time (n = 30)	<4 years	4 to 6 years	>6 years
Success Score	0.86 ± 0.89^{a}	1.53 ± 0.7^{cb}	1.30 ± 0.84^{ab}
Pre-assessment (n = 27)	Yes	No	
Success Score	1.70 ± 0.57^{a}	0.75 ± 0.89^{b}	
Onsite/ Offsite (n = 29)	Onsite	Offsite	Both*
Success Score	1.29 ± 0.86	1.41 ± 0.89	1.5 ± 0.71

**low sample size n < 3.*

Chapter 3: Habitat banking practices in the United States

Graphical synopsis of key background concepts, terms, and Chapter goals



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Chapter 3.1: Assessing conservation and mitigation banking practices and associated gains and losses in the United States

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Summary

Conservation and mitigation banks are widely used alternative mechanisms to traditional offsetting to compensate for unavoidable negative environmental impacts from development. Conservation and mitigation banks allow proponents to buy credits to offset negative residual impacts to achieve No Net Loss in ecosystem function and habitat area. While considered a valid approach in North America, with a long-standing practice of over 30 years, banking outcomes in terms of No Net Loss remain unclear. This study determines if mitigation and conservation banks in the United States have achieved No Net Loss between 1995 and 2020. A total of 12756 transactions for 1736 banks were tested for meeting No Net Loss requirements for credits to area yield-based mitigation ratios with a minimum of 1:1 ratio of impact to offset. We further tested if impact types were matched appropriately with offset types by transactions, to replace ecosystem function and habitat area to meet ecological equivalence. We conclude that most bank transactions using Preservation, Enhancement, and Re-establishment targeting wetlands, species, or multiple Mitigation-Targets meet No Net Loss requirements on a ratio base with overall good matches between impact and offset types. However, wetland transactions, making up most of all

assessed transactions (n = 10628), still miss matching appropriate impact to offset types in 25% of all cases, mainly due to Preservation not leading to any additional habitat area gain. This mismatch likely leads to a large proportion of wetland transactions not meeting ecological equivalence. Stream transactions mainly using Rehabilitation often miss No Net Loss targets and fitting impact to offset types, with the use of Rehabilitation as Mitigation-Method increasing significantly over the past 25 years. While the Preservation of wetlands and the Rehabilitation of streams can provide a multitude of benefits, both practices need to be revised on an ecological level to bridge the gap between Not Net Loss based on credit and area yield ratios and ecological equivalence.

3.1.1 Introduction

Land-use development has steadily increased in recent decades, leading to significant losses in biodiversity and large-scale habitat degradation and fragmentation on a global scale (Dirzo & Raven 2003). Driven by stakeholders like policymakers, financial institutions, non-governmental organizations (NGOs), and industry, conservation efforts have shifted from simply protecting certain species or areas to developing frameworks and approaches aimed to preserve a diverse array of functions, ecosystems services as well as species diversity adaptively and sustainably (Bull et al. 2013; McKenney & Kiesecker 2009). Offsetting is one such approach that has been widely adopted into legislation across the globe (e.g., Natura 2000 in Europe, U.S. Army Corps of Engineers Regulatory Guidance Letters; Global Inventory of Biodiversity Offset Policies; McKenney 2005). Offsetting is based on the principle that negative impacts on a species or ecosystem will be compensated for at the impact site (on-site) or somewhere else (offsite). The underlying principle of offsetting is that there is No Net Loss (NNL) of habitat area (minimum of 1:1 ratio of gains to losses), function, condition (quality), biodiversity, services, or other defined equivalency targets (Figure 3.1.1A; Bull et al. 2013; Gardner et al. 2011; Grimm & Köppel 2019).

The use of banking has been established as an increasingly popular and constantly evolving mechanism for delivering a required offset. Unlike traditional offsetting mechanisms, which are proponent-led (impact proponent is responsible for offset approval and implementation), banking allows the proponent to purchase credits using an accreditation system to secure gains elsewhere (Figure 3.1.1A & B; Boisvert 2015). Banking features many similarities to the proponent-led compensation mechanism, as the offset provided by the bank is the last step in the mitigation hierarchy. (Figure 3.1.1A). Banks are composed of the banking agreement between owner and regulatory agency; the physical banking areas (managed through preservation; enhancement; etc.); service area (geographic area in which a bank can sell credits to a proponent), and potential bank sponsors. Banks can be created and managed by government agencies, industry, NGOs, or private entities (USEPA 1995). In North America, banking types are divided into (Wetland and Stream) Mitigation Banks, Conservation Banks, In-Lieu Fee programs (ILFs), and Umbrella Banks (USEPA 1995; USFWS 2003 & 2019; White 2012; Table 3.1.1).

The banking mechanism features some unique differences compared to traditional proponent-led offsets. Firstly, moving responsibility away from the proponent reduces time and potentially the monetary investments required from the proponent (USEPA 1995; USACE 2008; White 2012). Secondly, perpetuity requirements for the banks are often met more successfully due to different sponsorships and changing management entities compared to traditional offsets (Carreras Gamarra & Toombs 2017; Moreno-Mateos et al. 2015). Similarly, banking in the North American context of proponent-led versus banking-led offsets can reduce risk since legal liability and responsibility for ecological offset failures are transferred to the bank (Figure 3.1.1B; Burgin 2008; Moreno-Mateos et al. 2015). While offsetting creates a market-driven environment, in the case of banking it allows landowners to use their land for conservation purposes rather than exploiting its resources. However, this incentivized form of ecosystem stewardship, requires loss of ecosystem aspects, services, and/ or biodiversity elsewhere. Banking is well suited for preserving high-quality habitat as well as securing larger areas of connected habitat, ensuring better connectivity, as opposed to a patchwork of offsets (White 2012). These benefits are also in the interest of the public as it allows time-sensitive projects to be actualized sooner (White 2012).

Banking faces many of the same technical issues as traditional offsetting mechanisms (Figure 3.1.1C). Issues of achieving ecological equivalency remain, despite being economically incentivized to achieve by decoupling ecosystem values from their complex context (Zu Ermgassen et al. 2017; Maron et al. 2016; Moreno-Mateos et al. 2015). Further issues include: poor transparency in reporting and collection of data (Carreras Gamarra & Toombs 2017; Quétier & Lavorel 2011); as well as, the lack of resources to enforce long-term monitoring by regulatory agencies. Some banks struggle with the inability to secure the necessary endowment funds for perpetual management which can lead to management, and sponsorship changes or bank closure (Gardner & Pulley Radwan 2005; Matthews & Endress 2008; Figure 3.1.1C). There has also been a disjunction between regulatory requirements and ecosystem function, meaning that compliance with regulation does not necessarily lead to good ecosystem function (Zu Ermgassen et al. 2017; Gutrich & Hitzhusen 2004; Maron et al. 2016; Matthews & Endress 2008; Figure 3.1.1C). Furthermore, un-avoided long-term losses at the impact site, offset degradation over time, as well as loss of irreplaceable ecosystem aspects are frequently insufficiently accounted for in the initial project and offset planning process due to high degrees of uncertainty (Bonds & Pompe 2003; Gutrich & Hitzhusen 2004; Figure 3.1.1A; C). The main, persisting issue for banks is the overuse of preserving habitat areas of low ecological value, and/or species that are not in immediate danger. This can have a similar effect when not meeting true ecosystem equivalency or not considering multi-dimensional, ecosystem services or biodiversity values (e.g., Burgin 2009; Carreras Gamarra & Toombs 2017; Zu Ermgassen et al. 2017; Maron et al. 2016; Moreno-Mateos et al. 2015).

Unified legislation and policy guidelines were released in 1995 in the United States as part of the Clean Water Act (CWA 404) on the role and the establishment of banks. Many states and regions laid the banking groundwork and practices independently, which resulted in distinct differences between established banking networks, across regions in the United States (Fox & Nino-Murcia 2005; Mead 2007). To unify banking practices, the United States Army Corps of Engineers (USACE) and Environmental Protection Agency (USEPA) as well as the Fish and Wildlife Service (USFWS) have evaluated projects and permits issued between 1995 and 2008, investing more resources into educating stakeholders and enhancing databases to ensure greater banking transparency in the future (USACE 2008; 2015).

To investigate if banks achieve ecological equivalency as well as NNL in terms of habitat area on a national scale, we used publicly available data on banks in the United States through the Regulatory In-lieu Fee and Bank Information Tracking System (RIBITS; Zu Ermgassen et al. 2017; USFWS 2003; Grimm 2021; Sonter et al. 2019).

Our main research questions are:

1) Are transactions conducted by habitat banks in the United States likely to achieve NNL for *Ecological Equivalency* and *Ratio Equivalency*, based on habitat area, and are there discernible differences in achieving NNL between different *Mitigation-Targets* and *Mitigation-Methods*?

2) What are the possible shortcomings and reasons for missing NNL and ecological equivalency targets?

3) Is *Preservation* overused as a *Mitigation-Method* and what targets and benefits does it provide?

We addressed question 1) by extracting bank transactions from RIBITS and assessing the impact, offset, and credits to determine *Gain:Loss ratios*, with the assumption that a ratio of 1:1 is accepted as NNL. Question 2) was answered by looking at how gains and losses are recorded in the RIBITS database and how well *Impact-Types* and *Offset-Types* match, hinting at *Ecological Equivalency* between gains and losses. Question 3) was investigated through an assessment of banks relying on *Preservation* as their main *Mitigation-Method*.

3.1.2 Methods & Analyses

Data were extracted from the Regulatory In-Lieu Fee and Banking Information Tracking System (RIBITS; last accessed December 31st, 2021). RIBITS contains up-to-date banking reports for the entirety of the United States. We extracted data from 4039 banks and In-Lieu Fee program (ILF) sites listed for the United States. Banks were defined as Conservation, Mitigation, and Umbrella banks since they all state that their end goal is to satisfy compensatory mitigation through *Preservation*, *Establishment*, *Re-establishment*, *Rehabilitation*, and *Enhancement* (McKenney 2005; Table 3.1.1). To investigate current banking practices, only approved banks were included in our analysis, as well as only banks having the required information listed (Table 3.1.1). No ILF sites were retained in our analysis. In-Lieu Fee programs and sites were excluded since they often do not provide the same environmental benefit as bank credits (Gardner 2011). This is because funds from a proponent are deposited into In-lieu fee funds managed by an NGO or government agency mostly for future environmental management. These funds are generally spent for future projects thus potentially allowing habitat loss or degradation without a timely offset (Gardner 2011). We also only considered banks established after the USACE guidelines of 1995 were put into place to have banks with a uniform reporting system. This led to the inclusion of 1736 banks for analysis (Table 3.1.1). A second dataset was used containing all the ledger transactions associated with banks providing information about impacts and offsets as well as Mitigation-Methods and Mitigation-Targets (n = 12756; Table 3.1.1).

Key variables

Bank types across regions

We divided the United States into 5 distinct regions: *Northeast*, *Southeast*, *Midwest*, *Southwest*, and *West*. The regional division was based on the geography of the United States as well as banking history to capture differences between broad ecoregions and development approaches (Fox & Nino-Murcia 2005). Bank Types were divided into six bank types consisting of *Wetland*, *Stream*, and *Species* banks and three combinations of these types. *Wetland* encompasses any bank offering credits related to Palustrine, Estuarine, and Lacustrine systems (Table 3.1.1; Cowardin et al. 1979). *Stream* banks encompass riverine systems with habitats contained within a channel generally bounded by upland areas and *Species* banks encompass any bank agreement designated towards a target species (mainly conservation banks; Table 3.1.1). A banking agreement targeting multiple species was recorded as *Multi-Species* and a bank agreement encompassing both stream and wetland systems as *Multi-Ecosystem*. Bank agreements focusing on both Species and Ecosystems were named *Group* banks (Table 3.1.1).

Compensatory Mitigation-Methods for banking

Mitigation-Method describes the method used to meet the compensatory mitigation goal. We identified five different methods: *Enhancement, Establishment, Re-establishment, Rehabilitation*, and *Preservation* (Table 3.1.1). *Enhancement* covers the manipulation of the physical, chemical, or biological characteristics of a habitat area to improve a specific ecosystem function (USACE 2008; Table 3.1.1). *Establishment* in the context of RIBITS and this study means creation of a habitat area, previously non-existent. *Re-establishment* has the same definition except that it is meant to rebuild a former habitat area. Both methods result in a gain in habitat area and ecosystem function (USACE 2008; Table 3.1.1). *Rehabilitation* aims at repairing the natural or historic function of a degraded ecosystem, resulting in ecosystem function gain. *Preservation* is defined as threat removal or prevention of a decline in ecosystem function or habitat area, while also covering maintenance and management mechanisms. (USACE 2008; Table 3.1.1).

Gain:Loss ratios – Ratio Equivalency

Transactions (n = 12756) linked to recorded impacts and credits were converted into *Gain:Loss* ratios. *Gain:Loss* ratios were calculated by credit yield for a given *Mitigation-Method* (gain) per acre/ linear feet compared to impact (loss). Example: Preservation of 10 acres yields 0.5 units credit per acre = gain of 5 acres and negative impact of 10 acres results in a *Gain:Loss* ratio of 0.5:1. Ratios, representing *Ratio Equivalency* between gain and loss, were assigned to four categories responding to meeting NNL (assuming a minimum of 1:1 *Gain:Loss* ratio under equivalent *Impact-Type* to *Offset-Type* assumption), acknowledging that many regulatory agencies require higher ratios (Categories: *Loss* = *Gain:Loss* < 0.25:1; *Partial* = *Gain:Loss* 0.25-0.9:1; *NNL* = *Gain:Loss* > 0.9-1.25:1; *Gain* = *Gain:Loss* > 1.25:1; Table 1). Ratio ranges account for inherent variability and inaccuracy of measuring offsets (e.g., Zu Ermgassen et al. 2017; Grimm 2021; Sonter et al. 2019). This way, each transaction is associated with one of the four NNL categories (*Loss; Partial; NNL; Gain*) as well as *Mitigation-Target* (*Wetland; Stream; Species; Group*) and *Mitigation-Method* (*Preservation; Enhancement; Rehabilitation; Re-establishment; Establishment*; Table 3.1.1) allowing comparison of the likelihood that a transaction linked to a certain *Mitigation-Target* or *Mitigation-Method* meets NNL in terms of *Ratio Equivalency*.

Matching Impact-Type to Offset-Type – ecological equivalency

Ecological Equivalency between impact and offset is an important aspect of the offsetting process. In a concrete example, a compensation lake was constructed in the Northern Boreal to replace lost habitat area and ecosystem function due to its loss through mining operations in the Alberta oil sands region (Ruppert et al. 2018). This example shows how an impact on habitat area and ecosystem function requires an offset that provides both aspects. *Impact-Type* to *Offset-Type* as an indicator for *Ecological Equivalency* of a transaction, was investigated by converting each impact for a transaction (if the information was available; n = 4331) to *Area and Function Los*s (loss of habitat area and ecosystem function) or *Function Loss* (degradation of ecosystem function but not area). Offset-Type for the transaction was determined by assuming that *Preservation*, *Enhancement*, and *Rehabilitation* lead to ecosystem function gain (*Function Gain*) and that *Establishment* and *Re-establishment* lead to habitat area and ecosystem function gain (*Area and Function Gain*). In our example from the oil sands, an offset through large scale stream enhancement, though potentially meeting *Ratio Equivalency*, would be deemed as not meeting *Ecological Equivalency* since it does only add ecosystem function, but not new habitat area compared to the initial loss. An appropriate match was labeled as *Match* (e.g., *Function Loss* Impact-Type matched with *Function Gain* Offset-Type). A mismatch resulting in a loss (*Area and Function Loss* Impact-Type matched with *Function Gain* Offset-Type) was labeled as *Mismatch*. Finally, a positive mismatch (*Function Loss* Impact-Type matched with *Area and Function Gain* Offset-Type) was labeled as *Overcompensate* since it adds both ecosystem function and habitat area value. This basic conversion allows us to determine if the Impact-Type matches the Offset-Type (e.g., a *Function Loss* Impact-Type should be associated with a *Function Gain* Offset-Type or *Area and Function Gain* Offset-Type) and how well bank transactions are approaching *Ecological Equivalency*, as other studies indicate a disjoint between NNL based on *Ratio Equivalency* and *Ecological Equivalency* (e.g., Carreras Gamarra & Toombs 2017; Zu Ermgassen et al. 2017). Impact and offset information were collected through a web scraping procedure based on transaction ID and keywords relating to *Mitigation-Methods* and ecosystem function or habitat area loss (rvest 1.0.2; Wickham 2021).

Preservation targets

Benefits provided by different Preservation targets measures were investigated by extracting detailed bank information for 64 banks operating mainly through Preservation. Benefits were based on the main targets and goals (n = 6) listed in reports for the individual banks (*Hydrogeomorphology* – HGM physical /chemical; *Invasive species control* – Invasive species presence thresholds and removal; *Habitat quality*; *Breeding pairs/ abundance* - e.g., minimum breeding pairs for a species – Habitat connectivity aspects – *Connectivity* and vegetation cover and presence thresholds – *Vegetation*). Benefits provided through Preservation targets received additional attention compared to other Mitigation-Methods since Preservation is often associated with no additional ecosystem function or habitat area gain in RIBITS which does not adequately represent the value of Preservation for different Mitigation-Targets (Bonde & Pompe 2003; Grimm & Köppel 2019; Sonter et al. 2019).

Statistical Analysis

All statistical analysis was done in R, version 4.1.0 (R-Core team 2020) and GitHub extensions.

Bank types across regions

To identify bank types that are characteristic of certain regions, we grouped banks into the five regions and calculated the proportionate presence of the six distinct banking types (Wetland; Species; Stream; Multi-Ecosystem; Multi-Species; Group). We used Pearson's Chi-squared test of independence for frequency analyses (Alpha < 0.05; Table A3.1.1). Results were plotted in a correlation plot, showing negative or positive correlations between regions and specific bank types based on residuals (corrplot 0.2-0). Significant correlations were tested for through post-hoc tests with an accepted Alpha of 0.05 and Bonferroni correction (Table A3.1.1; chisq.posthoc.test 0.1.2; Ebbert 2019).

Compensatory Mitigation-Methods for banking

To determine whether Mitigation-Methods have changed in their popularity and frequency of application over the years, transactions for the five Mitigation-Methods (Enhancement; Establishment; Re-establishment; Rehabilitation; Preservation; n = 12576) were analyzed through a linear model testing for trends for each method's utilization frequency over time (Proportion of transactions for each method per total transaction each year; 1995 to 2020). Significant increases or decreases in proportions were identified through accepted Alpha values of 0.05 and effect size estimated by r-squared values (R^2; Table A3.1.2; Hamilton et al. 2015).

Gain:Loss ratios – Ratio Equivalency

To elucidate if certain Mitigation-Targets or Mitigation-Methods were associated with meeting or missing NNL criteria, *Gain:Loss* ratios and their described four categories (Loss; Partial; NNL; Gain) for each of the 14010 transactions were analyzed through Pearson's Chi-squared test of independence for frequency analyses for *Gain:Loss* categories across Mitigation-Targets (Wetland; Stream; Species; Group; Table 3.1.1; Table A3.1.3) and used Mitigation-Method (*Enhancement; Establishment; Re-establishment; Rehabilitation; Preservation*; Table 3.1.1) with an accepted Alpha of 0.05. Results were plotted in a correlation plot, showing negative or positive correlations between *Gain:Loss* categories and specific Mitigation-Targets and Mitigation-Methods based on residuals (corrplot 0.2-0). Significant correlations were tested for through post-hoc tests with an accepted Alpha of 0.05 and Bonferroni correction (Table A3.1.3; A3.1.4; chisq.posthoc.test 0.1.2; Ebbert 2019). In detail, transaction numbers were plotted into a Sankey diagram showing transaction numbers per Mitigation-Target (Wetland; Stream; Species;

Group); Mitigation-Methods for each target and number of transactions associated with the four *Gain:Loss* categories. Sankey diagrams, showing a flow from one set of values to another, in our case allow a more in-depth look at how *Mitigation-Method* use and *Gain:Loss* categories vary in terms of transactions and NNL outcomes across different *Mitigation-Targets* like wetlands or species (networkD3 0.4; Allaire et al. 2017).

Impact-Type to Offset-Type – ecological equivalency

Impact to offset type corresponding to the three categories *Match*; *Mismatch* and *Overcompensate* were analyzed through Pearson's Chi-squared test of independence for frequency analyses for impact to offset type categories across *Mitigation-Targets* (wetland; stream; species; group; Table A3.1.5) with an accepted Alpha of 0.05. Results were plotted in a correlation plot, showing negative or positive correlations between *Impact-Type to Offset-Type* categories and specific *Mitigation-Targets* based on residuals (corrplot 0.2-0). Significant correlations were tested for through post-hoc tests with an accepted Alpha of 0.05 and Bonferroni correction (Appendix S5; chisq.posthoc.test 0.1.2; Ebbert 2019). The correlation plot and post-hoc analysis helps us determine if *Impact-Types* are generally matched with appropriate *Offset-Types,* and if not, whether these mismatches are related to specific *Mitigation-Targets* like wetlands or streams.

Preservation targets

Identified *Preservation* targets (n = 6) from the mentioned 64 banks were plotted in a simple heat map across the five regions (*Northeast; Southeast; Midwest; West; Southwest*) based on the frequency of targets mentioned by bank and region, with multiple targets being possible per bank (lattice 0.20-45; Sarkar 2021). This way we do not only showcase the diverse targets that preservation efforts can aim at but also are able to relate these findings back to differences in bank type distribution across regions.

3.1.3 Results

Bank types across regions

1736 banks were evaluated across the continental United States (Figure 3.1.2). 21% of the banks were in the *Midwest* (n = 369), 11% in the *Northeast* (n = 199), 13% in the *West* (n = 222), 49% in the *Southeast* (n = 845) and 6% in the *Southwest* (n = 101; Figure 3.1.2). Bank type distribution was significantly different across regions (Chi-squared = 208.8; df = 24; $p < 0.001$; Table A3.1.1). Most species related banks were in the western parts of the United States. The *West* was strongly associated with *Species* banks (18%; $p < 0.05$) as well as *Multi-Species* (15%; $p < 0.001$) and *Group* banks (15%; $p < 0.001$; Table A3.1.1). The *Southwest* was strongly correlated with *Species* banks (19%; $p < 0.05$; Table A3.1.1). Moving from west to east, banks in the *Midwest* and *Northeast* were mostly focused on wetlands (*Midwest:* 77%; $p < 0.001$; *Northeast*: 62%; $p < 0.05$; Table A3.1.1). Finally, the *Southeast* was distinguished by a high proportion of *Stream* banks from the other four regions (23%; $p < 0.05$).

Compensatory Mitigation-Methods for banking

We found that over the past 25 years, *Mitigation-Methods* recorded in the assessed transactions (n = 12756) changed in their utilization frequency (Figure 3.1.3A). *Enhancement* and *Preservation* measure usage did not change significantly over time with 16.2% (± 5.2) of yearly transactions on average for *Enhancement* and 29.4% (± 11.3) for *Preservation* transactions (Figure 3.1.3A). *Rehabilitation* measures (14.6% ± 5.5 yearly average) had a strong positive trend associated with them ($R^2 = 0.74$; $p < 0.001$; Table A3.1.2), increasing from less than 5% in the 1990s to around 15 to 20% of all yearly transactions in 2010 and onward (Figure 3.1.3A). *Re-establishment* (30.9% ± 7.8 yearly average) and *Establishment* (8.9% ± 6.9 yearly average) transactions decreased over time. Yearly *Establishment* proportionate transactions decreased by around 6% between 1995 and 2020 ($R^2 = 0.37$; $p < 0.001$; Table A3.1.2). *Re-establishment* proportionate transactions decreased by around 10% over time ($R^2 = 0.25$; $p < 0.05$; Table A3.1.2). Overall *Re-establishment* and *Preservation* were the predominant *Mitigation-Methods* from 1995 to 2020 with *Rehabilitation* usage rapidly increasing over time and *Establishment* and *Re-establishment* usage decreasing.

Gain:Loss ratios – Ratio Equivalency

Gain:Loss ratios differed across *Mitigation-Targets* (Figure 3.1.3B; Chi-squared = 419.02; df = 9; $p < 0.001$; Table A3.1.3). Transactions targeting wetlands (n = 10628) using mainly *Preservation* (37.6%), *Enhancement* (29.1%) and *Re-establishment* (24.7%; Figure 3.1.4A) were strongly associated with *NNL* (35.9% of transactions; $p < 0.001$) and *Gain* (37.7% of transactions; $p < 0.001$; Figure 3.1.3B). Stream transactions (n = 1647) mainly relying on *Rehabilitation* (39.1%), *Preservation* (25.5%) and *Enhancement* (20.6%; Figure 3.1.4B) were strongly related to *Partial NNL* (37.2% of transactions; $p < 0.001$) and *Loss* (12.6%; $p = 0.001$; Figure 3.1.3B). Transactions targeting species (n = 151), often relying on *Preservation* (59.6%; Figure 3.1.4C) were strongly associated with *NNL* (43.7% of transactions; $p < 0.001$; Figure 3.1.3B). Half of the group transactions (n = 330) were linked to *Preservation* and 26.9% to *Enhancement* (Figure 4d). Group transactions were positively related to *NNL* (44.5% of transactions; $p < 0.05$; Figure 3.1.3B).

Results for the Chi-Squared tests for *Mitigation-Methods* showed that *Gain:Loss* ratios differed between the methods (Figure 3.1.3C; Chi-squared = 5234.5; df = 12; $p < 0.001$; Table A3.1.4). Transactions linked to *Preservation* (n = 4671; Figure 3.1.4) were the most likely to lead to *Gain* ($p < 0.001$; Figure 3.1.3C). *Enhancement* (n = 3564; Figure 3.1.4) was associated with *NNL* ($p < 0.001$) and *Partial NNL* ($p < 0.001$; Figure 3.1.3C) and so was *Re-establishment* (n = 2848; *NNL*; $p < 0.001$; *Partial NNL*; $p < 0.001$; Figure 3.1.3C). *Rehabilitation* (n = 1242; Figure 3.1.4) and *Establishment* (n = 449; Figure 3.1.4) were likely to lead to *Loss* (*Rehabilitation*; $p < 0.001$; *Establishment*; $p < 0.001$; Figure 3.1.3C) and *Rehabilitation* to *Partial NNL* ($p < 0.001$). Overall *Preservation* was the most utilized *Mitigation-Method* for the four *Mitigation-Targets* followed by *Enhancement*, *Re-establishment*, and *Rehabilitation*. Wetland transactions and utilization of *Preservation* were the most likely to lead to *Gain* while stream transactions were most likely to only meet *Partial NNL* in terms of *Gain:Loss* ratios. *Enhancement* and *Re-establishment* were likely to meet *Partial* or *NNL* criteria. Species and group transactions mostly led to *NNL*. *Rehabilitation* and *Establishment* had the highest likelihood for leading to *Loss* in terms of *Gain:Loss* ratios.

Matching Impact-Type to Offset-Type - ecological equivalency

Investigating matches of *Impact-Type to Offset-Type* through Chi-squared tests of 4331 transactions (Figure 3.1.5B; Chi-squared = 98.005; df = 6; $p < 0.001$; Table A3.1.5) revealed that wetland transactions (n = 3702) were matching *Impact-Type to Offset-Type* the most often (79.5%; $p < 0.001$; Figure 3.1.5B) with *Mismatches* (23.9%) being mostly attributed to *Preservation* not meeting habitat area and ecosystem function loss. Stream transactions (n = 501) were the most likely to *Mismatch Impact-Type and Offset-Type* (25.6%; $p < 0.001$) or *Overcompensate* (10.0%; $p < 0.001$; Figure 3.1.5B), with *Mismatches* being linked to *Rehabilitation* (47.7%). Species transactions (n = 37) were not significantly related to any match type but fell mostly into *Match* (78.4%) and *Mismatches* (18.9%), with the latter being associated with *Preservation* (57.1%). Group transactions (n = 91) were strongly associated with *Overcompensation* (11.0%) between *Impact-Type* and *Offset-Type* ($p < 0.05$; Figure 3.1.5B). Cases of *Mismatch* in group transactions were in 55.6% of the cases due to utilizing *Enhancement* for habitat area and ecosystem function loss.

Preservation targets

The heat plot (Figure 3.1.5A) shows that overall *Hydrogeomorphology* (physical and chemical; n = 48 mentions), as well as parameters relating to *Vegetation* (n = 44 mentions), were the most common targets for banks using high frequency of Preservation. These targets were mostly associated with banks in the *Midwest* and *Southeast* and *Northeast*. *Invasive species control* was mainly associated with banks in the *Southeast* (n = 8 mentions) and *West* (n = 11 mentions) as well as species abundance and minimum amounts for breeding pairs (*Southeast*; n = 4 mentions; *West*; n = 6 mentions). *Connectivity* aspects were only mentioned in banks in the Southeast (n = 3 mentions) and *West* (n = 4 mentions) and *Habitat Quality* mostly just in the *West* (n = 7 mentions). Overall *Hydrogeomorphology* and *Vegetation* aspects were the predominant targets for Preservation focused banks while Preservation focused banks display a wide array of different targets distributed across the five geographic regions.

3.1.4 Discussion

The key findings of this study are:

Bank type distribution was inherently different across regions. *Species* focus was predominant in the western regions, *Wetland* focus in the *Midwest* and *Northeast* and an increased numbers of *Stream* banks in the *Southeast*. Usage frequency of *Mitigation-Methods* on a yearly basis has changed over time. *Rehabilitation* usage has increased from 1995 to 2020 and decreased for *Establishment* and *Re-establishment*. *Preservation* and *Re-establishment* were the most popular *Mitigation-Methods*. *Gain:Loss* ratios indicate NNL on a ratio basis for most transactions with wetland targets. Stream targets led to *Partial NNL*, and *Loss*. Species and group targets led to *NNL* on a ratio basis. *Preservation* as a *Mitigation-Method* was associated with ratio *Gains* while *Enhancement* and *Re-establishment* were likely to meet *NNL*. *Rehabilitation* and *Establishment* most often lead to *Partial NNL* or *Loss*. Wetlands matched *Impact-Type to Offset-Type* well but showed a high likelihood of misusing *Preservation*. Stream transactions tended to *Mismatch* or

Overcompensate, with *Rehabilitation* often not matching *Impact-Types*. Group transactions *Overcompensated Impact-Type to Offset-Type* while species transactions generally led to a *Match*. Investigating our findings requires looking at different aspects of banking; ecosystem availability and banking history as well as economic and ecological feasibility of *Mitigation-Methods* and *Mitigation-Targets*.

Bank types across regions – banking history and legislation of the United States and ecosystem availability

One of the main reasons for regional differences of bank types across the United States is the history of banking and management systems. Overall, there was a large focus on using banking for wetlands, likely since wetland banking was the first banking system with a legislative framework, dating back as early as the 1970s (Burgin 2009). Previous estimates suggest that around 75% of the United States wetlands are in private ownership, suggesting that transforming these wetlands into banks is another reason for the large focus on wetland banking (Scodari et al. 1995). Adoption of banking practices for species and riverine systems or more holistic mixed approaches have only come about more recently in all regions (Lawrence 2001).

Ecosystem availability and land use are important considerations for which *Mitigation-Methods* and *Mitigation-Targets* to focus on. Both the *Midwest* and *Northeast* have a heavy focus on wetlands, but also have a large proportion of agricultural area (Brown et al. 2005), in combination with some of the highest wetland losses in all of the United States (mainly Indiana, Illinois, Iowa, Ohio & Missouri; Dahl 1990 & 2000). The *Southeast* also had a high proportion of *Wetland* banks, but it also had a large focus on riverine systems and *Multi-Ecosystem* banks. Many *Stream* banks in the *Southeas*t can be attributed to the lower Mississippi delta in combination with a high degree of species' endangerment (e.g., Mobile Bay Rivers; Kesel 1989). Consequently, *Stream* and *Wetland* banks, or a combination of both, have been established in the *Southeast*. Similarly, the *w*estern regions incorporated a significantly higher percentage of *Species* banks and even *Multi-Species* or *Group* banks which were virtually non-existent in the other three regions.

These trends suggest on the one hand that the availability of ecosystems plays an important role in the distributional trends of banking types and on the other hand that higher availability, of for instance wetlands or streams, ultimately will lead to more impacts needing offsetting, increasing banks for these targets. For instance, the rapid development of conservation banks for preserving wildlife communities in California can be linked back to large-scale, rapid urbanization and many unique and sensitive ecosystems and communities like the chaparral (Bunn et al. 2014; Venturas et al. 2016). The strong species focus in the *West* and *Southwest* is likely a relic of the legislation in those areas. Going back to 1995, California started creating species-specific banks for endangered or threatened species, which is widely acknowledged as the first proper functioning species bank model (Lawrence 2001). The focus on species contrasts other regions where other agreements and legislative policies were in place (e.g., wetland banking agreements; habitat conservation plans; Fox & Nino-Murcia 2005). Guidelines released by the USFWS (2003) and USACE (1995) have made banking approaches more uniform, while some differences remain considering the analyzed data. Evaluations within the next 20 to 30 years should reflect these changes.

Gain:Loss ratios and Impact to Offset types - How likely is a Ratio and Ecological Equivalency across Mitigation-Targets and Mitigation-Methods?

Wetland targets

Wetlands as part of bank transactions were the most common *Mitigation-Targets* and used a high frequency of *Preservation*, *Re-establishment*, and *Enhancement* to provide ecosystem function and/ or habitat area. *Re-establishing* wetlands in the United States has a long-standing history and well-established guidelines, supported by historic records on habitat area and ecosystem function (USEPA 2000; USFWS 2013; 2014). *Re-establishment*, while decreasing in popularity over time, is still one of the most effective and widely applied *Mitigation-Methods*. Though time-consuming, costly, and labor-intensive, *Re-establishment* is more likely to achieve the goals of *Ecological Equivalency* compared to other methods by considering aspects like connectivity, water quality, flow rates, food supply, and the development of spawning, nursery, and rearing habitat availability (USFWS 2013; 2014). An example is the wetland *Re-establishment* in the Sacramento-San Joaquin Delta in California showing tangible results for long-term colonization and wetland progression (Miller & Fuji 2010). *Re-establishment*, providing habitat area and ecosystem function is the main reason for wetlands matching *Impact-Types to Offset-Types* well by re-establishing historic wetlands. This trend is encouraging as it shows that *Ecological Equivalency* will likely be met in addition to meeting *Ratio Equivalency*.

Preservation was the main cause for mismatches of *Impact-Type to Offset-Type* in wetland transactions. This ecological mismatch is concerning. The United States has lost large proportions of its wetlands in the past thus incorporating banks into increased efforts to preserve the remaining ones (USFWS 2013; 2014). This fact has likely led to the high usage of *Preservation* in banks. From an economic standpoint, *Preservation* seems appealing due to the lower monetary and time investment to maintain an existing ecosystem. Furthermore, banking is often targeted at current or former farmland or degraded areas that are adjacent to ecologically important areas like wetlands which explains an increased interest in preserving adjacent systems while converting degraded developed land back into viable ecosystems (Liebesman & Plott 1998; Dahl 2000). Studies suggest that in the case of banking, inefficient long-term management can reduce diversity in *Mitigation-Methods* (Matthews & Endress 2008). In a chosen example case study extracted from RIBITS, bank management and funding changed over time in a wetland bank in Virginia. Bank area was consequently reduced by 50% as well as reducing the amount of initially designated area for *Re-establishment* and *Rehabilitation* while keeping the *Preservation* area almost constant. These often-occurring changes and shifts showcase how non-compliance, unrealistic goals, and/or management shortcomings can reduce utilization of certain *Mitigation-Methods*, partially explaining the consistent popularity of *Preservation* measures (Matthews & Endress 2008). While our results show a multitude of potential benefits provided through *Preservation*, many of them are not sufficiently supported by scientific evidence to meet *Ecological Equivalency* (e.g., Zu Ermgassen et al. 2017; Maron et al. 2016). Basic maintenance measures like plant cover or reduction of the number of non-native species are often a poor measure of overall habitat quality, often deemed too lenient with some essential ecosystem types like dry end wetlands being overlooked almost completely (e.g., Fox & Nino-Murcia 2005; Gutrich & Hitzhusen 2004; Matthews & Endress 2008).

Overall, wetland banking provides high *Gain:Loss* ratios due to high ratios of *Preservation* supported by other studies (e.g., Kihslinger et al. 2019) and matches *Impact-Type to Offset-Type* well, especially through *Re-establishment* and *Enhancement*. However, a quarter of the assessed transactions inadequately matched *Impact-Type to Offset-Type*, missing *Ecological Equivalency*, due to *Preservation*. While *Preservation* fills an important role in protecting existing wetlands in the United States its role in habitat banking should be revised to better match *Impact-Type to Offset-Type* and bridge the gap between *Ratio Equivalency* and

Ecological Equivalency (Zu Ermgassen et al. 2017; Fox & Nino-Murcia 2005; Gutrich & Hitzhusen 2004; Maron et al. 2016; Matthews & Endress 2008).

Stream targets

Stream targets often only reached *partial NNL* through *Gain:Loss* ratios, as well as mismatching habitat area and ecosystem function losses with just ecosystem function gain through riverine *Rehabilitation*, missing *Ecological Equivalency* which is supported by other studies focusing on streams in the United States (e.g., Palmer & Hondula 2014). Compensatory mitigation for streams is a difficult target to approach in proponent-led offsets and it seems that habitat banking is suffering from similar issues (e.g., Gibson et al. 2005; Sweeney et al. 2004). Literature suggests that three main problems persist for effectively offsetting stream impacts: habitat availability, use of improper offsetting techniques, and underestimation of development impacts (Gibson et al. 2005; Palmer & Hondula 2014; Sweeney et al. 2004).

Habitat availability for *Re-establishing* or *Establishing* streams is sparse. Stream construction (*Establishment*) often needs to be incorporated into larger projects or landscape-level planning. The same applies to *Re-establishment* which in most cases are large-scale and expensive projects, potentially suited for Umbrella banks but not feasible or available to most private banks (Roni et al. 2008). *Overcompensation*, as shown in our results for streams can be attributed to these large projects exceeding impacts both on a ratio as well as ecological level. *Enhancement* or *Rehabilitation* is the most logical choice when it comes to streams, especially considering small, degraded, farmland-adjacent streams or habitat area loss in urban streams where the area for physical habitat creation is simply not available (Larson et al. 2001; Suren 2009; Sweeney et al. 2004). This disjoint between losing stream habitat area and offsetting it through function gains will ultimately lead to a consistent loss in-stream habitat area. Underestimated development impacts and improper offsetting techniques make it even harder to provide accurate estimates on Impact and Offset quantities but can be improved through more comprehensive assessment and mitigation methods such as Index of Biotic Integrity (IBI; Gibson et al. 2005; Teels et al. 2004; Roni et al. 2008; Sweeney et al. 2004). Considering that many stream transactions do not meet Ecological and *Ratio Equivalency*, the increased number of permit authorization involving stream habitat area and ecosystem function, and an increase in *Rehabilitation* usage as the main *Mitigation-Method* for stream impacts, underlines the need to rework the current approach to compensate for losing physical stream habitat area (Lave et al. 2008; Teels et al. 2004).

Group and Species targets

Overall, *Preservation* was popular for group and species targets with both targets generally meeting *Ratio Equivalency* as well as *Ecological Equivalency* with group targets often *Overcompensating* through *Establishment* and *Re-establishment*. *Preservation*, being popular for both *Mitigation-Targets*, can be attributed to species and group targets often being associated with conservation banks. Conservation banks aim to permanently protect and manage sites for endangered species, threatened species, or species at risk. The aim is to offset adverse impacts to the protected species occurring off-site. Benefits for (endangered) species for example can be the preservation of corridors linking two habitat patches or protecting wintering habitats (Fox & Nino-Murcia 2005). Species not able to exist in modern transformed semi-natural and cultural landscapes will require larger more complex efforts currently not achievable by individual banks but targeted through Umbrella banks (Webb 2008). An example here is pilot transboundary projects like the restoration agreement between the United States and the Dominican Republic for protecting overwintering habitats for migratory birds abroad, benefiting US bird populations (USFWS 2011). Overall preserving already existing habitat is perhaps one of the only feasible methods for banking in some cases, and for many endangered and threatened species. However, mismatches of meeting ecological equivalency for species were due to Preservation not providing any additional habitat area. On a species level, *Re-establishment* tends to be more difficult but still often yields successful results for instance in the case of the San Martin titi monkey at private conservation areas (*Plecturocebus oenanthe*; Allgas et al. 2016) or endangered aquatic plant communities in the Czech Republic (Kaplan et al. 2014). These examples and our results showcase the opportunity *Re-establishment* poses in providing vital habitat for endangered species. Overarching management, like Umbrella banks, could help facilitate utilizing *Re-establishment* more for endangered species, as simple habitat *Preservation* will likely not be enough to slow the current loss of biodiversity (e.g., Berton et al. 2012; Pullin 1997; Sonter et al. 2019).

Group targets often led to ratio-based *NNL* as well as *Overcompensating Impact-Type* and *Offset-Type*, which introduces increased additivity, another important aspect for current and future banking practice. While many banks combine different *Mitigation-Methods* in their bank areas (e.g., bank area = 10 acres comprised out of 20% *Rehabilitation*; 70% *Preservation*; 10% *Enhancement*), multiple *Mitigation-Targets* tend to create positive feedback loops between different ecosystems or a species and an ecosystem (e.g., Cimon-Morin & Poulin 2018; Strassburg et al. 2020). Group targets furthermore make use of identifying priority areas for both ecosystem function and biodiversity (Cameron et al. 2017; Rondinini & Chiozza 2010; Strassburg et al. 2020). *Mismatches* due to *Enhancement* measures, show that a few specifically enhanced ecosystem aspects are often not enough when aiming at multiple species or ecosystems with the risk of *Enhancement* in one area being detrimental in another area (Langler & Smith 2001; Salt & Freudenberger; Sonter et al. 2019). Group transactions still only comprise a small proportion of the overall number of bank transactions but hold the potential to provide valuable guidance on how to reach better *Ecological Equivalency* for other *Mitigation-Targets*.

3.1.5 Conclusions

Our study shows that *Ratio Equivalency* is achieved in most bank transactions in the United States, while there was a clear disjoint in achieving *Ecological Equivalency*, which is in line with other studies (e.g., Zu Ermgassen et al. 2017; Kihslinger et al. 2019; Maron et al. 2016; Sonter et al. 2019). The main reasons for *Mismatches* were the high usage of *Preservation* in wetland transactions and *Rehabilitation* in stream transactions. While *Rehabilitation* has increased in its use frequency over the past 25 years, its potential for ecological equivalency seems limited in the context of stream impacts, which often lead to loss of habitat area. *Mismatches* are potentially due to improper offsets, area availability, costs, and official guidelines focusing on area ratios rather than *Ecological Equivalency* (Gibson et al. 2005; Palmer & Hondula 2014; Sweeney et al. 2004). *Preservation* is often overused in wetland and transactions that, despite greatly exceeding *Ratio Equivalency*, do not meet *Ecological Equivalency* in one-quarter of all assessed wetland transactions. While we showcase the wide array of potential *Preservation* benefits, *Preservation* should not be used as a Panacea since it does not meet requirements for additionality and *Ecological Equivalency*. *Re-establishment* for wetlands showed a high frequency of meeting *Ratio* and *Ecological Equivalency* while being one of the most widely used *Mitigation-Methods* which is an encouraging finding. *Preservation* plays an important role for conservation banks and their role to protect habitat for endangered species which showed even higher rates for *Ecological* and *Ratio Equivalency* when part of group transactions, targeting multiple species or ecosystems mostly attributed to adaptive management and positive feedback loops (e.g., Cimon-Morin & Poulin 2018; Strassburg et al. 2020).

Chapter 3.2: Current capacity, bottlenecks, and future projections for Mitigation and Conservation banking in the United States based on the past 26 years

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Summary

Habitat banking in its many iterations is an established and popular mechanism to deliver environmental offsets. The United States can look back at over 30 years of banking experience with the underlying framework and policies being consistently updated and improved. Given the increased demand in habitat banking, we provide insights into how bank area capacity is distributed across the United States for four different bank targets (wetlands, streams, multiple ecosystems, species) based on information extracted from the Regulatory In-lieu Fee and Bank Information Tracking System as well as estimating future capacities and area reserves through a predictive modeling approach based on data from the past 26 years. Future predictions indicate a decrease in available reserves for banks targeting wetlands or multiple ecosystems, with potential bottlenecks relating to large reserves being limited to the southeast and release schedules not catching up to the current and anticipated demand. Banks targeting species or streams are predicted to meet future demand, with species banks (conservation banks) following a different legislative and operational approach based on the listing of endangered species. The majority of

current reserves for all four bank types is restricted to very few service areas with around one-third of all bank areas still awaiting release, limiting their availability on a broader scale. Strategic planning networks are necessary to meet future demand on a national scale and to identify areas suitable for banking or likely to experience future environmental or developmental stress.

Abbreviations

YAC - Yearly Added Capacity (Area in acres/ linear feet)

UCA – Unreleased Capacity (Area in acres/ linear feet)

AUR - Accumulated Unwithdrawn Reserve (Area in acres/ linear feet)

3.2.1 Introduction

Relevance of offsetting

Increased land use and development and the associated loss in habitat, ecosystem function, and biodiversity have led to the implementation of legislative requirements and frameworks in many countries, to prevent further losses or utilize equivalent compensatory measures (McKenney & Kiesecker 2009). Offsetting, aimed at simultaneously allowing anthropogenic development while ensuring appropriate mitigation and compensation measures meeting No Net loss (NNL) requirements, has been a widely implemented yet often understudied tool (Bull et al. 2013; Gardner et al. 2013). No Net Loss as a goal refers to the practice of providing mitigation or compensation measures that are equal to or outweigh the harmful impact exerted by a development project or anthropogenic activity (Bull et al. 2013; McKenney & Kiesecker 2009; Figure 3.2.1A). Allowing a harmful activity or project and compensating said impacts through offsetting is only permitted after following the previous steps in the mitigation hierarchy; avoidance, minimization, and restoration or rehabilitation (McKenney & Kiesecker 2009; Figure 3.2.1A). Offset gains can be "in-kind", meaning similar to what is lost (biodiversity; ecosystem function; habitat area) or "out-of-kind", with the latter often being a question of feasibility or flexibility (Bull et al. 2015). Offset losses can occur over time due to non-compliance by the proponent of the development project, failure of the constructed offset due to impaired function, structural degradation, or inadequate scope as well as exempt (not yet regulated) or unaccounted impacts occurring periodically or as one-time events post offset construction (e.g., Arlidge 2018; Theis et al unpublished).

What is habitat banking?

Habitat banking comprised out of conservation and mitigation banking in the case of the United States, as a special case of the traditional offsetting mechanism, has gained significant traction over the past decades (Santos et al. 2015). While banking follows the same mitigation methods and mitigation hierarchy as regular offsetting practice dictates (Bull et al. 2013; Figure 3.2.1), these created, enhanced, restored, or preserved areas are established and managed by a third party as opposed to the proponent itself comprised of the bank sites, banking agreement and service area (area in which the bank can sell their credits) (USEPA 1995; Santos et al. 2015; USACE 2008; 2015). Proponents instead purchase a required credit amount for their expected impacts, corresponding to an equal species, and ecosystem service or habitat value from the bank, set by the responding agency or government body (Burgin 2008; Figure 3.2.1B). This paper, unless specified, will refer to 'banks' as the sum of site, credits, agreement and service area.

Current banking practices, issues, and demand in the United States

The United States banking framework is based on two different legislations leading to the establishment of Mitigation and Conservation banks. There are different mechanisms for compensating for the approved adverse effects under each legislation. A proponent can be solely responsible for the offset (permittee responsible), pass responsibility to a third party (third-party, e.g., credits from a bank) or has the option to pay money into a compensation fund (In-Lieu fee; ILF) managed by a nonprofit or government in lieu programs, which in term use the funds for current or future offsetting activities (USEPA, 1995; Fleischer & Fox 2012; Table 3.2.1). The earliest legislation pertains to Section 404 of the Clean Water Act (1972), aiming to protect wetlands in the United States and compensate for negative impacts, with negative impacts permitted through the United States Army Corps of Engineers (USACE). Mitigation Banks refer to banks selling credits used to offset negative impacts on streams or wetlands (USEPA 1995; Stein 2000; USACE 2008; 2015; Table 3.2.1). Mitigation banks have been widely established throughout the United States for instance to protect coastal wetlands in the Southeast or to compensate for urbanization in the metropolis area of the Northeast or agricultural land development in the Midwest impacting vital ecosystems like prairie wetlands (FWS 2013 & 2014). Mitigation banks have the official goal of NNL and guidance by the United States Environmental Protection Agency (USEPA) has been adjusted over the years in Memorandum Agreements. Conservation banking, modeled after mitigation banking, and based on the Endangered Species Act (1973) refers to landowners permanently protecting species habitat, translated into credits that can be sold to proponents affecting species habitat somewhere else. Conservation banks are approved by the Fish and Wildlife Service (FWS 2003; Fox & Nino-Murcia 2005; Table 3.2.1). Conservation banks and their underlying framework were mainly developed along the West Coast of the United States (California 1995) to preserve and protect endemic species and unique ecosystem types like the Chaparral (FWS 2003; Venturas et al. 2016). Lastly, so-called Umbrella Banks exist which are established to run multiple offsetting sites on a regional level under single institution funding (Carreras Gamarra & Toombs 2017; Table 3.2.1). Largely regarded as the pioneers of banking, the United States can look back at more than 30 years of available data and thousands of established banks (Carreras Gamarra & Toombs 2017; FWS 2003; White 2012). Accepted advantages for mitigation banks and conservation banks over traditional offsets are perpetual contracts and long-term management of the banked area as well as being able to protect larger connected areas as opposed to multiple spaced out and not connected offset parcels. Purchasing credits before the impact approval can reduce the time for a proponent to receive their impact permit thus providing potential business advantages (e.g., Berlin & Malone 2019; Field 2015; White 2012)

While widely adopted, mitigation banking faces certain pitfalls and controversies, both ethically as well as operational and administrative (Bull et al. 2013, Maron et al. 2016). These persisting issues mostly concern the lack of transparency in reporting (Quétier & Lavorel 2011), poorly designed metrics for the area to credit conversion and a general tension between supporters of environmental versus economic priorities (Maestre-Andrés et al. 2020), and the lack of long-term funds to meet the perpetuity requirements (Boisvert 2015). Case studies show that long-term management is often difficult, coinciding with area or ownership changes in banks, reducing the effectiveness of provided offsets and benefits. Especially more complex mitigation methods like re-establishment or enhancement of ecosystem aspects can lead to time-lags, delayed credit release or failure, and consequent restructuring of the bank (Matthews &

Endress 2008; Figure 3.2.1B). Adding to that, administrative favoritism of preservation over creation, linked to cost-benefit ratios and the overall often complex and convoluted process of registration, evaluation, and approvals that reduces enforcement as well as asocial control aspects (Gorissen et al. 2020), lead to reduced effectiveness of the banking principle. Many current mitigation banks are targeted at restoring degraded farmland or preserving adjacent ecosystems often in anticipation of future losses. For instance, the Midwest and Northeast have been experiencing high degrees of wetland losses and degradation in past, making them suitable for mitigation bank practices to offset and prevent further losses (Dahl 1990 & 2000). However, preserving a habitat of low ecological value or not in immediate danger of loss or degradation on a large scale can lead to not meeting true ecological equivalency while potential negative impacts may never occur (Burgin 2009; Moreno-Mateos et al. 2015). While many mitigation banks target degraded farmland as mentioned before, others are created through immediate demand (e.g., for road construction, housing) that relies on fast approval and turnaround times (Bunn et al. 2013). This struggle to avoid time-lags while identifying and protecting vital ecosystems early enough is being tackled through more recent developments in landscape-level planning networks for instance in France and the Netherlands (Bunn et al. 2013; Moilanen 2013).

Biodiversity and impaired species are the drivers for an increased demand for conservation banks for preserving wildlife communities. For instance, early developments of conservation banks in California can be attributed to, rapid urbanization and many unique and sensitive ecosystems like the California Floristic Province with a uniquely high number of endemic plants (Bunn et al. 2014; FWS 2013; Venturas et al. 2016). Conservation banks have the advantage of permanently protecting habitats for endangered or threatened species while increasing connectivity between patchy habitats. However, like mitigation banking, preservation, rather than creating new habitat is the main concern for conservation banks. Creation component versus preservation component requirements for affected habitat is not always effective in preventing the overuse of preservation and the consequent loss of species habitat. Economic drivers such as price per acre per species vary greatly and have led to imbalances in conservation efforts, with the focus on 'expensive' species. Conservation bank establishment, conservation outcomes, and success criteria are still debated (Discussion papers 2003; 2007; Fleischer & Fox 2012; Poudel et al. 2019) with newer amendments to the Endangered Species Act from 2016 being officially withdrawn as of July 2018. Another noticeable development is the passing of

Bill 2087 in California which allows for large-scale mitigation credit establishment through Regional Conservation Investment Strategies by the FWS in the future (Assembly Bill No. 2087; 2016).

Biodiversity, as well as overall ecosystem function and habitat area losses, are increasing on a global scale and banking financed by the private sector could provide a mechanism to slow this loss in the future (e.g., Powers & Jetz 2019; Reid et al. 2019). However past and current issues need to be addressed for each of the two main banking types to ensure banking as a long-term sustainable offsetting mechanism (Gorissen et al. 2020).

Future interest & demand – is the future of banking secure?

Aside from the stated shortcomings and pitfalls one major aspect of banking practices, demand, and availability, is rarely considered and often lost in ecological and biodiversity-focused studies. As popularity and incentives in the market-driven environment of habitat banking increase so does the demand. Consequently, the number of banks and areas held by banks in the United States has increased steadily over the past 30 years (Poudel et al. 2018). However, using the collected data, assessing current capacities as well as future predictions on supply and demand has been done only insufficiently on a national level (Poudel et al. 2018; Saeed 2004; Sapp 1995; USACE 2015). Improving banking practices and avoiding pitfalls will be of limited use if reserves and the availability of banks and banked area cannot be ensured in the future. Understanding how both bank types (mitigation and conservation) operate in the United States and whether the current banking trends and performance will meet the increasing demand will improve existing and future guidance and policy decisions and help set the path for the future.

Therefore, we aim to answer the following research questions to assess current and future banking potential for conservation and mitigation banks: (1) Based on trends from the past 26 years, what are future predictions for bank reserves (currently available area) and, unreleased capacity (maximum available area in the future) across different bank types? (2) Which regions hold potential reserves and unreleased capacity and where or for what bank types are capacity bottlenecks likely to occur?

Given the high demand for banking credits in combination with banking being an established and consistently updated practice and the large proportion of degraded habitat in the United States being suitable for banking through restoration, we hypothesize that banks will be able to meet future demand based on newly added banks, bank size and operating capacities through forecast predictions (e.g., Poudel et al. 2018 & 2019; RIBITS 2022).

3.2.2 Methods

Dataset

Data for this study was acquired through the Regulatory In-Lieu Fee and Banking Information Tracking System (RIBITS), extracting information on 4055 banks and ILFS sites for the United States (last accessed December 18th, 2021). Only approved banks were included in this study to estimate current and future capacities, as well as banks having information on size, credit availability, and ledger transactions as well as bank type associated with them (Table 3.2.1). Furthermore, banks established before 1995 were excluded from the analysis due to the previous use of a non-uniform reporting system. We used bank data and transactions up to December transactions 31st 2020, basing predictions on whole years. Overall, the sorting process yielded 1636 banks with the necessary information available.

Key variables

Bank types

We divided the 1636 conservation and mitigation banks by their RIBITS designations which refer to targets. Mitigation banks were subdivided into banks targeting wetlands (Wetland; n = 897), streams (Stream; n = 253) or multiple ecosystems (Multi-Ecosystem; n = 385). Due to the low sample size, we combined conservation banks that target single or multiple species (Species; n = 101; Table 3.2.1).

Current and Future predictions for capacity and reserves – bank metrics

Bank metrics for current and future capacity and reserve were *Yearly Added Capacity (YAC)*, *Unreleased Capacity* (UCA), and *Available Unwithdrawn Reserve* (AUR).

Yearly Added Capacity (YAC)

Yearly Added Capacity (YAC) is based on the number of new banks each year per bank type and their summed area (Acres; Linear feet for Stream banks; Table 3.2.1). Incorporating bank number and size into this metric allows us to identify trends on whether bank size and number increase or decrease over time and how potential trends could play into future predictions. For instance, an increasing trend in bank numbers and size over the past 26 years would a) indicate an increasing demand in banking, as well as advances in policies that allow for bank establishment, and b), would indicate a likely future increase in yearly added bank numbers and size. *Yearly Added Capacity* is not cumulative, as it is calculated for each year independently (1995 to 2020).

***Unreleased Capacity* (UCA)**

This metric is based on the area for each bank type each year that has not been yet approved for release through credits (Table 1). For instance, a wetland bank founded in 1998 with 100 acres and 25 acres released would have a UCA of 75 acres in that year. UCA is due to release schedules and performance criteria that determine when and how much of a bank area can be released in the form of credits to be available for proponents. It is a useful indicator metric since it captures how fast bank release schedules are or how well banks meet performance criteria. UCA is a cumulative metric, meaning UCA from the previous year's affects UCA for the next year.

***Available Unwithdrawn Reserve* (AUR)**

The metric is based on the area for each bank type each year that is available to be withdrawn in the form of credits bought by proponents (Table 3.2.1). For instance, if a species bank had 45 acres of released area available that was not withdrawn in 2002, that would respond to its AUR. AUR is a cumulative metric, meaning AUR from the previous year's carries over to the next year until withdrawn.

Bottlenecks

UCA outpacing AUR indicates higher demand and slower release, potentially creating bottlenecks of unreleased areas (*release schedule bottleneck*) for specific bank types. AUR outpacing UCA indicates low demand for specific bank types/ demand not exceeding credit release or bank establishment. High AUR for a specific bank type could indicate a low demand (*demand bottleneck*) or regional restrictions based on bank location and service area (*regional bottleneck*). A decrease in YAC indicates a decrease in newly established banks per year and/ or bank size, consequently affecting UCA and AUR. For instance, if no banks are established in

2022, AUR and UCA would consequently decrease (*supply bottleneck;* Field 2015; Poudel et al. 2018; Watson et al. 2019).

Statistical Analyses

Future predictions for YAC, UCA, and AUR across bank types

Future predictions for YAC, UCA, and AUR were done through univariate time series modeling, based on the 26 years of data we extracted, through Auto-Regressive Integrated Moving Average (ARIMA) modeling, in R (4.1.0 R Core team 2020). The ARIMA model is generally used to derive information from past data to inform future predictions (Hyndman & Athanasopoulos 2021). We tested and selected a total of 12 individual models (YAC; UCA and AUR for each of the 4 bank types; Table A3.2.1 – A3.2.4). Each model predicted YAC, UCA, or AUR for the 4 bank types up to 2030. Predictions were done on a step-by-step basis, meaning the first prediction for instance for UCA for Wetland banks for 2021 was based on the data from 1995 to 2020 and the prediction for 2022 based on the data from 1995 to 2020 plus the 2021 prediction (Hyndman & Athanasopoulos 2021; Hyndman & Khandakar 2008). Each model was based on three main components (p = is the number of autoregressive terms (AR); d = is the number of non-seasonal differences needed for stationarity; q = is the number of lagged forecast errors in the prediction equation (MA). These components determine the model fit which is measured through the Akaike information criterion (AIC), Akaike information corrected criterion (AICc) and, Bayesian information criterion (BIC; Hyndman & Athanasopoulos 2021; Hyndman & Khandakar 2008). An in-detail example for model selection can be found in the supplements (S3.2.1).

Step one was to test for stationarity of the time series through a Dickey-Fuller test. Stationarity is a requirement that needs to be met before fitting the model. Significant results indicate that the stationarity requirement is given. Non-stationarity requires a stepwise correction. The number of corrections to reach significant results for the Dickey-Fuller test determines d. For instance, stationarity without correction necessary means d = 0, one correction means d = 1. Steps 2 and 3 included determining p and q which correct for autocorrelation. In step 2, q was determined through the ACF plot (Autocorrelation plot). The ACF plot is a correlogram showing serial correlation changes over time in the time series data (Supplements 5). Lags meeting significance in the plot determine p. For instance, an ACF plot with 2 lags meeting significance would result in p = 2. The same approach was used for q and the PACF plot (Partial autocorrelation plot). In step 4, after ensuring stationarity and determining appropriate p, d, and q terms for each model, AIC, AICc, and BIC were compared with other models provided by the auto function from the forecast package to rule out errors and ensure the fit model was selected. The final step was to check each model's residuals through a Ljung-Box test for autocorrelation of the residuals (non-significant results indicate no autocorrelation of residuals). After that, each of the 12 models was run to predict YAC, UCA, and AUR for the 4 bank types and forecasts plotted with 80 and 95% confidence intervals (Hyndman & Athanasopoulos 2021; Hyndman & Khandakar 2008). Current trends and future predictions for YAC, UCA, and AUR are meant to identify *release schedule bottlenecks, demand bottlenecks,* and *supply bottlenecks.* Trends in YAC, UCA and, AUR between 1995 and 2020 were analyzed through linear models (Response variable: YAC; UCA; AUR; Predictor variable: Year). Significant increases or decreases were identified through accepted Alpha values of 0.05 and effect size estimated by r-squared values (R^2; Hamilton et al. 2015).

Current reserves and capacities across bank types and regions

To showcase the status of the 4 bank types, we calculated the proportionate amount of withdrawn area for each bank type (not available anymore since sold to proponents) compared to UCA and AUR. Plotted as pie charts these estimates show if a specific bank type currently exhibits notable trends in terms of withdrawn or available area. If Wetland banks for instance had 99% of their total possible area withdrawn it would indicate a severe lack of currently available reserves (AUR) and future capacities (UCA). *Regional bottlenecks* and areas of high reserves were identified through selecting the top 100 banks with the highest AUR and UCA area values as of 2020, related to their designated bank type (Wetland; n = 56; Stream; n = 11; Species; n = 19; Multi-Ecosystem; n = 14) and mapped in GIS to capture their location and service area. Total capacity (all bank areas for a bank type summed), UCA, and AUR were and related to the overall proportion for each bank type. For instance, if all Wetland banks (n = 897) have a summed size of 100.000 acres, UCA of 25.000 acres and AUR of 20.000 acres and the 56 Wetland banks in the top 100 hold 60.000 acres, UCA of 10.000 acres, and AUR of 10.000 then that comprises 60% of the total Wetland bank capacity, 40% of total UCA and 50% of total AUR, pointing to large capacities and reserves sitting with a small number of wetland banks, potentially limited to specific regions which were identified through our map.

3.2.3 Results

Future predictions and reserves

Wetland Banks

YAC for Wetland banks over the past 26 years ranged from its lowest value of 5085 acres as part of newly established banks per year in 2006 to its highest yearly added value of 35919 acres in 2015 (mean: 16039 ± 12986 Acres; Figure 3.2.2A). There was no significant increase or decrease in YAC from 1995 to 2020 ($p = 0.254$; $R^2 = 0.015$; Table A3.2.5). Results from the ARIMA model for YAC for Wetland banks (1;1;1; AIC 552.65; Table A3.2.1) show that YAC for Wetland banks is predicted to be at 16554 Acres (95% CI: -13426|46535 Acres) in 2030 which is an increase of 3.2% from the previous yearly mean (Figure 3.2.2A). UCA for Wetland banks increased significantly over the past 26 years from 9546 Acres in 1995 to 129884 Acres in 2020 ($p < 0.001$; $R^2 = 0.88$; Table A3.2.5; Figure 3.2.2A). Future predictions from the ARIMA model (1;2;1; AIC 475.59; Table A3.2.1) show that UCA is predicted to increase to 175258 Acres by 2030 (95% CI: 66613|283903 Acres) marking a 34.9% increase. Like UCA, AUR increased significantly over time from 4543 Acres in 1995 to 106871 Acres in 2020 ($p < 0.001$; $R^2 = 0.96$; Table A3.2.5; Figure 3.2.2A). The ARIMA model (2;2;3; AIC 475.59; Supplements Table 1) suggests an AUR reduction of 30.1%, to 74808 Acres by 2030 (95% CI: -35827|185444 Acres). Overall, YAC is predicted to stay consistent for Wetland banks by 2030 with UCA further increasing and AUR decreasing, reducing available reserves by 2030.

Stream Banks

YAC for Stream banks, measured in Linear Feet varied greatly over the past 26 years from 2942 Linear Feet in 2004 to 2315912 Linear Feet added in a single year in 2012 (mean: 886495 ± 1139603 Linear Feet; Figure 3.2.2B). Stream banks were not listed before 2001. Overall, YAC increased significantly up to 2020 ($p < 0.05$; $R^2 = 0.27$; Table A3.2.6; Figure 3.2.2B). Future predictions for YAC for Stream banks (ARIMA 0;1;1; AIC 757.83; Table A3.2.2) suggest an increase to 953537 Linear Feet (95% CI: -2082203|3989277 Linear Feet) in yearly established new Stream bank area (7.6% increase from 1995 to 2020 yearly mean). UCA for Stream banks increased significantly over time from 1167 Linear Feet in 2001 to 6648679 on 2020 ($p < 0.001$; $R^2 = 0.77$; Table A3.2.6; Figure 3.2.2B). Predictions for 2030 (ARIMA 1;2;3; AIC 724.74; Table A3.2.2) show a continued increase by 38.6 % to 9212782 Linear Feet (95% CI:

2226190|16199373 Linear Feet). AUR increased from 11561 Linear Feet in 2001 to 1324203 in 2020 ($p < 0.001$; $R^2 = 0.83$; Table A3.2.6). AUR is predicted (ARIMA 2;2;3; AIC 668.47; Table A3.2.2) to increase from its 2020 level by 42.1% to 1881987 Linear Feet (95% CI: 701462|3062512 Linear Feet). Overall, YAC, UCA, and AUR for Stream banks have been increasing over the past 26 years and are predicted to follow that trajectory for the next 10 years.

Multi-Ecosystem Banks

Multi-Ecosystem banks increased in YAC from 1995 to 2020, ranging from 481 Acres established in 1996 to 14535 Acres in 2011 (mean: 5076 ± 4204 Acres; Figure 3.2.2C). While the increase over the past 26 years was significant ($p < 0.05$; $R^2 = 0.22$; Table A3.2.7), YAC for Multi-Ecosystem banks is predicted to decrease by 32.7% by 2030 to 3417 Acres (95% CI: -12220|19055 Acres) of yearly added Multi-Ecosystem bank area (ARIMA 1;1;0; AIC 486.35; Table A3.2.3). UCA for Multi-Ecosystem banks increased significantly over the past 26 years from an initial 224 Acres to 48870 Acres in 2020 ($p < 0.001$; $R^2 = 0.84$; Table 3.2.7; Figure 2C). The current (2020) UCA is predicted to almost double by 2030 (+83.3%; 89573 Acres; 95% CI: 47497|131650 Acres; ARIMA 0;2;1; AIC 442.49; Table A3.2.3). AUR increased from 123 Acres to 10653 Acres in 2020 ($p < 0.001$; $R^2 = 0.74$; Table A3.2.7; Figure 2C). Future predictions for the next 10 years (ARIMA 1;2;3; AIC 431.18; Table A3.2.3) show a steep decrease in AUR to the point of reaching 0 by 2026 (95% CI: -19241|19416 Acres). While increases are predicted for newly established Multi-Ecosystem areas in the future, much of that area is predicted to contribute to UCA while available reserves in AUR are predicted to decline to the point of depletion.

Species Banks

Species banks (conservation banks) showed a consistent YAC between 1995 and 2020 (mean: 4060 ± 22301 Acres; Figure 3.2.2C; $p = 0.264$; $R^2 = 0.012$; Table A3.2.8) except for 2014 (>100000 Acres established). Results from the ARIMA model for YAC for Species banks (0;1;0; AIC 585.54; Table A3.2.4) show that YAC for Species banks is predicted to decrease to 1788 Acres (95% CI: -173229|176806 Acres) in 2030 which marks a decrease of 44% from the previous yearly mean (Figure 3.2.2D). UCA for Species banks increased from around 1000 Acres in 1995 to 37686 Acres in 2020 ($p < 0.001$; $R^2 = 0.51$; Table A3.2.8; Figure 3.2.2D). UCA is predicted to decrease over the next 10 years with approaching 0 by 2028 (-1696 Acres; 95%

CI: -509714|506322 Acres; ARIMA 0;2;0; AIC 540.81; Table A3.2.4). AUR similarly to UCA increased over the past 26 years from 622 Acres to 50324 Acres as of December 31st, 2020 ($p < 0.001$; $R^2 = 0.62$; Table A3.2.8; Figure 3.2.2D). Compared to the predicted decrease in UCA, AUR for Species banks is predicted to increase up to 116550 Acres (95% CI: 52382|180717 Acres) in 2030 (ARIMA 1;3;3; AIC 474.17; Table A3.2.4). This predicted area constitutes a 131.5% increase from the current AUR. Species banks are predicted to slightly decrease in their yearly added capacity while potentially moving large areas from unreleased to released, reducing UCA while increasing AUR.

Current reserves and capacities across bank types and regions

Out of the total area that has been added through Wetland bank (mitigation banks) establishment (assessed n = 897) between 1995 and 2020, 43% were withdrawn as part of proponent transactions and are no longer available. 26% of the proportionate total bank area is currently available to be bought as credits and 31% may become available in the future depending on release schedules and performance criteria (Figure 3.2.3A). 65% of the total Stream bank (mitigation banks) area (assessed n = 253) from 1995 to 2020 has been withdrawn so far with 29% awaiting future release and a current reserve of 6% of total Stream bank area, recorded in Linear Feet (Figure 3.2.3B). Assessed Multi-Ecosystem banks (mitigation banks; n = 385) between 1995 and 2020 had more than half of their total established area (55%) withdrawn for 55%. 37% of the total area is currently unreleased and 8% are available for proponent credit transactions to be used for offsetting approved negative development impacts (Figure 3c). Finally, Species banks (conservation banks) have a current reserve of 23% of their total established area, 18% currently unreleased, an overall 59% withdrawn in transactions between 1995 and 2020 (Figure 3.2.3D). The highest current reserves (AUR) across bank types were attributed to Wetland and Species banks, while Wetland and Multi-Ecosystem banks have the currently highest proportion of yet unreleased area (UCA). Species and Stream banks had the most proportionate area withdrawn between 1995 and 2020.

Mapping the top 100 banks contributing to UCA and AUR showed that the majority were Wetland banks (n = 56) followed by Species banks (n = 19), Multi-Ecosystem banks (n = 14), and Stream banks (n = 11; Figure 3.2.4). These 56 Wetland banks comprising 6% of all assessed Wetland banks are currently holding 58% of all established and assessed Wetland bank area between 1995 and 2020 (~243000 Acres) and 51% of total AUR (~54000 Acres), as well as 69% of UCA (~89000 Acres). The main distribution for these Wetland banks was in the Southeast of Texas; Southeastern Louisiana; Mississippi, Georgia, and Florida (Figure 3.2.4A). The 19 Species banks (19% of all assessed Species banks) cover an area of 185000 Acres (86% of total Species bank area), as well as 86% of all AUR (~43000 Acres) and 92% of UCA (~35000 Acres). Most of these banks were in central Texas, Oklahoma, Southern Florida, and California with single banks in Wyoming, Maine, and Kansas (Figure 3.2.4A). The total established area for the 14 assessed Multi-Ecosystem banks (3.5% of all Multi-Ecosystem banks) comprises 27% of the total established area between 1995 and 2020, 39% of total AUR (~4000 Acres), and 35% of total UCA (~17000 Acres). These 14 banks were in Northeast Texas, Mississippi, Florida, and Georgia (Figure 3.2.4A). Finally, the 11 Stream banks (4% of all Stream banks) located in Mississippi, West Virginia, North Carolina, and South Carolina held 63% of the total Stream bank area (14130000 Linear Feet), 58% of total AUR (756000 Linear Feet), and 68% of total UCA (4551000 Linear Feet). These 100 banks contributing the most to AUR and UCA overlap with the general banking distribution in the United States, identifying especially the Southeast, parts of the Midwest, and the West-Coast as banking hotspots in terms of density (Figure 3.2.4B).

3.2.4 Discussion

We identified several key findings related to our hypotheses that should be summarized here:

Yearly added Capacity (YAC) has been consistent for Wetland and Species banks and overall increased for Multi-Ecosystem banks and Stream banks. Unreleased Capacity (UCA) has accumulated between 1995 and 2020 for all four bank types. Future predictions suggest a similar trend for 2030 for Mitigation banks (Wetland, Stream, and Multi-Ecosystem banks) while predicting a decrease in UCA for Conservation (Species) banks. Available Unwithdrawn Reserves (AUR) increased over time and are predicted to decrease greatly for Wetland and Multi-ecosystem banks by 2030, while predicted to increase for Species and Stream banks. As of December 31^{st,} 2020, Wetland banks and Species banks had the largest proportionate amount of AUR compared to the overall available banking area for each respective type. Stream and Multi-ecosystem banks showed low percentages of total bank area being available in reserves. Bank area with possible future availability (UCA) was the highest for Wetland and Multi-ecosystem banks and the lowest for Species banks. Banks contributing the most to AUR and UCA were mostly Wetland banks. The largest Wetland and Multi-Ecosystem banks AUR and UCA are currently sitting in the Southeastern United States. Species bank AUR and UCA were predominantly associated with the Western United States, namely Wyoming, California, and Texas. Areas for large Stream bank AUR and UCA were in the South and along the Eastern Seaboard. Overall AUR and UCA for all four bank types are linked to a few individual banks and specific states and regions compared to the overall number of 1636 assessed banks

Wetland banks and Multi-Ecosystem banks – high capacities but decreasing reserves?

Wetland and Multi-ecosystem banks showed similar past and predicted future trends concerning UCA and AUR as well as spatial distribution for reserve and capacity hotspots. Both bank types showed constant and/ or increasing yearly added capacity (YAC), with both bank types having large increases of UCA over time (currently ~30% of the total established area; Figure 3 A; C), which makes a *supply bottleneck* unlikely. However, AUR is predicted to decline sharply by 2030, which also speaks against a *demand bottleneck*. The scenario of declining reserves could potentially come true since newly established banks are not able to release area in the form of credits fast enough due to release schedules or not meeting performance criteria, with accumulating unreleased area outpacing the available area that constitutes the current reserves as shown in the results. Both bank types have already passed the turning point in the past five years, with AUR declining. In the case of Wetland and Multi-Ecosystem banking, this would mean a *release schedule bottleneck.* This issue is supported by other studies and the general literature, pointing out that release schedules and bank operation can often change over time and range from switches in bank sponsor to changes in area allocation all the way to bank failure and potential closure in the future (e.g., Gardner et al. 2013; Reiss et al. 2009; Vaissière et al. 2017). For instance, a study from Florida, where large proportions of our studies' Wetland bank UCA and AUR were located, found that while mitigation bank compliance was over 40%, 17% of the assessed bans were unlikely to meet permit criteria. Furthermore, credit release was often not or insufficiently tied to ecological criteria but rather financial or operational benchmarks (Reiss et al. 2009). This high level of uncertainty for ecological functionality in combination with noncompliance and delays in credit release could explain current declining AUR trends for Wetland and Multi-Ecosystem banks while accumulating UCA.

Another aspect to factor in is ownership. An estimated 75% of all wetlands in the continental United States are privately owned (Scodari et al. 1995). Turning private land owned by a multitude of smaller stakeholders into banks or acquiring larger portions for umbrella banks could delay the operating process further, explaining longer startup time for Wetland and Multi-Ecosystem banks (Bunn et al. 2013; Grimm 2020). Finally, there is regionality. The highest reserves and unreleased capacities for Wetland and Multi-Ecosystem banks were in the Southeast, especially Florida, Georgia, Mississippi, and Southeast Texas. While the demand for protecting and mitigating impacts to ecologically valuable and vital wetlands in the Southeast is high, it is somewhat worrisome that capacities and reserves seem to be almost exclusively limited to this region. UCA and AUR in these regions make up around 50% of the total UCA and AUR of all assessed Wetland and Multi-Ecosystem banks. Another factor in the case of the large amount of UCA in the Southeast is most likely due to future anticipated demand in mitigation credits. Final rules from USACE and (US)EPA in 2008 state a preference for mitigations banking as opposed to on lieu fees or proponent led offsetting, signaling both developers as well as bankers the need to secure more area for banks (USACE 2008; Pittman & Waite 2009; Vaissière et al. 2017). Long-term anticipation and regulatory favoritism hold a potential danger for these banks and areas since regulations and frameworks are constantly changing and so are market dynamics and development needs. In a worst-case scenario, triggered by a switch from banking towards alternative measures, as well as decreasing prices, these large wetland areas could simply stay unrestored and unmanaged. Vice versa a banking boom would also reduce investment and advancement of alternative offsetting tools which could also increase competition amongst banks, wanting to release their anticipated credits, leading to bank failure and closures (Pittman & Waite 2009; Robertson 2004; Vaissière et al. 2017).

Our predictions show that in the case of Wetland banks, large accrued past reserves should be sufficient in compensating for release delays or future anticipated impacts. While past and current demand for Wetland banking credits is high, large-scale establishment especially in the Southeast is likely equally seen as a preservation measure for future anticipated impacts (e.g., Reiss et al. 2014; Spieles 2005). Multi-Ecosystem banks however in a worst-case scenario, given past trends and data, would be depleted by 2030 due to lower reserves and the increased demand for more complex and diverse ecosystem services provided through banks (e.g., Dadisman 2020; Deal et al. 2012).

Stream banks – Stream rehabilitation in the United States

Stream mitigation banks showed a different trend from Wetland and Multi-Ecosystem banks in terms of predicted increases in accumulated reserves coinciding with increases in unreleased capacity in the future. This predicted increase in reserves in Linear Feet based on past data could be linked back to stream banking increasing more recently (the early to mid-2000s) in popularity as opposed to Wetland mitigation banking as well as operational differences (Julian & Weaver 2019; Lave et al. 2008). Stream mitigation banking heavily relies on habitat rehabilitation and is often aimed at degraded urban streams are agriculture adjacent (e.g., Lave 2021; Theis et al. unpublished). These often-small streams are aimed to be restored through management through a banking agreement. Small scale and tangible goals through restoration could foster faster release schedules and realization of stream mitigation banks compared to some of the larger Wetland mitigation banks or approaches including ecosystem establishment, struggling from well-known and still persistent issues with long-term ecological processes or over-simplification of wetland complexity and bank failure (e.g., Mateos 2018; Whigham 1999).

Large scale stream mitigation banks on the other hand are often part of overarching land-use planning strategies on a watershed level (e.g., BenDor & Riggsbee 2011; Chastant 2007; Glickauf & Keebaugh 2009; Harding 2001). Banks being part of land-use planning strategies is still new but more and more common for Stream banks and holds the unique advantage of considering surrounding development and residential impacts, better financial and operational means, and overall better connectivity to the rest of the watershed and its ecological processes (e.g., Chastant 2007; Glickauf & Keebaugh 2009; Harding 2001). Overall current and predicted increases in demand for stream mitigation banking area and credits paired with a faster turn-around time and credit release potentially based on favoring rehabilitation over ecosystem establishment could explain the predicted scenario for 2030. Current reserves for stream banks only made up 6% of the total banked area, suggesting that given its relative novelty, stream migration banking is not yet as mature compared to for instance wetland mitigation banking, providing another explanation for the future predictions with stream mitigation banking eventually reaching a point where reserve usage will be outpaced by the accumulation of unreleased area and credits.

Using rehabilitation over ecosystem establishment and its issue of potentially missing ecological equivalency are not considered in this scenario (e.g., Fox & Nino-Murcia 2005; Grimm 2021; Vaissière et al. 2017). Stream mitigation banking's popularity in the Southeast aligns with findings from current literature (e.g., Chastant 2007; Glickauf & Keebaugh 2009; Harding 2001; Lave 2021; Lave et al. 2008). Overall capacity, UCA, and AUR held by a few individual Stream banks point to similar potential bottlenecks compared to Wetland and Multi-Ecosystem banking, mainly being a regional bottleneck with chances for a *release schedule bottleneck* in the future should Stream banking follow a similar trajectory as other mitigation banking practice. A *regional bottleneck* could be prevented by extending Stream banks to other major watersheds (especially predicted to experience future water stress) and building on existing and new land-use planning strategies (e.g., potential applications in Colorado and the West; BenDor & Riggsbee 2011; Julian & Weaver 2019).

Species banks – Regulatory drivers and market drivers in unison

Conservation banks, targeting single and multiple species deviate from the other trends predicting a future decline in UCA and an increase in AUR. These predictions are since conservation banks have different operational and establishment drivers compared to mitigation banks. The main driver here is the listing of imperiled species or the need to conserve species habitat prone to current and future development by the Fish and Wildlife Service (FWS 2003; Poudel et al. 2019). An example here is the greater sage-grouse (*Centrocercus urophasianus).* Though not officially listed under the endangered species act, due to a wide range of legislative issues and competing stakeholder interests (e.g., listing of large range species would impact many different economic branches), scientific evidence of large-scale habitat degradation and consequent long-term species endangered has led to the establishment of the United States' largest single-species conservation Bank in Wyoming in 2015 (e.g., FWS 2015; Holloran et al. 2010; LeBeau et al. 2018). Given the large range, the sage-grouse needs to maintain healthy populations, banked area, and land held by the Sweetwater River Conservancy encompass over 700,000 acres with plans to establish similar banks in other neighboring states (e.g., FWS 2013). Other examples are the Florida panther or vernal pool crustaceans in California (e.g., Bunn et al. 2014; Pienaar & Kreye 2015; Poudel et al. 2019).

While Stream banks often aim at restoring degraded stream habitat and Wetland mitigation banking utilizing a wide array from restoration to the establishment, conservation banking mainly relies on habitat preservation in combination with land management and maintenance (e.g., Fox & Nino-Murcia 2005). The example of the sage-grouse showcases that preserving large areas of the current habitat through a bank does not match currently approved impacts but is in anticipation of future impacts. Thus, release schedules will provide more and more area to be available in credits over time before the demand is there. This explains the potential large current and future reserves of species credits and decline in unreleased capacities. While a wide variety of endangered species are covered by conservation banks, a large proportion is yet to be included in conservation banking with issues concerning migratory and large-range species persisting and slowly being addressed by Umbrella banks or cross-boundary agreements (e.g., for migratory bird and bat species; Kark et al. 2015). Our findings suggest that species banks do not adhere to any of the four bottlenecks due to their different operational approach and legislative framework. Imperiled species with a limited range and likely to be affected by climate and land-use change and ongoing urbanization could potentially lead to a rapid increase in newly established conservation banks on private land, considering that large proportions of endangered are indeed on private land (Clancy et al. 2020; Poudel et al. 2019). Risk reduction for investors and incentivization will be even more important in the future to target private land since species listings outpace bank establishment and funding provided through public and government-owned agencies. (e.g., Clancy et al. 2020; Kerkvliet 2021; Stein et al. 2008).

General considerations

Aside from YAC, UCA, and AUR, there are several key aspects that should be considered for future banking practices. Future environmental and climate change stress will impact banks greatly both in terms of performance criteria as well as areas where mitigation might be needed. Plans to mandate climate change mitigation are ongoing and are likely to be part of future policies and banking requirements (e.g., Baer 2015; Latimer et al. 2007; WRI 2022). With an increased demand for all four bank types, perceived and actual completion could increase while establishment and initiation costs are already high, increasing risk and uncertainty for current and prospective bank owners. Future banking practices and guidance needs to focus on assuring that conservation priorities align with feasibility and revenue expectations of owners while increasing transparency on said costs and reported data, which is still a persisting issue (e.g., Clancy et al. 2020; Kerkvliet 2021; Poudel et al. 2019; Stein et al. 2008). RIBITS as a centralized database needs to be improved in terms of provided data as well as data clarity. Currently, only a certain proportion of banks has full reports associated with them, costs and investments are in most cases not accessible and area to credit or ecosystem service to credit conversions as well as initial impact type and extent are hard to trace which make difficult to determine if ecological equivalency was achieved. While RIBITS is an excellent data repository for broad banking characteristics, the more important in detail data that would warrant in detail guidance and potential policy changes is largely still unavailable.

3.2.5 Conclusions

Banking frameworks designed as offsetting mechanism alternatives, have become increasingly popular over the past 26 years, and will continue to do so according to the data as well as different organizations and countries aiming to establish banking as a widespread global mitigation mechanism (Santos et al. 2015). Land, usable for banks will potentially decrease in the future due to prime areas already being used as banks in combination with further land development in the conterminous United States (Fox & Nino-Murcia 2005). Findings from our study conclude that based on past trends for supply and demand, mitigation banks targeting wetlands and multiple ecosystems could experience a *release schedule bottleneck* given that demand seems to outpace credit release, while there are no indications for *demand* or *supply bottlenecks* based on newly established banks and credit withdrawal. Advance credit release through mitigation fee programs could help address this issue as well as expand the network to avoid *regional bottlenecks*, especially given future climate stress predictions (WRI 2022). Stream mitigation banking is predicted to meet current and future demand, with the main driver potentially being faster turnaround times due to large use of rehabilitation efforts, land-use planning, and stream mitigation banking being a younger practice compared to mitigation banking, becoming more popular in the mid-2000s. Future trends could be like wetlands mitigation and multi-ecosystem banking considering this time lag (BenDor & Riggsbee 2011). Current mitigation banking capacities and reserves are focused heavily on the southeastern United States. Future developments likely will see a shift in regionality to other areas which are not, yet part of the banking network given predicted climate and land-use changes (Powers & Jetz 2019). Conservation banks, targeting single and multiple species, do not seem to be experiencing bottlenecks in the same manner as mitigation banks, due to their demand mostly being driven by species listings and delisting's under the ESA. Future practice for conservation banks will be faced with issues of aligning financial feasibility for owners with ESA and FWS conservation goals while implementing an increased number of transboundary agreements to cover a wider variety of endangered species (Bunn et al. 2014; Pienaar & Kreye 2015; Poudel et al. 2019).

Chapter 3: Figures and Tables

Chapter 3.1 Figures

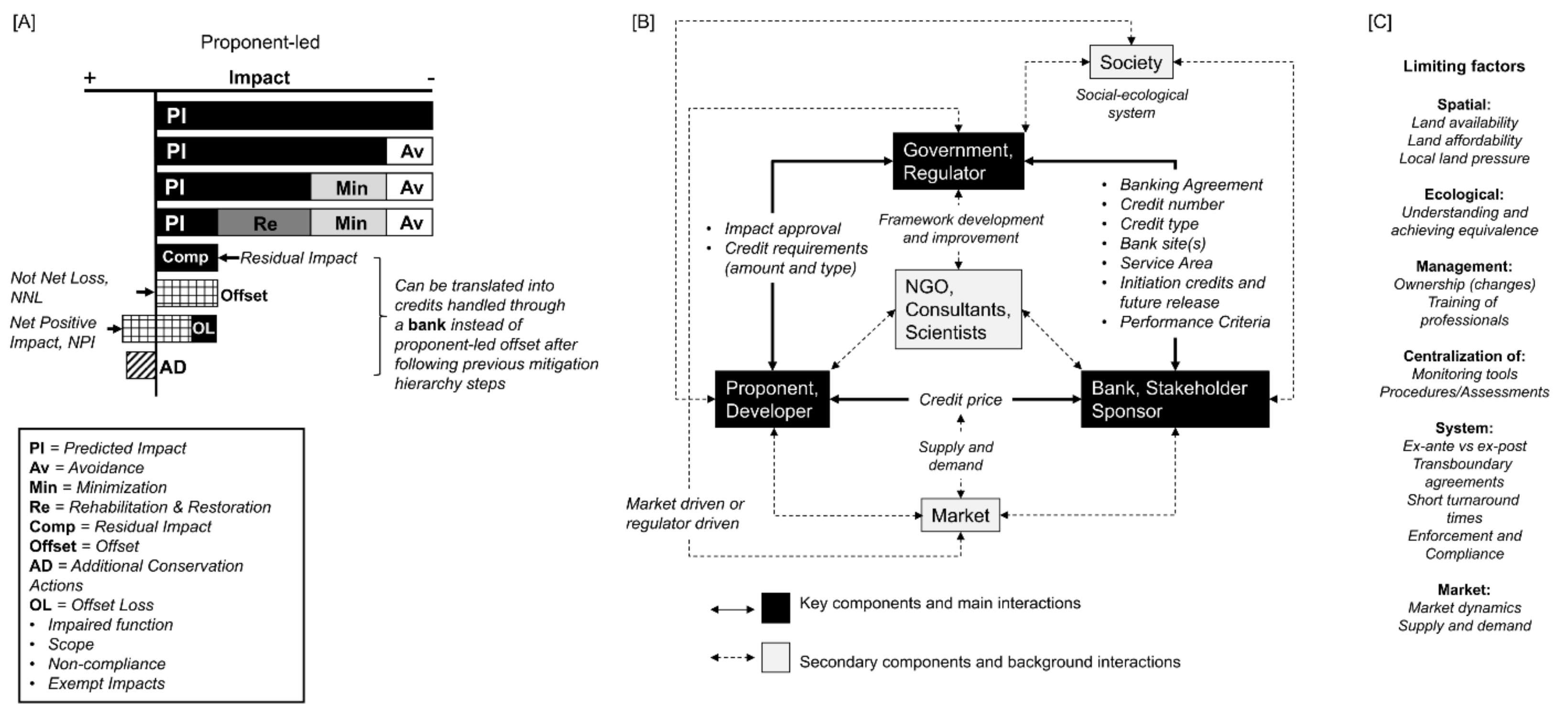


Figure 3.1.1. *Mitigation hierarchy outline following the required steps (Avoidance, Minimization, Restoration & Rehabilitation) before allowing for harmful impact to be compensated for through an equivalent or larger offset and banking principle and role of banking credits in the traditional offsetting scheme (a). Interaction between proponent, regulatory agency, and bank to approve development projects, mitigate, and compensate negative environmental effects through translating losses into credit amounts (b) and associated limitations of banking practices according to literature (c) (based on Bull et al. 2013; Burgin 2008; McKenney 2005; USACE 2008).*

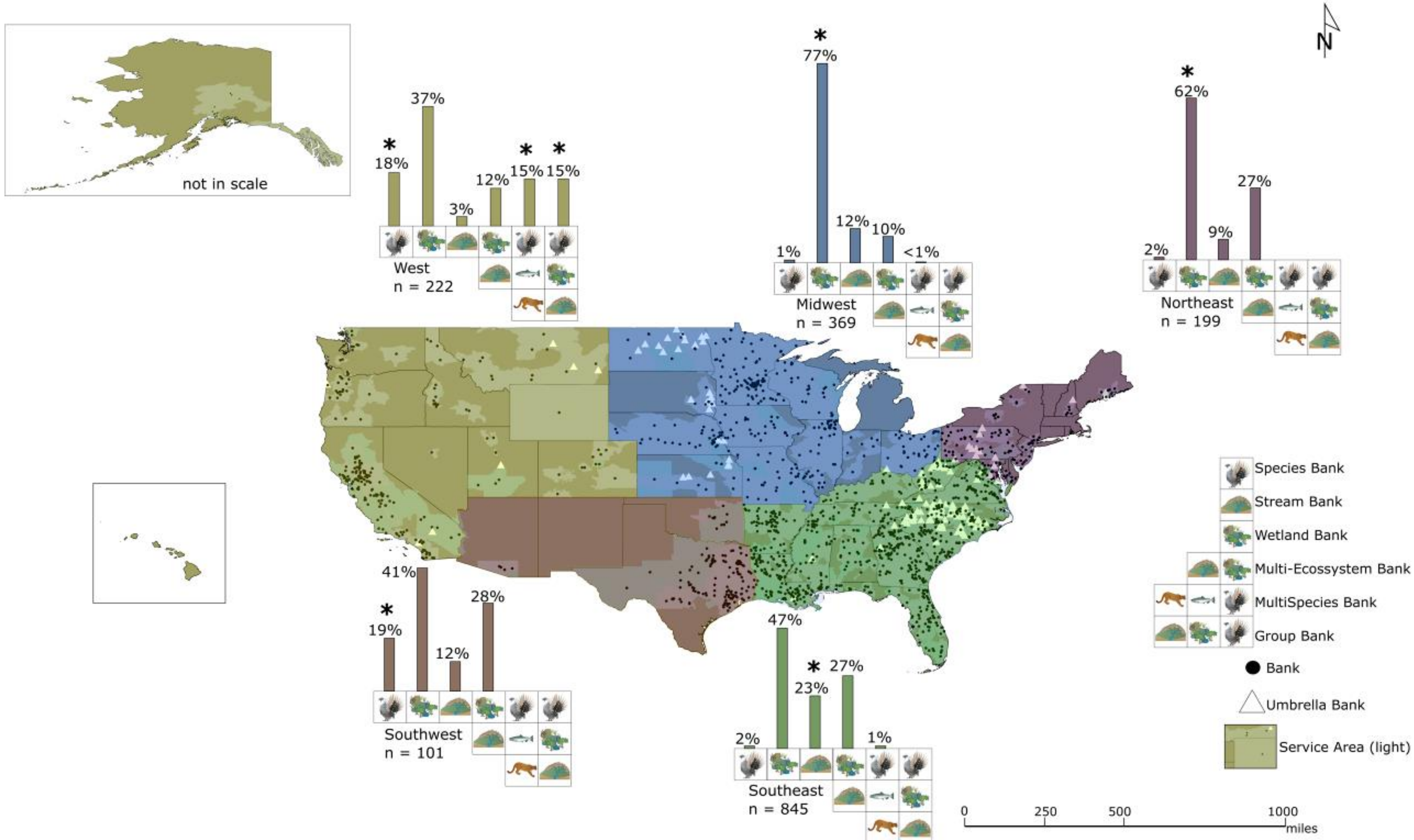


Figure 3.1.2. *Bank Type (Species; Wetland; Stream; Multi-species; Multi-ecosystem; Group; n = 1736; as of 2020) distribution across regions, * shows a significant positive correlation of Bank type and Region (Northeast; Southeast; Midwest; Southwest; West) based on Chi-squared test (Appendix S1). Light areas indicate service areas (areas in which banks can legally sell credits) and are divided into conservation and mitigation banks (banks; black dots) and Umbrella banks (white triangles). (Digital symbols attribution Jason C. Fisher; Tracey Saxby; Emily Nastase; Jane Hawkey; ian.umces.edu/media-library*

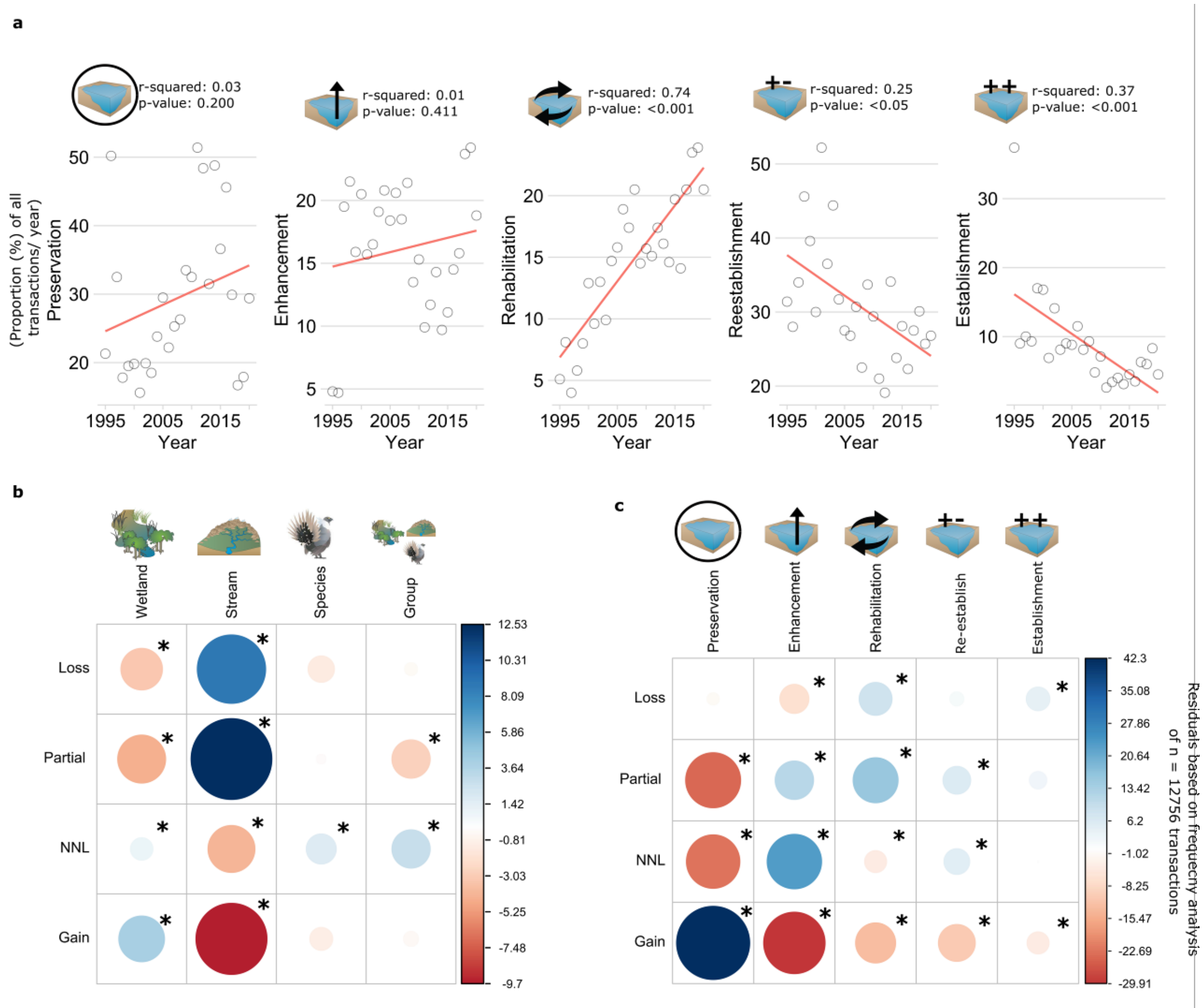


Figure 3.1.3. *Used Mitigation-Methods (a) in the extracted transactions (n = 12756; Enhancement; Establishment; Preservation; Re-establishment; Rehabilitation) in the percentage of total transactions each year ranging from 1995 to 2020 with linear model output, p-values, and r-squared values indicating significant changes in method utilization over time (Appendix S2). Chi-squared test of independence (b) for Mitigation-Target (wetland; stream; species; group) and Mitigation-Method (Enhancement; Establishment; Preservation; Re-establishment; Rehabilitation) (c) in relation to meeting NNL under a Gain:Loss ratio of 1:1 for each transaction (n = 12756; NNL categories = Loss; Partial, NNL; Gain). Positive residuals indicate a positive correlation between two categories and circle size indicates residual magnitude. Post-hoc analysis for Chi-squared test based on standardized residuals with Bonferroni correction indicating significance of positive or negative correlations by * (Appendix S3; S4). (Digital symbols attribution Jason C. Fisher; Tracey Saxby; Emily Nastase; Jane Hawkey; ian.umces.edu/media-library).*

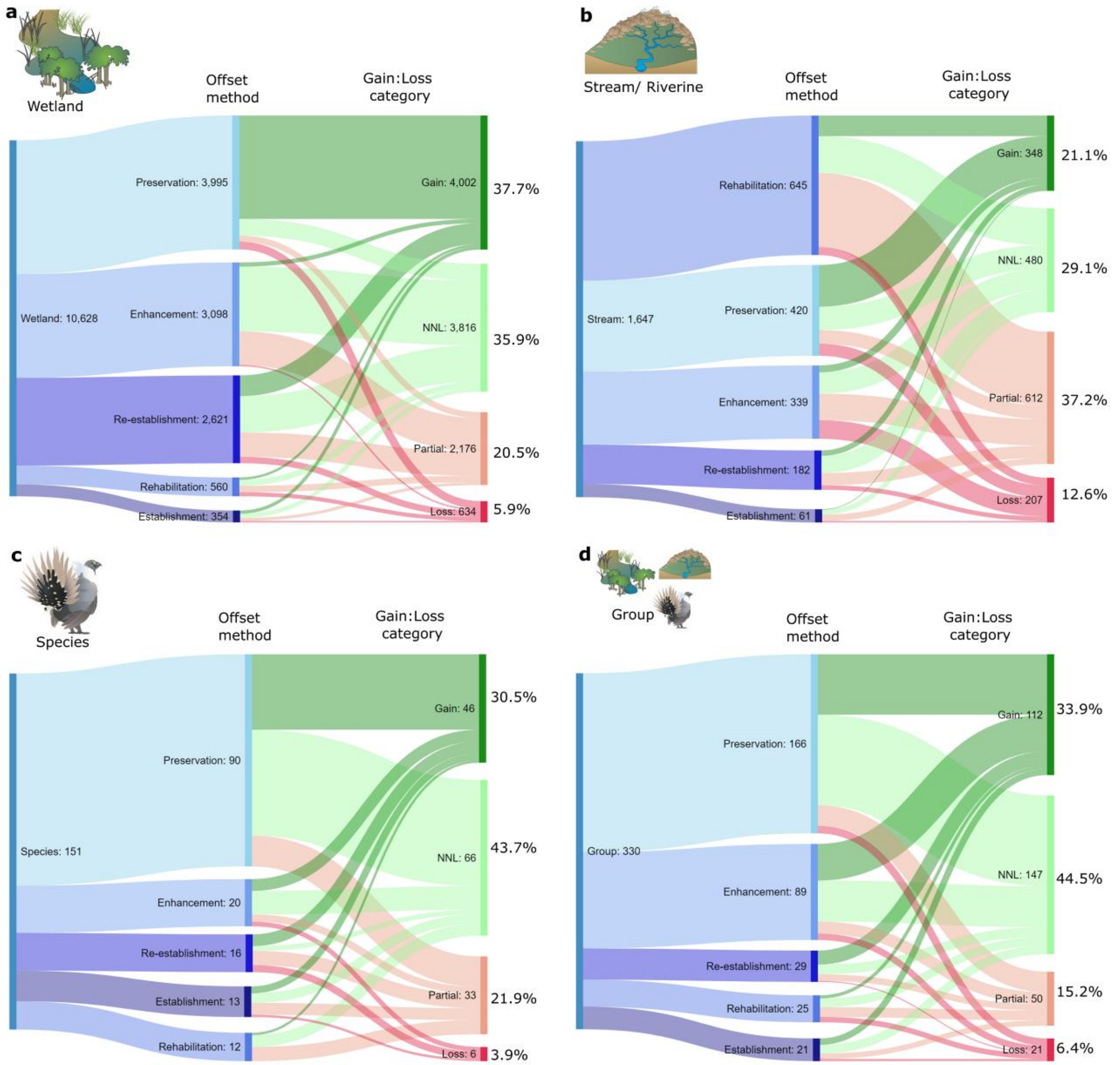


Figure 3.1.4. *Sankey diagram showing distribution of extracted transactions (n = 12756) across transaction targets type (wetland; stream; species; group), Mitigation-Method (Enhancement; Establishment; Preservation; Re-establishment; Rehabilitation) and transactions leading to one of four NNL/ Ratio Equivalency outcomes based on Gain:Loss ratios (Loss: <0.2:1; Partial: 0.2 to 0.9:1; NNL: >0.9 to 1.25:1; Gain: >1.25:1). Green flows indicate meeting or exceeding NNL Gain:Loss ratios while red flows indicate not meeting Ratio Equivalency requirements. (Digital symbols attribution Jason C. Fisher; Tracey Saxby; Emily Nastase; Jane Hawkey; ian.umces.edu/media-library).*

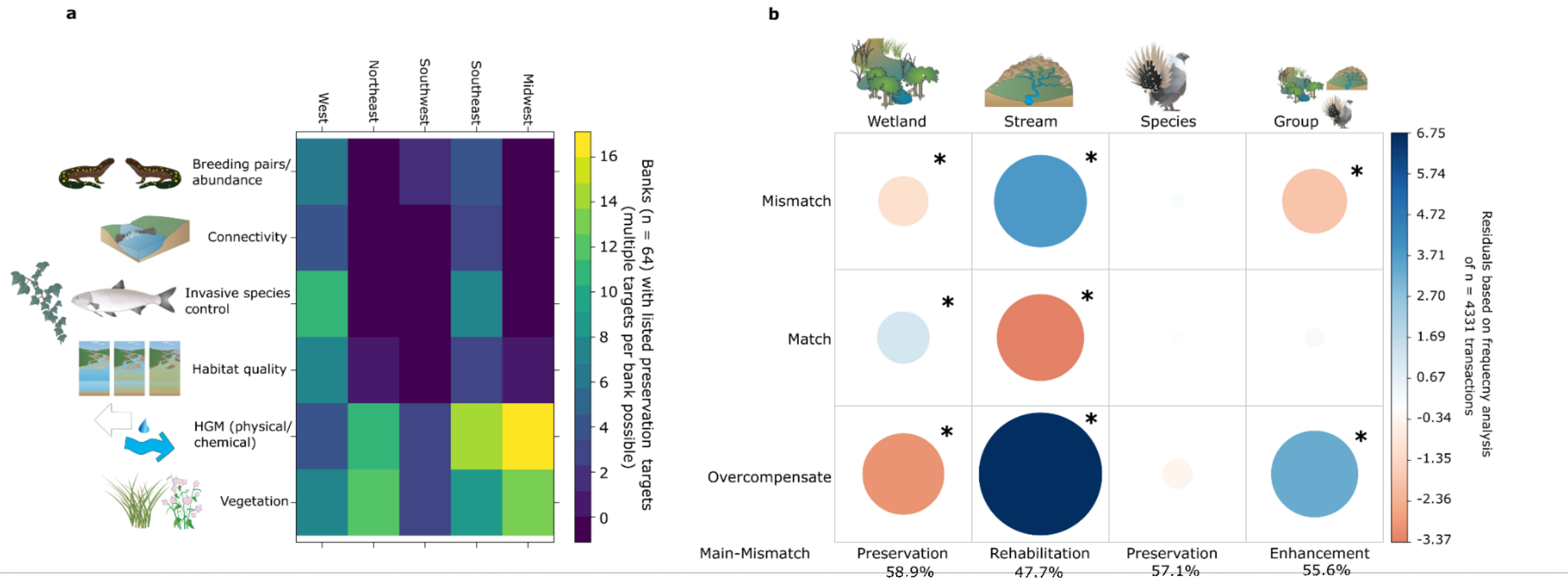


Figure 3.1.5. *Heat map of 64 analyzed banks (a) utilizing Preservation and their main associated targets (Breeding pairs/ species abundance; Habitat connectivity; Invasive species control; Habitat quality; Hydrogeomorphology; Vegetation) across the five main geographic regions (Northeast; Southeast; Midwest; Southwest; West). (b) Chi-squared test of independence for frequency table analyses. Categorical variables are composed out of Mitigation-Target (wetland; stream; species; group) and Impact-Type to Offset-Type category (Match; Mismatch; Overcompensate. Positive residuals indicate a positive correlation between two categories and magnitude. Post-hoc analysis for significant correlations for Chi-squared test based on standardized residuals with Bonferroni correction of positive or negative correlations indicated by * (Appendix S5). (Digital symbols attribution Jason C. Fisher; Tracey Saxby; Emily Nastase; Jane Hawkey; ian.umces.edu/media-library).*

Chapter 3.1 Tables

Table 3.1.1. *Overview of used variables, extracted data and definitions.*

Bank types:		Reference: RIBITS; (US)FWS; (US)EPA; USACE	
Mitigation	**Conservation**	**ILF (In-lieu fee program)**	**Umbrella**
A site where wetlands, streams, or riparian areas are established, rehabilitated, enhanced, or preserved to offset authorized by the Department of Army permits.	Permanently protected sites managed for endangered species, threatened species, or species at risk. The aim is to offset adverse impacts to the protected species occurring off-site. Permits managed by USFWS.	Rehabilitation, establishment, enhancement, and/or preservation of habitat area or ecosystem function through funds paid to a governmental or non-profit natural resources management organization. The operation and use of an in-lieu fee program are governed by an in-lieu fee program instrument thus differing from mitigation banks as well as allowing out of kind mitigation.	One banking instrument that dictates general requirements for an array of current and future sites (e.g., management and oversight of individual site plans to add future sites to the program).

Bank types according to bank targets:					
Wetland	**Stream**	**Species**	**Multi-Species**	**Multi-Ecosystem**	**Group**
Targeting wetlands	Targeting riverine systems	Targeting a specific species	Targeting multiple species	Targeting multiple ecosystems	Targeting ecosystems and species

Extracted main variables: Bank Number & Bank transactions linked to impacts	
Bank numbers	
n = 1736	Number of approved banks post 1995 excluding ILF sites divided into the 6 targets and assigned to one of 5 geographic regions (Northeast; Southeast; Midwest; West; Southwest)

Bank transactions linked to impacts	
	Approved withdrawn transactions in RIBITS for the 1736 banks linked to:
	a) Transaction target (Wetland; Species etc.);
n = 12756	b) Impact size, offset size in acres (linear feet for streams), and credit amount/ method;
	c) Mitigation method (Establishment; Preservation etc.);
	d) Date of the transaction;
n = 4331	e) Impact-Type to Offset type

a)	**Mitigation-Targets;** individual transactions listed in RIBITS and their designated target; *Mitigation-Targets differ compared to Bank targets that there is no distinction between Multi-species, Multi-ecosystem, and Group transactions but summed simply as Group transactions. Wetland (n = 10628) Stream (n = 1647) Species (n = 151) Group (n = 330)
b)	**Gain:Loss**; size in area (acres/ linear feet) of impacted area and area used to offset impact as well as credit yield per area and Mitigation-Method. Gain:Loss ratio (calculated by credit yield for a given Mitigation-Method (Gain) per acre compared to impact (Loss)) converted into categories responding to meeting NNL as defined as a minimum of 1:1 Gain:Loss ratio (acknowledging that many regulatory agencies require higher ratios). Categories account for inherent variability and inaccuracy of measuring offsets. **Loss** — Gain:Loss <0.25:1 **Partial** — Gain:Loss 0.25-0.9:1 **NNL** — Gain Loss >0.9-1.25:1 **Gain** — Gain:Loss >1.25:1 Example: Preservation of 10 acres yield 0.5 units credit per acre = gain of 5 and negative impact on 10 acres results in a Gain:Loss ratio of 0.5:1 corresponding to "Partial NNL". Gain:Loss ratios are used as an indicator for meeting NNL in terms of **ratio equivalency**.
c)	**Mitigation-Methods**; the method used to achieve said offset as defined in section 404 of the Clean Water Act.
n = 449	**Establishment;** means the manipulation of the physical, chemical, or biological characteristics present to develop a habitat area that did not previously exist. Establishment results in a gain in habitat area and ecosystem function.
n = 2848	**Re-establishment;** means the manipulation of the physical, chemical, or biological characteristics of a site with the goal of returning natural/historic functions to a former habitat area. Re-establishment results in rebuilding a former habitat area and results in a gain in habitat area and ecosystem functions.
n = 1242	**Rehabilitation;** means the manipulation of the physical, chemical, or biological characteristics of a site with the goal of repairing natural/historic functions to a degraded habitat area. Rehabilitation results in a gain in ecosystem function but does not result in a gain in habitat area.
n = 3564	**Enhancement;** means the manipulation of the physical, chemical, or biological characteristics of a habitat area to heighten, intensify, or improve a specific ecosystem function(s). Enhancement results in the gain of selected ecosystem function(s) but may also lead to a decline in other ecosystem function(s). Enhancement does not result in a gain in habitat area.
n = 4671	**Preservation;** means the removal of a threat to, or preventing the decline of, habitat area and ecosystem function by an action in or near those areas. This term includes activities commonly associated with the protection and maintenance of habitat area and ecosystem function through the implementation of appropriate legal and physical mechanisms. Preservation does not result in a gain of habitat area.
d)	**Recorded date** of each of the 12756 transactions (year); ranging from 1995 to December 31st, 2020.
e)	**Impact-Type to Offset-Type;** individual transactions listed in RIBITS their Impact-Type ('Ecosystem Function Loss'; 'Function and Habitat Area Loss') and Offset-Type

(Preservation; Enhancement; Rehabilitation = Ecosystem Function Gain; Establishment; Re-establishment = Ecosystem 'Function and Habitat Area Gain'). Impact-Type to Offset-Type is used as an indicator for meeting **ecological equivalency.**

Wetland (n = 3702) Stream (n = 501) Species (n = 37) Group (n = 91)

Chapter 3: Figures and Tables

Chapter 3.2 Figures

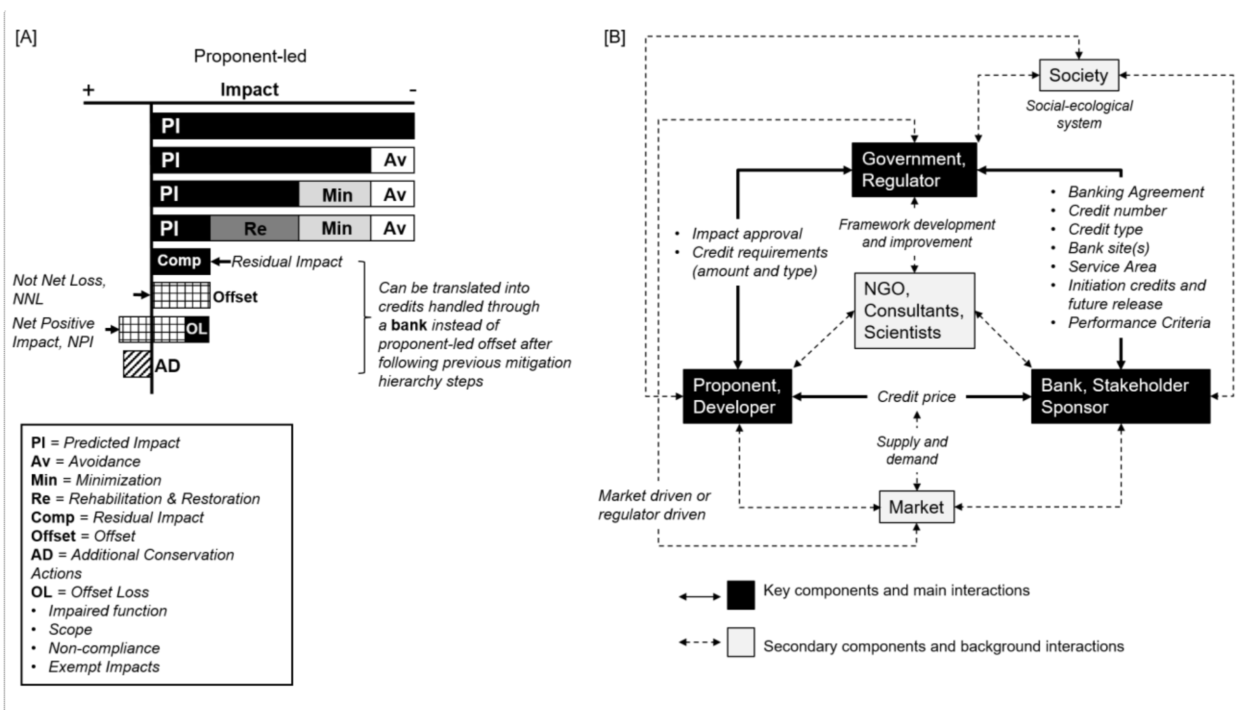
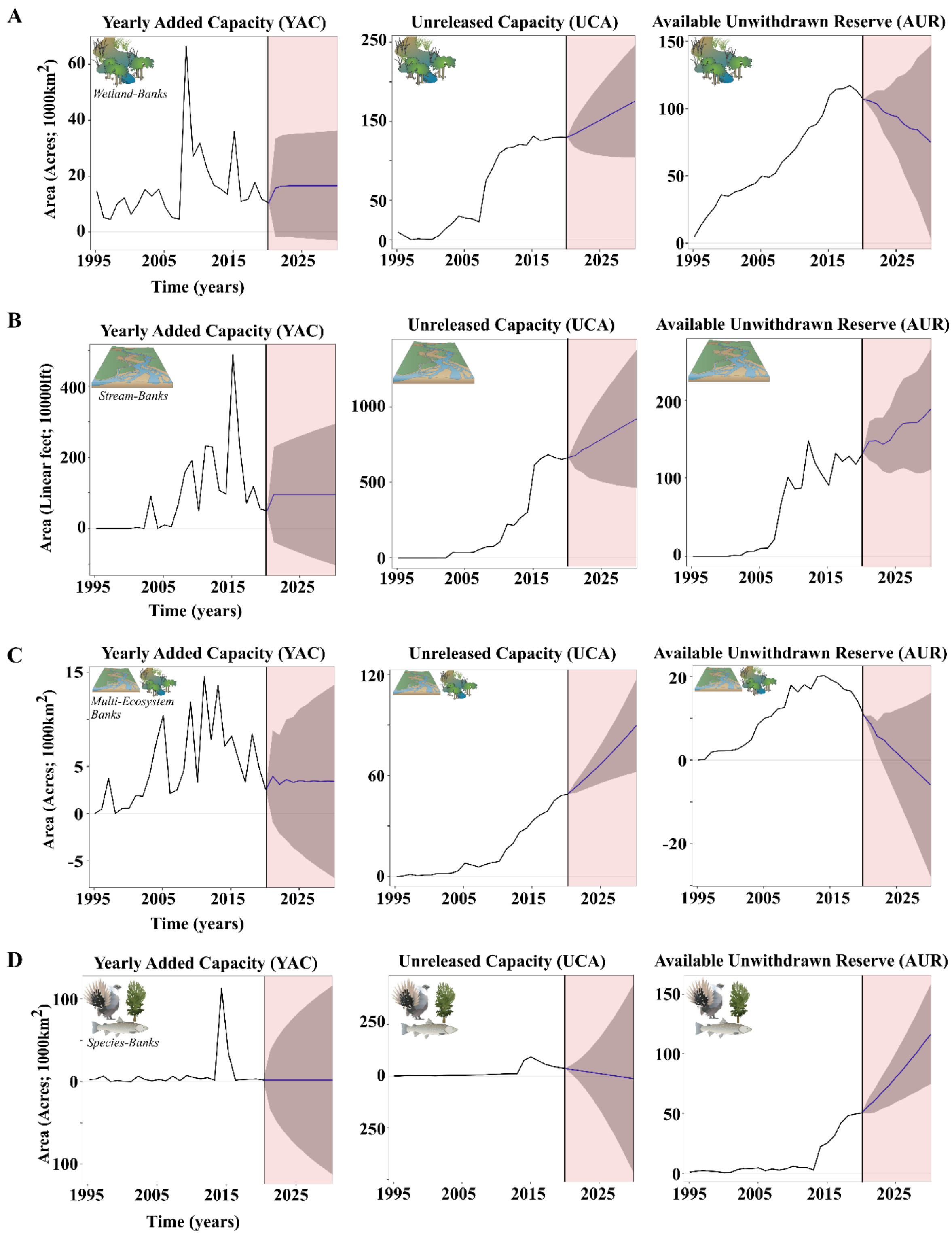


Figure 3.2.1. *Mitigation hierarchy outline, following the required steps (Avoidance, Minimization, Restoration & rehabilitation) before allowing for harmful impact to be compensated for through an equivalent tor larger offset (A). Banking principle and role of banking credits in the traditional offsetting scheme as well as interactions between proponent, regulatory agency, and bank (key components) to approve development projects, mitigate, and compensate negative environmental effects through translating losses into credit amounts and secondary or background components (e.g., NGOs, Market dynamics (based on McKenney & Kiesecker 2009; Vaissière, & Levrel 2015).*

Figure 3.2.2. *Plotted Yearly Added Capacity (YAC); Unreleased Capacity (UCA) and Available Unwithdrawn*

Reserve (AUR) for the four bank types (Wetland (A); Stream (B); Multi-Ecosystem (C); Species (D)) between 1995 and 2020 based on 1636 assessed banks. Future predictions up to 2030 for all bank types based on ARIMA models (Appendix Tables 1 – 4) and 95% Confidence Intervals (Appendix Table 1 – 4; Model selection walkthrough in Appendix 1). (Digital symbols attribution Tracey Saxby; ian.umces.edu/media-library).



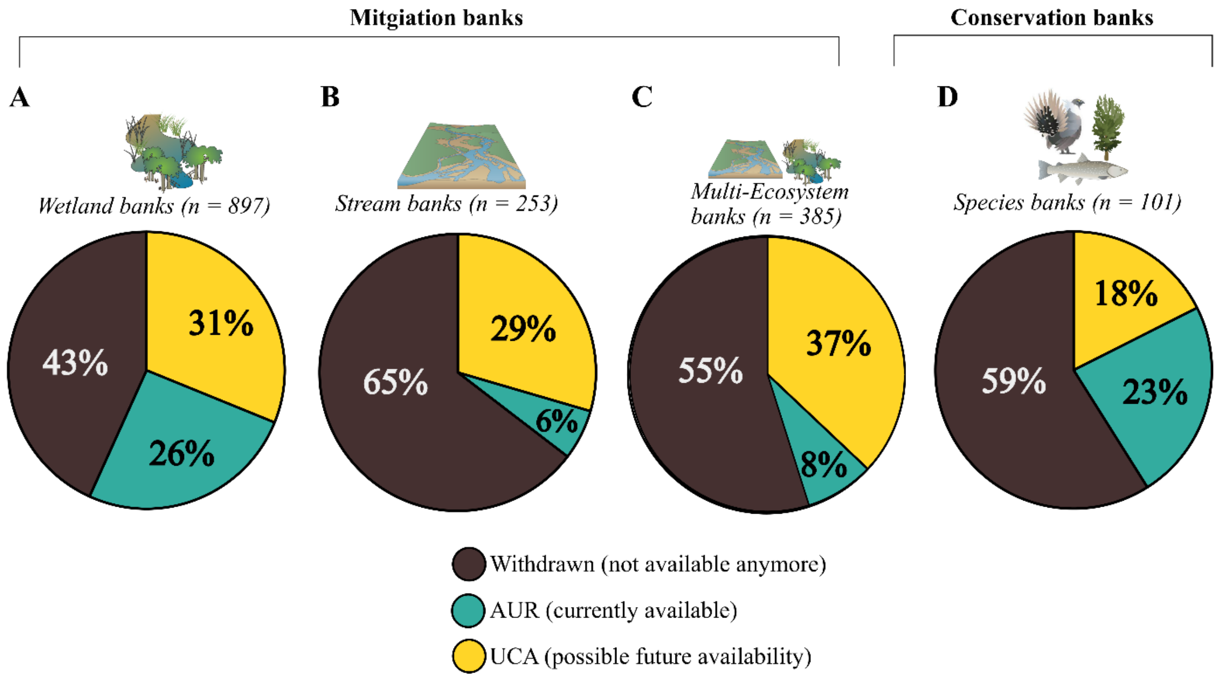


Figure 3.2.3. *Withdrawn area through credit transactions between proponent and bank in contrast to Unreleased Capacity (UCA) and Available Unwithdrawn Reserve (AUR) for the four bank types (Wetland (A); Stream (B); Multi-Ecosystem (C); Species (D)) based on totaled data from 1995 and 2020 based on 1636 assessed banks. (Digital symbols attribution Tracey Saxby; ian.umces.edu/media-library).*

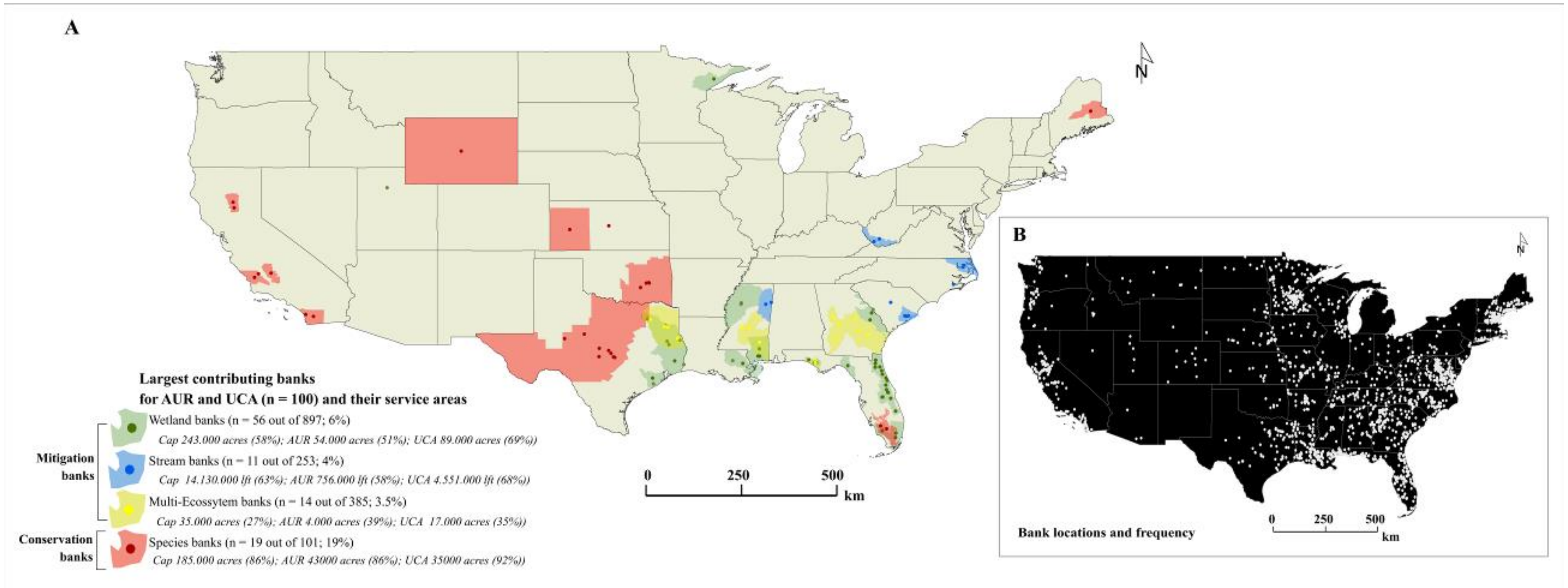


Figure 3.2.4. *(A) Mapped bank location and service area for the top 100 highest Unreleased Capacity (UCA) and Available Unwithdrawn Reserve (AUR) contributing banks across the four banks types (Wetland; Stream; Multi-Ecosystem; Species) based on totaled data from 1995 and 2020 based on 1636 assessed banks. Individual banks can have multiple locations as primary and secondary etc. bank areas. UCA; AUR and total encompassing bank area for each bank type in the n = 100 subset are compared to the overall UCA; AUR and total bank areas for each bank type n = 1636. (B) Bank distribution (location) across the United States based on geospatial data extracted from RIBITS.*

Chapter 3.2 Tables

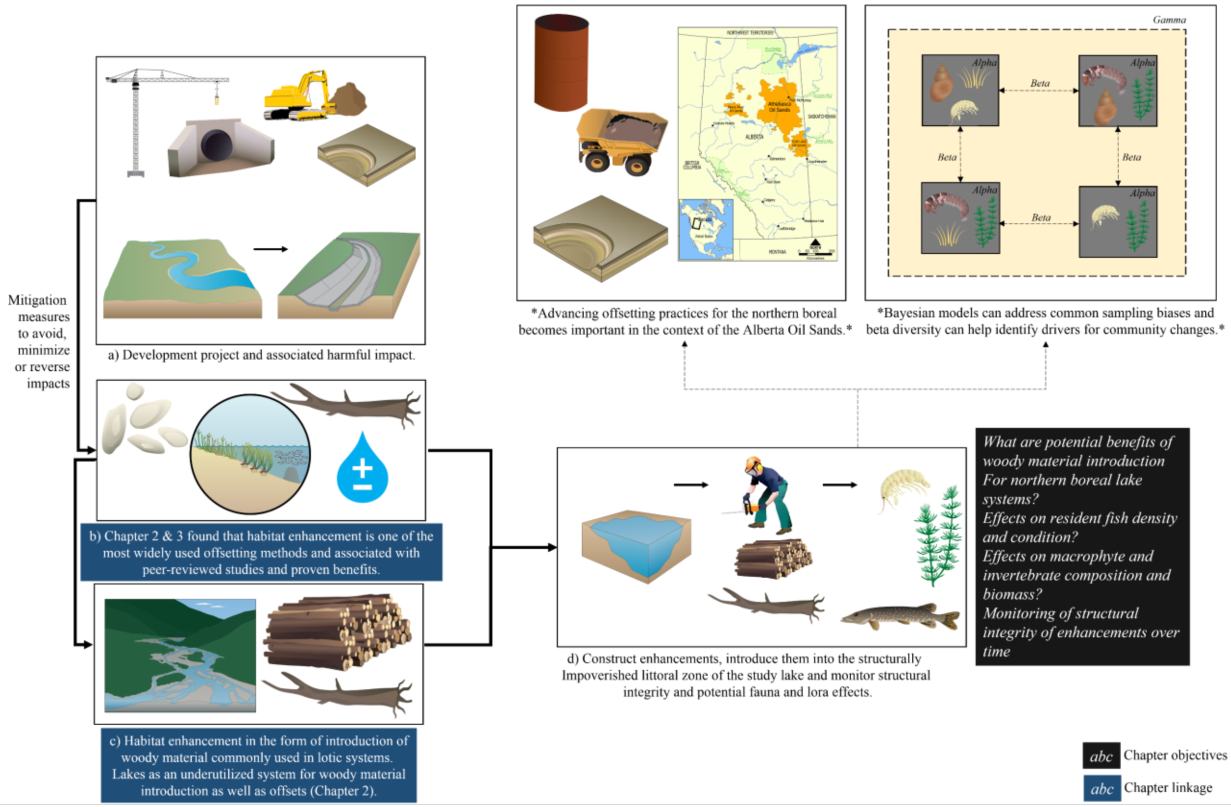
Table 3.2.1. *Overview of key definitions for bank types in RIBITS as well as according to bank targets, divided into Mitigation and Conservation banks targeting Wetlands; Streams; Multi-Ecosystems and Species. Extracted and calculated main variables (bank metrics) for the bank types based on 1636 assessed banks (Added Capacity (YAC); Unreleased Capacity (UCA) and Available Unwithdrawn Reserve (AUR)).*

Bank types:		Reference: RIBITS; (US)FWS; (US)EPA; USACE	
Mitigation	**Conservation**	**ILF (In-lieu fee program)**	**Umbrella**
A site where wetlands, streams, or riparian areas are established, rehabilitated, enhanced, or preserved in order to offset authorized by the Department of Army permits.	Permanently protected sites managed for endangered species, threatened species, or species at risk. The aim is to offset adverse impacts to the protected species occurring off-site. Permits managed by USFWS.	Rehabilitation, establishment, enhancement, and/or preservation of habitat area or ecosystem function through funds paid to a governmental or non-profit natural resources management organization. The operation and use of an in-lieu fee program are governed by an in-lieu fee program instrument thus differing from mitigation banks as well as allowing out of kind mitigation.	One banking instrument that dictates general requirements for an array of current and future sites (e.g., management and oversight of individual site plans to add future sites to the program).
Bank types according to bank targets:			
	Mitigation banks		**Conservation banks**
Wetland (n = 897)	**Stream (n = 253)**	**Multi-Ecosystem (n = 385)**	**Species (n = 101)**
Targeting wetlands	Targeting riverine systems	Targeting multiple ecosystems	Targeting a specific/ multiple species
Extracted main variables: Bank Number & Bank transactions linked to impacts			
Bank numbers			
n = 1636	The number of approved banks post 1995 sites divided into the 4 Bank types with transaction and bank information to calculate bank metrics.		
Bank metrics (yearly basis 1995 to end of 2020 and for each of the 4 Bank types)			
Yearly added capacity: Number of new banks per bank type in a year and their size (acres; linear feet for stream area). E.g., 23 Wetland banks established in 1998 with a total size of 4221 acres. **Indicator:** Indication for new bank numbers and bank size. Captures trends if bank size and bank numbers increase or decrease over time (*supply bottleneck*). **Unreleased capacity (UCA):** Area for each bank type each year that was not approved be available in credits yet either due to a release schedule or the bank not meeting performance criteria. UCA is cumulative, meaning UCA from e.g., 1997 impacts total UCA for 1998, etc. **Indicator:** Indication for how fast bank release schedules is or how well banks meet performance criteria. UCA outpacing AUR indicates higher demand and slower release, potentially creating bottlenecks of unreleased area (*release schedule bottleneck*) for specific bank types. **Available Unwithdrawn Reserve (AUR):** Area for each bank type each year that is available to be withdrawn in credits. AUR is cumulative, meaning AUR from e.g., 1997 impacts total AUR for 1998, etc. **Indicator:** Indication for demand, e.g.: are credits for a specific bank type being withdrawn or are they just accumulating/ unused while being available. AUR outpacing UCA indicates low demand for specific bank types/ demand not exceeding credit release or bank establishment. High AUR for a specific bank type could indicate a demand bottleneck (*demand bottleneck*) or regional restrictions based on bank location and service area (*regional bottleneck*).			
Credit release schedule: specifications of benchmarks and performance milestones that are necessary to be met by a bank to release further area in the form of credits. Mitigation and conservation bank release schedules have			

commonly comprised a mix of ecological criteria (management plan) and operational aspects: e.g., easement and financial assurance). Advance credits can be issued through in-lieu fee programs prior to providing the actual mitigation benefit or gain.

Chapter 4: Habitat enhancement in northern boreal lakes – methodological and theoretical considerations for assessing efficacy and project outcomes

Graphical synopsis of key background concepts, terms, and Chapter goals



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Chapter 4.1: Coarse woody habitat use and structural integrity of enhancements over time in a shallow northern boreal lake assessed in a Bayesian modeling approach

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Summary

1. The introduction of coarse woody habitat has been a widely adopted management practice for restoring and enhancing freshwater aquatic ecosystems. Although responses of aquatic fish and invertebrate communities have largely been documented for lotic systems, benefits for lentic ecosystems have been mostly unevaluated.
2. We tested the responses of fish populations to coarse woody habitat structures through a Bayesian modeling approach in a northern boreal lake in Alberta, Canada by enhancing a stretch of littoral zone with low structural complexity through introduction of coarse wood bundles and whole tree structures. The study site was split into three treatments, a

Spaced treatment (structures 30 m apart), a *Clustered* treatment (structures 15 m apart), and an unaltered area (*Control*).

3. Catch per unit effort and Catch per unit area data was collected over 2 years and posterior model predictions showed an increase in habitat use of the enhanced areas by resident fish (spottail shiner - *Notropis hudsonius*; northern pike - *Esox lucius*; white sucker - *Catostomus commersonii*; brook stickleback - *Culaea inconstans*), while no probable effect on overall fish health, measured in *Relative Weight*, was linked to the enhancements.

4. Across the two-year study, wood bundles degraded faster compared to the whole tree drops, coinciding with leveling off catch per unit effort and catch per unit area predictions near wood bundles, although catch predictions increased near the whole tree structures. Structural degradation set in as early as 1 week post construction for wood bundles and was mostly related to anchoring aspects.

5. Results from our study provide evidence for the benefits provided by coarse woody habitat within northern boreal lake systems and highlights the short-lived nature of wood bundles build with biodegradable substances, while offering evidence on the feasibility and utility of predictive modeling frameworks in addressing pseudoreplication and providing informative value for ecological studies.

4.1.1 Introduction

The loss and degradation of fish habitat in aquatic ecosystems in recent decades has led to increased restoration and mitigation efforts across the globe (French McCay & Rowe 2003; Sass et al. 2017). Specifically, threats of loss and degradation of fish habitat from development projects and urbanization pose an especially hard to address challenge as countries aim to balance the needs of economic activity with preservation of nature (Francis & Schindler 2006; Jennings et al. 1999). Consequently, many countries have implemented offsetting measures into legislation to compensate for irreversible changes or harm because of anthropogenic influences (e.g., Moilanen 2013; Schoukens & Cliquet 2016). This has led to many restoration or offsetting projects utilizing habitat enhancements to conserve ecosystem services (Theis et al. 2019).

The introduction of coarse woody habitat (CWH) is a common technique used in aquatic ecosystems aimed to reverse local negative effects or increase fish health and productivity (Wills et al. 2004). Coarse woody habitat fills a key role in lentic and lotic systems and has been shown to aid in developing structure for fish to utilize across life stages (Everett & Ruiz 1993; Sass et al. 2011; Sass et al. 2019). Notably, CWH is often used in stream restoration efforts to create scour pools, increase bank stability, provide cover, and allow invertebrate colonization (Larson et al. 2001; Sass et al. 2019). Coarse woody habitat introduction can also lead to an increase in organic matter retention and changes in macroinvertebrate communities (Francis & Schindler 2006; Glaz et al. 2009). However, a key shortcoming of enhancing physical habitat structure through CWH, is the high rates of structural failures when exposed to different flow regimes, flooding, and seasonal changes (Shields et al. 2004). Further, few studies have tested for temporal changes of CWH introduction and therefore the evaluation of potential long-term benefits for fish habitat enhancement remains relatively unknown (Pander & Geist 2016).

Another issue in restoration and enhancement studies is the often overall small or confounded treatment sample size, leading to pseudoreplication (e.g., Lazic et al. 2020). Not meeting genuine replication requirements can lead to wrong biological inferences. Emphasizing approaches based around predictions has become more and more popular in ecological studies proposing a solution to this issue by taking advantage of probabilistic statements and conclusions as opposed to focusing solely on parameter estimates (e.g., Billheimer 2019; Lazic et al. 2020). Aside from the so called 'unit of analysis' problem of pseudoreplication, spatial autocorrelation often coincides with biological sampling procedures, especially when testing the effect of habitat modification due to close spatial proximity of structures, animal movement and habitat coupling effects (e.g., Dormann 2007; Zuckerberg et al. 2020).

The reported and proven effects of introducing or re-introducing CWH have seldom been investigated and shown for boreal lake systems (Sass et al. 2011; Sass et al. 2019). Northern boreal lakes have been exposed to an increasing amount of development (e.g., mining; Leppanen et al. 2017) and climate induced stress (e.g., Schindler et al. 1996), while experiencing other extreme disturbances on a regular basis (e.g., long winters; fire regimes), often limiting woody habitat input (Gennaretti et al. 2013). The wood regime of northern boreal lakes is generally characterized by long residence times (spanning 1400 years for black spruce) and low recruitment rates (5.8 pieces per century; Gennaretti et al. 2013). Reclamation, restoration, and offsetting measures in these areas are often unsuccessful and sparsely evaluated (Theis et al. 2019). Coarse woody habitat introduction could potentially provide short- and long-term benefits for northern boreal lake systems. To better understand the ecological influences of CWH for fish communities in northern boreal lake systems, we conducted a 3-year field study that tested for the response of different fish species in a shallow northern boreal plain lake while exploring a predictive modelling framework addressing pseudoreplication concerns and spatial autocorrelation offering a new perspective on the often-stated replication crisis. Here we tested (1) the effect of structure types of CWH (patchy distribution versus even, whole trees versus wood bundles) on catch per unit effort (CPUE) and catch per unit area (CPUA), (2) the effect of structure type on species condition, and (3) integrity of the structure types over time and long-term benefit provision. We hypothesize that the enhanced littoral fish habitat will lead to increased probabilities of having a higher relative abundance due to refuge and food availability, which would consequently increase fish condition. We further hypothesize that environmental factors will reduce structural integrity of enhancements over time consequently reducing the provided mentioned benefits on fish abundance over time.

4.1.2 Methods

Study Lake

Lake Steepbank is a small (185.4 ha, shallow, 16 m maximum depth) northern boreal plain lake in Alberta, Canada (Figure 4.1.1). The fish community in the lake is comprised of four species: northern pike - *Esox lucius*; white sucker - *Catastomus commersonii*; spottail shiner - *Notropis hudsonius* and brook stickleback - *Culaea inconstans* (Figure 1e). The remote location (2.5 hours to the nearest town, logging road access) and bag limits (one Pike > 70 cm) renders immediate anthropogenic influences as negligible.

Enhancement Sites and Treatments

We conducted a CWH count to assess structural richness and suitable areas for enhancement. Surveyed littoral area transects (n = 26) were 100 meters in length, where accessible, and surveyed up to a depth of 2 m using wetsuits (Table S1; Figure 1c). Coarse woody habitat in the individual transect was counted and recorded. Coarse woody habitat was split into two categories, small and large, depending on size and diameter (small: diameter 10-15 cm and 1.5 – 3 m length, large: >15 cm diameter and > 3 m). Structural richness was measured by dividing CWH counts over the total surveyed area (CWH/100 m^2; Figure 4.1.1C; Table S4.1.1). The area on the Northeast end of the lake was chosen as a suitable enhancement site, due to its low structural richness (0.2 CWH/100 m^2) and shallow littoral zone (Figure 4.1.11B; Table S4.1.1). The area was divided into three zones, with a different treatment type. Coarse woody habitat structures were built on site from local trees and deadwood, being jack pine, *Pinus banksiana*, white spruce, *Picea glauca* and aspen poplar, *Populus tremuloides michx* (Inkpen & Van Eyk 1986). Coarse woody habitat enhancements were divided into two types: coarse wood bundles and whole tree drops. Wood bundles consisted of eight logs per bundle with a minimum diameter of 10 cm per log and were cut to 1.8 m in length. Bundles were tied together with biodegradable hemp rope and four burlap sandbags were used as weight per bundle. Wood bundles were submerged at a depth of 1.5 to 2 meters. Tree drops were whole trees (4 to 6 m length) half submerged, anchored into the ground on shore and additionally tied to the remaining stump. Two trees per enhancement were put into a V-shape to provide additional structure and cover. Fully crowned trees were chosen to ensure slower degradation time compared to older, fallen trees and to add additional structural complexity (Newbrey et al. 2005; Sass et al. 2011). There were three zones of treatments within the lake to test for the effect of spacing between treatment types (Figure 4.1.1B). Zone one of the enhancement areas received a *Clustered* treatment with five bundles and five tree structures. Trees and bundles were spaced 15 meters apart. Zone two was left as an unenhanced *Control* area and was situated between the two enhanced areas (Figure 4.1.1B). A *Spaced* treatment was applied to zone three, with five bundles and five tree structures spaced 30 meters apart (Figure 4.1.1B).

Catch per unit effort and catch per unit area assessment

Fish were sampled before enhancement, four weeks after enhancement, as well as one- and two-years post enhancement (July to August). Samples were collected through seine netting and minnow traps. Minnow traps were set for 12 hours from 8 pm to 8 am. Here 10 sets with three traps per enhanced area and *Control* area were set at a time over a period of four weeks. Traps were baited with Aluminum foil (visual attractant), cheese puff snacks (Cheetos) and onions (experience based). Seine transects were 50 meters long and done once a week on days without set minnow traps. Each seine haul covered one 50-meter transect per area and was repeated three times. Captured fish were weighed (grams), Research measured (millimeters, fork length and total length) and then released as per License. Catch per unit effort for minnow traps and CPUA for seine hauls was used for relative abundance predictions and posterior probability differences between treatments and across time done in R (version 4.1.1) and a Stan extension (version 2.21.2), through a Bayesian multilevel model. The Bayesian model aims to provide posterior probabilities of CPUE and CPUA (predicted) differences across time and between treatments (predictor, e.g., probability that a random sample from A > B by taking the model and prior information into account). Predictions are based on the posterior of the model, derived from repeated simulations. The required prior for Bayesian models which contains general knowledge about the data at hand was chosen through the brms package for CPUE and CPUA (brms version 2.16.1; Table A4.1.4-4.1.7 & S4.1.1-4.1.8). We chose non-informative priors covering distributions with a range of uncertainty larger than any plausible parameter value (class "b" – population level effect, normal prior 0, 30 CPUA and 0, 3 CPUE; Gelman & Hill 2007). Weakly informative priors or improper priors were discarded due to poor model performance (Lemoine 2019). The model included a blocking factor for each treatment acknowledging that samples for each sampling time and treatment are derived from the same area (hierarchical structure). The utilized Markov Chain Monte Carlo (MCMC) simulation was run with 20000 iterations on 4 chains with an accepted Delta value of 0.9999 and rhat ($\hat{R}$) (ratio of the variance of a parameter across all chains) value of < 1.05 as well as controlling for Bulk Effective Sample size (Bulk ESS) and Tail Effective Sample size (Tail ESS) values of > 100. Both estimates can be used as a diagnostic for sampling efficiency in the posterior results (Vehtari et al. 2019). Trap location and seine location were marked through GPS and used to determine if sampling was done in proximity to wood bundles or tree structures (Type) to allow for testing whether proximity to a specific enhancement type influences CPUE or CPUA results. Spatial autocorrelation was controlled through Moran's I estimates based on the longitude and latitude data of samples (e.g., are CPUE near a bundle related to CPUE of a nearby Tree structure; Table A4.1.3). Alpha values < 0.05, indicate spatial autocorrelation (forecast 8.15; Hyndman 2021).

Species Condition

Weight in combination with species specific slope and intercept values from the FSA package for R were used to calculate fish condition as measured in *Relative Weight* (Wr) (Blackwell et al. 2000; Ogle et al 2021; FSA 0.9.0; FSAdata 0.3.8). *Relative Weight* is one of the most accepted and reliable condition indicators for fish (Blackwell et al. 2000). Predicted mean *Relative Weight* was calculated through a separate Bayesian Model and compared through probability of differences (15000 iterations; 3 chains; Delta 0.9999; Prior Shiner = 0, 175; Prior Pike = 0, 140; Prior Sucker = 0, 120; Table A4.1.8-4.1.10).

Structural Integrity and long-term Benefits

We regularly checked on the structural integrity of the enhancements (1 Day; 1 Week; 1 Month; 3 Months; 1 Year & 2 Years post construction; Table A4.1.2). Due to the shallow depth and fixed location, visual and snorkeling assessments were feasible for the large structures as opposed to fish surveys. Structural changes, degradation or failure was documented and transformed into six categories per enhancement type (%) ranging from full structural integrity and fixed location to complete structural failure and moving of the structure (Table 4.1.1; Table A4.1.2; Figure 4.1.4). Each of the categories responded to one of three structural aspects exposed to stressors (Integrity; Anchoring; Placement; Table A4.1.2). Mean structural integrity (SI in %) for, wood bundles and tree structures, was plotted over time and compared between enhancement types (ezANOVA for type III; type as factor and time as case identifier; ez version 4.4-0, Lsmeans comparison, Alpha < 0.05; Shapiro Wilk test for normality; Levene's test for homogeneity of Variance; Table A4.1.4) and with corresponding predicted mean CPUA and CPUE values (Table A4.1.11; A4.1.12). This comparison estimates the relationship between structural integrity and provided benefits throughout time. With the objective to test the long-term benefits and natural life cycle of CWH for northern boreal lakes, no repairs or adjustments were made.

4.1.3 Results

Catch per unit effort and catch per unit area

Catch per unit effort (fish per trap over 12 hours) predictions increased over time and differed between enhanced and unenhanced areas as well as across years and treatments (Figure 4.1.2; Table A4.1.6). Mean minnow trap predictive CPUE for the *Control* area was at 0.20 (± 1.34) fish per trap per hours for all four sampling times. Catch per unit effort predicted means for the enhanced areas increased from an average of 0.16 (± 0.24) fish/hr pre-enhancement to 0.61 (± 0.24) fish/hr, 4 weeks post construction, 1.06 (± 0.24) fish/hr (7-fold increase) after one year and 1.27 fish/hr (± 0.24) (8-fold increase) after 2 years (Figure 4.1.2; Table A4.1.6). The probability of CPUE being higher between enhanced and unenhanced treatments increased for the *Spaced* treatment from 0.90 (4 weeks post; Figure 4.1.2B), to 0.99 (1 year post; Figure 4.1.2C) and 0.99 (2 years post; Figure 4.1.2D) and for the *Clustered* treatment from 0.88 (4 weeks post; Figure 4.1.2B), to 1.0 (1 year post; Figure 4.1.2C) and 1.0 (2 years post; Figure 4.1.2D). Predicted

CPUE between the *Clustered* treatment and *Spaced* treatment did not differ for the first 2 sampling periods but was higher for the *Clustered* treatment 1- and 2-years post construction with 1.15 (± 0.24) fish/hr compared to 0.97 fish/hr (± 0.24; Probability 0.70; Figure 4.1.2C) and 1.55 (± 0.24) fish/hr compared to 0.95 fish/hr (± 0.24; Probability 0.95; Figure 4.1.2D).

Pre-enhancement seining hauls yielded a predicted mean CPUA of around 10.44 (± 2.90) fish/100 m^2 and did not differ across the three areas (Figure 4.1.3A; Table A4.1.7). Predicted mean CPUA differed over time and between enhanced areas and the unenhanced *Control* section as well as over time and treatments (Figure 3; Table A4.1.7). Predicted CPUA for the *Control* area ranged between 7.85 (± 2.91) fish/100 m^2 to 10.91 (± 2.90) fish/100 m^2 over the four sampling periods and increased for the enhanced areas from the initial 10.66 (± 2.90) fish/100 m^2 to 13.52 (± 2.90) fish/100 m^2, 18.49 (± 2.90) fish/100 m^2 and 24.06 (± 2.92) fish/100 m^2 after two years (Figure 4.1.3; Table A4.1.7). The probability of CPUA being higher between enhanced and unenhanced treatments increased for the *Spaced* treatment from 0.87 (4 weeks post; Figure 4.1.3B), to 0.98 (1 year post; Figure 4.1.3C) and 0.99 (2 years post; Figure 4.1.3D) and for the *Clustered* treatment from 0.94 (4 weeks post; Figure 4.1.3B), to 1.0 (1 year post; Figure 3c) and 1.0 (2 years post; Figure 4.1.3D). Predicted mean CPUA was higher (Probability 0.99) in the *Clustered* (29.22 ± 2.91 fish/100 m^2) treatment after two years compared to the *Spaced* treatment (18.90 ± 2.93 fish/100 m^2; Figure 4.1.3D).

Mean CPUE and CPUA based on posterior predictions were higher for the enhanced areas compared to the *Control* section (probabilities ~ 99% after 2 years) with the *Clustered* treatment showing higher predicted mean CPUE and CPUA values compared to the *Spaced* treatment 2 years post enhancement.

Species-specific catch per unit effort and catch per unit area

Increases in predicted mean CPUE and CPUA based on the model's posteriors were observed in species levels and species-specific proportionate presence, with each species contributing differently to the changes.

Spottail shiner relative abundance estimated by predicted CPUE and CPUA differed across treatments as well as treatment and time. Spottail shiner catches stayed consistent in the *Control* area with predicted minnow trap CPUE ranging between 0.13 (± 0.23) and 0.19 (± 0.23) fish/hr (Figure S4.1.1; Table S4.1.1) and predicted mean seine haul CPUA of 7.64 (± 2.97) fish/100 m^2 to 9.42 (± 2.96) fish/100 m^2 over the four sampling periods (Figure S4.1.2; Table S4.1.2). Mean predicted relative abundance differences of spottail shiners across treatments increased in their probability over time. Predicted spottail shiner catches for the *Spaced* treatment were higher for all three post enhancement sampling periods with 11.78 (± 2.97), 15.32 (± 2.97) and 17.56 (± 2.96) shiners/100 m^2 caught seining and 0.54 (± 0.23), 0.84 (± 0.23) and 0.81 (± 0.23) fish/hr caught in minnow traps compared to 0.15 (± 0.23) fish/hr (minnow trap) and 10.24 (± 2.97) fish/100 m^2 (seine) for the pre-enhancement assessment (Figure S4.1.1; S4.1.2; Table S4.1.1; S4.1.2). A similar increase in predicted mean catch compared to the pre-enhancement assessment was observed for spottail shiners caught in minnow traps in the *Clustered* treatment area (0.15 ± 0.23 to 0.51 ± 0.23, 1.01 ± 0.23 and 1.28 ± 0.23 fish/hr) and through seining, increasing from an initial predicted mean CPUA of 9.85 (± 2.95) fish/100 m^2 to 13.28 (± 2.96), 18.32 (± 3.00) and 27.09 (± 2.94) fish/100 m^2 (Figure S4.1.1; S4.1.2; Table S4.1.1; S4.1.2).

The probability of CPUE for shiners being higher between enhanced and unenhanced treatments increased for the *Spaced* treatment from 0.86 (4 weeks post; Figure S4.1.1), to 0.98 (1 year post; Figure S4.1.1) and 0.98 (2 years post; Figure S4.1.1) and for the *Clustered* treatment from 0.84 (4 weeks post; Figure S4.1.1), to 0.99 (1 year post; Figure S4.1.1) and 1.0 (2 years post; Figure S4.1.1). Mean CPUA for shiners being higher between enhanced and unenhanced treatments increased in its probability for the *Spaced* treatment from 0.83 (4 weeks post; Figure S4.1.2), to 0.97 (1 year post; Figure S4.1.2) and 0.98 (2 years post; Figure S4.1.2) and for the *Clustered* treatment from 0.91 (4 weeks post; Figure S4.1.2), to 0.99 (1 year post; Figure S4.1.2) and 1.0 (2 years post; Figure S4.1.2). Catch per unit effort and CPUA for the *Clustered* treatment were noticeably higher in year two compared to the *Spaced* treatment (Probability 0.93 & 0.99).

Juvenile northern pike predicted CPUA differed across treatments and treatments and time (Figure S4.1.4; Table S4.1.4). Differences in predicted CPUE were only captured to a small degree between treatments (Figure S4.1.3; Table S4.1.3). Juvenile pike predicted relative abundance was low in the *Control* area across time with minnow trap CPUEs of 0.01 (± 0.05), <0.01 (± 0.05), 0.01 (± 0.05) and 0.01 (± 0.05) fish/hr and a CPUA for seine hauls of 0.40 (± 0.36), 0.08 (± 0.36), 0.08 (± 0.36) and 0.08 (± 0.36) fish/100 m^2 (Figure S4.1.3; S4.1.4; Table S4.1.3; S4.1.4). Juvenile northern pike relative abundance increased slightly for the *Spaced* treatment according to the predicted mean, from pre-construction trap CPUE of 0.02 (± 0.05) to 0.04 (± 0.05), 0.06 (± 0.05) and 0.04 (± 0.05) fish/hr (4 weeks, 1 year, 2 years post-enhancement) and changed from 0.47 (± 0.36) to 0.37 (± 0.36), 0.56 (± 0.36) and 0.63 (± 0.36) fish/100 m^2 caught in seine hauls (Figure S4.1.3; S4.1.4; Table S4.1.3; S4.1.4). Juvenile northern pike CPUE predictions increased for the *Clustered* treatment from pre-construction levels of 0.01 (± 0.05) to 0.06 (± 0.05), 0.06 (± 0.05) and 0.08 (± 0.05) fish/hr caught in minnow traps (4 weeks, 1 year, 2 years post-enhancement) and CPUAs of 0.23 (± 0.36) to 0.56 (± 0.36), 0.78 (± 0.36) and 1.03 (± 0.36) fish/100 m^2 caught in seine hauls (Figure S4.1.3; S4.1.4; Table S4.1.3; S4.1.4).

The probability of CPUE for juvenile northern pike being higher between enhanced and unenhanced treatments increased for the *Spaced* treatment with probabilities of 0.72 (4 weeks post; Figure S4.1.3), 0.77 (1 year post; Figure S4.1.3) and 0.68 (2 years post; Figure S4.1.3) and similarly for the *Clustered* treatment with probabilities of 0.79 (4 weeks post; Figure S4.1.3), to 0.75 (1 year post; Figure S4.1.3) and 0.86 (2 years post; Figure S4.1.3). Mean CPUA for juvenile northern pike being higher between enhanced and unenhanced treatments increased in its probability for the *Spaced* treatment from 0.72 (4 weeks post; Figure S4.1.4), to 0.83 (1 year post; Figure S4.1.4) and 0.86 (2 years post; Figure S4.1.4) and for the *Clustered* treatment from 0.82 (4 weeks post; Figure S4.1.4), to 0.92 (1 year post; Figure S4.1.4) and 0.97 (2 years post; Figure S4.1.4).

White sucker CPUA predictions differed across treatments and treatments and time (Figure S4.1.6; Table S4.1.6). White suckers went from being almost completely absent to present in the treatment areas while negative predictions show the consistent absence of white suckers in the *Control* area throughout all sampling times (Table S4.1.5; S4.1.6). White sucker detected presence predictions went in the *Spaced* treatment from <0.001 (± 0.06) for CPUE and <0.01 (± 0.32) CPUA to a maximum of 0.11 (± 0.06) fish/hr caught in minnow traps 2 years post enhancement and 0.63 (± 0.32) fish/100 m^2 through seining 1-year post enhancement (Figure S4.1.5; S4.1.6; Table S4.1.5; S4.1.6). White sucker catch increases were detected in the *Clustered* treatment in seining (0.04 ± 0.32 to 0.40 ± 0.32, 0.70 ± 0.32 and 0.82 ± 0.32 fish/100 m^2) and minnow traps (<0.001 ± 0.06 to 0.02 ± 0.06, 0.06 ± 0.06 and 0.16 ± 0.06 fish/hr) (Figure S4.1.5; S4.1.6; Table S4.1.5; S4.1.6). White suckers, though low in relative abundance, increased steadily in their predicted presence in the *Clustered* treatment over time.

The probability of CPUE for white sucker being higher between enhanced and unenhanced treatments did increase for the *Spaced* treatment, with probabilities of 0.64 (4 weeks post; Figure S4.1.5), 0.73 (1 year post; Figure S4.1.5) and 0.89 (2 years post; Figure S4.1.5) and similarly for the *Clustered* treatment with probabilities of 0.60 (4 weeks post; Figure S4.1.5), to 0.76 (1 year post; Figure S4.1.5) and 0.94 (2 years post; Figure S4.1.5). Probability of predicted mean CPUA for white sucker being higher between enhanced and unenhanced treatments increased for the *Spaced* treatment from 0.72 (4 weeks post; Figure S4.1.6), to 0.92 (1 year post; Figure S4.1.6) and 0.88 (2 years post; Figure S4.1.6) and for the *Clustered* treatment from 0.82 (4 weeks post; Figure S4.1.6), to 0.94 (1 year post; Figure S4.1.6) and 0.96 (2 years post; Figure S4.1.6).

Brook stickleback mean predicted CPUE was low throughput all sampling times and treatments. (Figure S4.1.7; Table S4.1.7). Probabilities for CPUE differences were low ranging around 0.6 across treatments and years (Figure S4.1.7). Catch per unit area based on posterior predictions did not change noticeably across time and treatments. Mean predicted CPUA for the *Spaced* treatment was at 0.23 (± 0.26) with the highest catch being 2019 (0.30 ± 0.26). Mean predicted CPUA for the *Clustered* treatment was higher (0.31 ± 0.26) though probabilities rarely exceeded 0.7 (Figure S4.1.8; Table S4.1.8). Mean predicted CPUA for the *Control* treatment was lower (0.10 ± 0.26) compared to the other two and did not change in any direction over time, with difference probabilities ranging between 0.24 and 0.34 (Figure S4.1.8).

Overall, brook stickleback abundances did not or only slightly change. White sucker predicted presence went from being mostly absent to present and increased in both enhanced treatments slightly over time. Spottail shiner predicted relative abundance increased the most compared to all species across time and treatments with high probabilities of higher abundances in enhanced areas compared to the unenhanced *Control* (> 95%), followed by northern pike (> 75%).

Species Condition

Fish condition measured in predicted *Relative Weight* was tested for spottail shiner, northern pike, and white sucker in 2020. Spottail shiner data was readily available for all treatments and time periods due to their high abundance. Predictions for spottail shiner in Steepbank Lake surpass reference population condition values by 48.5% on average (Table A8). Fish in the *Spaced* treatment area showed a higher predicted condition compared to the pre-enhancement samples form 2018 (+13%; Probability: 0.66) *Clustered* treatment in 2020 (+15%; Probability: 0.55) and *Control* treatment in 2020 (+10%; Probability: 0.63; Table A4.1.8), with the *Clustered* treatment displaying the lowest predicted values. Data availability allowed us to compare *Relative Weight* for northern pike for 2020 for both treatments to a lake wide reference for 2020. The *Relative Weight* prediction was slightly higher for the enhanced areas; (*Clustered*: + 10%; Probability: 0.63; *Spaced*: 11%; Probability: 0.55) compared to the lake mean (Table A4.1.9). Overall predicted condition for northern pike for the lake was higher compared to the FSA reference values (Table A9; +25.6%). Due to overall sample sizes and presence absence restrictions (insufficient detection in the *Control* area and pre-enhancement), overall condition for white suckers from both enhanced areas from 2020 was compared to specimens caught in other parts of the lake while seining and in minnow traps (Table A4.1.10). *Relative Weight* for suckers based on the posterior predicted mean from enhanced areas was not different compared to the lake reference (+0.41%; Probability: 0.53; Table A4.1.8). Predicted *Relative Weight* for the lake and the enhanced areas was not different compared to other sucker populations (Table A4.1.10; +0.21%).

Structural integrity and long-term benefits

Results from our frequent structural integrity assessments show a clear difference between the whole trees anchored on shore and the submerged wood bundles (Figure 4.1.4; Table 4.1.1; Table A4.1.4). Although structural integrity was high 1 day, 1 week and 1-month post construction (~80% for all structures), the bundles lost integrity faster, with a mean integrity score of 65% after 12 months with logs missing and single sandbags detaching or moving from their original position compared to the whole tree structures (90% structural integrity after 12 months, Figure 4). Finally, after 2 years, all bundles were around or below 30% structural integrity with the logs spread out across the whole area (Figure 4.1.4F). Seven out of the 10 dropped and anchored doubletree structures were still in place after 2 years with structural integrity loss only becoming noticeable after 2 years (Figure 4.1.4F). One was damaged after the first year and two more after the second due to human intervention. Structural integrity between the two utilized enhancement types differed greatly after 1 month ($p < 0.001$; $F = 2.467$; $dfn = 1$; $dfd = 10$, Figure 4.1.4C; Table A4.1.4) with anchored trees staying structurally intact to a large degree. The overall inclusion of structure type in the Bayesian model (Table A4.1.11; A4.1.12) based on GPS location of samples showed different CPUE and CPUA predictions in the model over time (Figure 4.1.4). Predicted mean CPUE and CPUA for minnow traps and seining closer to CWH structures leveled off with decreasing structural integrity between year 1 (1.02 ± 0.34 CPUE; 18.07 ± 3.91 CPUA) and year 2 (1.08 ± 0.35 CPUE; 19.524 ± 3.89 CPUA; Figure 4.1.4E; 4.1.4F; Table A4.1.11) with a probability of 0.82 for CPUE and 0.73 for CPUA to yield higher catches along tree structures in year 2 (1.46 ± 0.34 CPUE; 28.56 ± 3.89 CPUA; Figure 4.1.4F; Table A4.1.12). Overall, positive benefits measured through CPUE and CPUA persisted throughout time near enhancements with high structural integrity although leveling out in areas with structural degradation and failure, starting after 2 years post construction. Samples were not spatially auto correlated within treatments as indicated by Moran's I calculations (Table A4.1.3; CPUE *Clustered* = 0.23; CPUE *Spaced* = 0.69; CPUA *Clustered* = 0.33; CPUA *Spaced* = 0.88).

Structural stressors over time

Coarse woody habitat bundles

Proportionate stressor presence showed that CWH bundles experienced sandbag leaks (16.7%), rope tears (16.7%) and first log and sandbag detachments (both 8.3% presence) 1 week after construction (Table 4.1.1). Anchor related issues like sandbag detachment (41.2%) and leaks (17.6%) increased in proportionate presence 4 weeks after construction with log detachment becoming a more predominant issue (23.5%). Similar issues persisted after the 3 months assessment (rope tear 4.3%; sandbag leak 21.7%, sandbag detached 34.8%; log detached 30.4% and minor moves 8.7%; Table 4.1.1). Structural integrity issues like rope tears (26.1% & 28%) and log detachment (30.4% & 36%) increased after year 1 and 2. The year 2 assessment recorded minor (12%) and major structural (24%) moves of CWH bundles (Table 4.1.1). Overall structural stressors for CWH bundles were recorded as early as 1 week post construction with structural integrity and anchor degradation. Anchor related issues predominantly increased between 1- and 3-months post construction and later were replaced by structural integrity failure and structural moves 1- and 2-years post construction (Table 4.1.1).

Tree structures

Tree structure integrity was overall very high for the first 3 months with only a minor stressor presence of undone tree joints (9.1%) and a breakage (9.1%) 3 months post construction (Table

4.1.1). Connecting joints of the two trees and tree breakage (both 23.1%) increased after 1 year with minor structural shifts being recorded (15.4%). Minor shifts (31.6%) and major moves (15.8%) increased in proportionate presence after 2 years as well as the first cases of anchoring issues (Table 4.1.1). Ripped anchor lines (15.8%) or unearthed anchors (21.1%) contributed to integrity degradation of the tree structures, while undone joints (10.5%) or broken trees (5.3%) were only present on a small scale to structural loss 2 years post constriction (Table 4.1.1).

4.1.4 Discussion

Catch per unit effort and catch per unit area effects

Species specific abundance predictions based on the collected data were closely linked to the introduction of CWH. The structural assessment of littoral habitat complexity covering the whole lake, as well as pre-enhancement fish sampling through trapping and netting, showed low abundance and structural complexity for the littoral area (1.18 CWH/100 m^2) of the lake, as well as the selected enhancement areas (0.24 CWH/100 m^2). Given the study period (2 years) as well as the local scale of the study we do not assume an increase in lake-wide population size or biomass, but rather a change in habitat use and area-dependent recruitment for the four fish species. Especially juvenile and fast-growing, short-lived species (e.g., spottail shiner) likely responded faster from structural enrichment providing nursery and rearing habitat (Fodrie & Levin 2008). Coarse woody habitat appeared to benefit three out of four fish species in the lake. The largest catch increase and modeled abundance was observed for spottail shiner which are a fast growing, schooling fish species, usually reaching a maximum age of 3 to 5 years (Ignasiak et al. 2017; Smith & Kramer 1964). Spottail shiner show a distinct separation between juvenile and adult specimen diet. Juvenile shiner predominantly feed on rotifers, algae, and small crustacean species while adults prefer larger crustaceans, fish eggs, as well as flies, damselfly, and dragonfly larvae (Smith & Kramer 1964).

Coarse woody habitat introduction has been shown to increase food sources such as periphyton and invertebrates (e.g., Coe et al. 2009; Theis et al. unpublished; Tullos et al. 2006). Thus, increased food availability, as well as additional refuge and habitat use are likely the driving factors increasing spottail shiner abundance in the enhanced areas, while the *Control* areas showed no change in relative abundance (Everett & Ruiz 1993; Sass et al. 2011). Habitat use by northern pike and white sucker differed compared to spottail shiner, mainly due to the age selective nature of our sampling gear regarding large-bodied fish species. Neither minnow traps nor seining collected adult specimens of northern pike. Northern pike tend to be more versatile in habitat use and are not limited to shallow lake regions with high vegetation density (Pierce 2012). Flexibility in habitat selection gives predators an advantage to exploit prey sources more effectively. The created CWH structures provide refuge for juvenile and young of the year (YOY) northern pike from predation. The increase in northern pike abundance was mostly noticeable through seining. Minnow trap catches of northern pike were lower and directly related to spottail shiner presence in the traps (pike exclusively caught in traps with shiner presence), suggesting that northern pike did not respond to the trap bait itself but fish movement in the traps. It also suggests the active preying of young northern pike on spottail shiner, linking the local increase of northern pike to the increase in spottail shiner relative abundance. Overall, the modeled increase in northern pike catches shows that CWH structures are being used by juvenile and YOY northern pike providing refuge and food. While spottail shiner and northern pike populations model predictions responded faster to the structural enhancements (noticeable 4 weeks post construction), White sucker abundance only increased noticeably 1 to 2 years post construction. No suckers (except 1) were caught in the area pre-enhancement, showing that the unstructured sandy littoral area was most likely not utilized at all. White suckers are predominantly benthic feeders and changes in benthic community due to enhancements, CWH degradation and organic matter accumulation along enhancements, tend to take longer than the immediate structural benefits or early colonization by invertebrates or periphyton (Coe et al. 2009; Francis & Schindler 2006; Tullos et al. 2006). Overall, benefits provided by CWH introduction for early life-stages of white suckers tend to take longer to manifest and thus potentially delayed the habitat use and abundance response for this species. Finally, there was no change in brook stickleback predicted relative abundance as a response to habitat enhancement. Brook stickleback populations in these lake systems are generally small and do not contribute significantly to the overall ecosystem biomass. Brook sticklebacks tend to be present in higher numbers in smaller streams and creeks rather than in lake systems (Nelson & Atton 1971; Reisman & Cade 1967). Considering no changes pre- and post-enhancement suggests a more generalist habitat selection for brook stickleback in northern boreal lakes, not specifically being tied to CWH introduction as well as overall low population numbers in the study lake (~5 to 10 specimens for all sampling gears per sampling season throughout the whole lake). Overall, most predicted differences exceeding probability of 90% were noticeable 2 years post construction underlining the required time for enhancements to be likely to impact fish abundance in a notable manner.

Fish condition in Relative Weight

Fish condition is another important factor generally assessed for enhancement or restoration projects. Increased shelter and food availability are often linked to an increase in fish condition (Baldigo et al. 2008). Using *Relative Weight* assessed in 2020, 2 years post enhancement, through posterior predictions based on the sampled fish, for the three species responding to the enhancements shows that for our study lake predicted fish condition did not differ in most cases across treatments, though slightly higher in the enhanced areas with low probabilities for differences rarely exceeding 60%. White sucker *Relative Weight* did not differ between the overall lake mean and the enhanced areas. Increased predicted relative abundance but no clear condition differences show that white sucker benefited from the CWH in terms of habitat use but did not improve condition, with overall condition being deemed not above normal according to the reference populations provided in the FSA package for R (Ogle et al. 2021). These results seem logical since fish in the lake are already healthy in terms of weight and that sampled white suckers were mostly juvenile or YOY.

Northern pike displayed a similar response. Predicted *Relative Weight* for pike in the enhanced areas was slightly higher than the overall lake means (~10%) with low predicted probabilities for differences. This difference could be due to the increase in spottail shiner abundance, providing the main food source for northern pike in the system. *Relative Weight* predictions for all three samples were above the FSA reference values (~25%) (Ogle et al. 2021), indicating an overall healthy pike population. Similarly, to white suckers, benefits for adult pike through CWH enhancements would need to be monitored differently. Finally, spottail shiner condition predictions increased the most for the *Spaced* enhancement area. This difference could be due to density dependency. Higher competition for food and the overall density effect of schooling fish could have led to a reduced beneficial effect of CWH on condition for the *Clustered* treatment (e.g., Casini et al. 2016; Johnson 2007), while no spatial autocorrelation was detected within treatments that would support abundance differences due to the closer proximity of the structures to each other (e.g., Hawkins et al. 1983). The overall higher predicted condition for spottail shiner compared to the FSA reference (Ogle et al. 2021) (~50% higher), is likely due to the high parasite load of *Ligula intestinalis* in Shiners in the lake systems in the study area (Finn et al. unpublished). Dissections of spottail shiner have shown to carry parasite loads ranging between 18 to 41% of their body weight in 95% of all sampled shiners (Finn et al. unpublished), differing from reference rates of around 5% (Szalai et al. 1989). These findings point out that habitat use, or relative abundance increases do not necessarily translate to condition benefits in already healthy populations, especially when considering low probabilities for difference across sites.

Structural integrity and benefits over time

Most enhancement projects face structural issues, struggling to keep the enhancements in place (D'Aoust & Millar 2000; Gregory & Davis 1992). This especially applies for the wood bundles, mainly due to the fact of using degradable rope and hemp bags. Combined with the predicted CPUE and CPUA data, and probabilities differences of 77.5% after 2 years, these structural differences suggest that wood bundles provide benefits on a shorter timescale and may not be suitable for long-term enhancement measures, given the materials used to bundle and weigh down the structures. Our findings underline the key importance of anchoring for woody habitat enhancements and show the progression of anchoring issues translating to structural degradation and stress exposure over time leading to the ultimate dispersal of the bundles. Often recorded for lotic systems these principles clearly apply to lentic systems as well and their system internal movements and seasonal stressors like wind and snow especially in northern boreal environments. Heavier and more enduring anchors could potentially help prevent these early on setting issues (D'Aoust & Millar 2000; Gregory & Davis 1992; Shields et al. 2004). Utilizing whole tree structures and their rooted leftover stumps as anchors has proven to provide substantially more stability as indicated by our results. Many habitat enhancement projects are aimed to provide positive ecosystem effects on a long-term scale or in perpetuity but sustained benefits especially in aquatic ecosystems are still often questionable (D'Aoust & Millar 2000; Pander & Geist 2016). Taking our findings into account, many large-scale wood-bundle introductions may not provide the expected benefits for fish populations on the desired time scale when using biodegradable materials. Nonetheless, the advantage of lake systems compared to lotic systems is that even when structural integrity is lost, the woody material stays in the system. Our observations showed that the individual logs stayed in piles or spread out across the area to some degree. Catch per unit effort and catch per unit area predictions leveled out along wood bundles compared to whole tree structures, hinting that logs that stay in the general area still providing partial benefits. However, this could also be attributed to a not yet apparent time-lag (e.g., Wilson et al. 2011). The whole tree structures stayed in place even when exposed to ice cover in the winter and accumulated organic matter due to their branches and V-shaped make-up. Based on that, use of both structures in the littoral zones of shallow lakes should maximize beneficial effects for fish populations, with the caveat that anchored whole tree structures provide benefits long-term past the 2-year mark while wood bundle benefits tend to plateau after 1 to 2 years.

Bayesian modeling framework

Utilizing a Bayesian modeling framework for this study shows the valuable contribution models like this can provide for cases struggling with issues of pseudoreplication or estimation of parameters that would provide skewed inference in frequentist models. Results can be reported in a standardized fashion and different reporting guidelines, ensuring reproducibility and clarity, already exist and can be readily used (Rindskopf 2020). Many studies in Ecology face problems of scale and time dependency and sensitivity with both often being expensive or in some cases outright not feasible from a logistical standpoint (Lemoine et al. 2016; Laplanche et al 2019; Shaw et al. 2021). Treatment replicates are often low in cases of remote locations or when concerning endangered and threatened species or ecosystems while sampling and study design often introduces dependencies and pseudoreplication by default (Lemoine et al. 2016; Laplanche et al 2019; Shaw et al. 2021). The same applies for enhancement and restoration measures where a single river stretch or lake is targeted without adequate reference sites or replicates available, especially in remote areas like the northern boreal. Our study highlights the current and future use for a Bayesian approach when dealing with such issues while retaining informative values of observations in the form of posterior means and probabilities of treatment differences (Laplanche et al. 2019; Lazic et al. 2020).

4.1.5 Conclusions

Coarse woody habitat introduction in the littoral zone of shallow lakes has shown to be beneficial for multiple fish species typical for northern boreal lake systems, due to increased habitat use and food availability based on Bayesian model predictions (>90% probability after 2 years). Increases measured through CPUE/ CPUA predictions linked to enhancement types seem to level off over time for wood bundles with a high probability of over 75%, hinting at the temporal nature of wood bundle enhancements and their structural degradation over time. Wood bundle structures can provide fast benefits but do not seem viable for long-term use when built and anchored with biodegradable materials with structural integrity loss over time following an almost immediate onset of anchoring issues, days after construction. Anchored whole tree structures seem to keep their structural integrity longer and thus provide benefits over a longer time compared to submerged bundles. Coarse woody habitat offers an option for early-stage ecosystem enhancement, most viable for newly created lakes with low structural complexity or converted mine pits or reservoirs (McCullough & Van Etten 2011). Loss of structural integrity needs to be monitored frequently and CWH introduction should be conducted alongside potentially other measures that ensure long-term benefits or meet other requirements for target species not responding to CWH structures. Coarse woody habitat in northern boreal systems is readily available in the area in most cases and is cost efficient while being 100% degradable. Although heavily utilized for streams, CWH holds the potential to be used more frequently in lake systems and for species not yet considered. Utilizing a Bayesian modeling framework with non-informative priors in combination with spatial autocorrelation considerations seems to provide a feasible alternative to a frequentist approach and might be especially valuable for studies dealing with pseudoreplication concerns and treatment size issues, a more and more common issues in remote and impaired ecosystems or in cases of endangered species (Lemoine et al. 2016; Laplanche et al 2019; Shaw et al. 2021).

Chapter 4.2: Measuring beta diversity components and beneficial effects of coarse woody habitat introduction on invertebrate and macrophyte communities in a shallow northern boreal lake; implications for offsetting

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Summary

Structural habitat enhancement has been long established as a popular tool to counter habitat loss due from land-use and development. One enhancement approach is the introduction of Coarse Woody Habitat (CWH) to improve the establishment of macrophyte, macroinvertebrate, and fish communities. Here we test for the effects of CWH in Northern boreal lakes in the context of mitigation projects. We constructed Coarse Woody Habitat structures in a structure-less littoral zone of Lake Steepbank within the Oil Sands Region of Alberta, Canada. Enhancement structures featured increased macrophyte and invertebrate richness and biomass compared to reference sites and pre-treatment assessments over the course of three years. Enhanced sites also retained improved richness (macrophytes), diversity (macroinvertebrates) and biomass (both), despite STIN loss and degradation of enhancement structures over time. Using beta diversity components, constituting richness agreement, community differentiation and site relationships, and testing their relative importance revealed that replacement was more dominant for invertebrates and increasing similarity more important for macrophyte communities' post-enhancement. Our study shows the value of CWH addition for macroinvertebrate and macrophyte communities in what is otherwise a structure-less environment. Community changes over time showcase how beta diversity should be more strongly incorporated in restoration and enhancement studies to quantify community shifts that otherwise would not be captured in alternative diversity measures.

4.2.1 Introduction

Habitat loss and degradation is widely recognised as the main driver behind biodiversity loss in aquatic freshwater systems (Dudgeon et al. 2006; Reid et al. 2018). Landscape changes and habitat impairment due to anthropogenic development projects that display low natural recovery potential are especially hard to address. This has led to many countries adopting offsetting regulations and frameworks, such as the European Union Habitat Directive or the Canadian Fish Habitat Framework (Carreras Gamarra et al. 2018; Schoukens & Cliquet 2016). Here the offsetting frameworks can target different levels of equivalency, which is centered on the concept of No Net Loss (NNL) (Vaissière et al. 2017). No Net Loss can range from no net loss of ecosystem services and weak sustainability (e.g., Pearce & Atkinson 2017; resource or degraded ecosystem can be replaced with human substitute) to no net loss of biodiversity and strong sustainability (e.g., Daly 1991; natural resources or systems are not substitutable and complement to human-made services and capital).

Given global biodiversity declines in freshwater ecosystems (e.g., Dudgeon et al. 2006; Reid et al. 2018), the goal of these policy efforts is to prevent further species losses through the implementation of offsetting and conservation frameworks allowing for the translation of them to on-the-ground applications (Cowx & Portocarrero Aya 2011; Theis et al. 2019). A commonly used approach for enhancing aquatic habitat is the creation of offsetting structures using Coarse Woody Habitat (CWH). Coarse woody habitat provides structural enrichment and shelter provisioning, as well to create rearing and spawning habitat for many different species and taxa (Czarnecka 2015). Although extensively studied for lotic systems, knowledge on CWH benefits and feasibility for lentic systems, though increasing in application and case studies, is still scarce (Sass et al. 2019). Further, there is a dearth of knowledge with long-term monitoring and evaluation of structured experiments going beyond basic changes in population sizes, which are necessary to reduce key uncertainties (e.g., long-term benefits, appropriate success metrics, monitoring timeframes) when using CWH introductions in lentic systems (Sass et al. 2019).

Northern boreal lake ecosystems have become increasingly exposed to development stress through mining and natural resource extraction (Doig et al. 2015), where policy and regulations require proponents of a designated project to offset or compensate for habitat loss. Compensation requirements are based on specific target metrics (e.g., productivity) and long-term management. Coarse woody habitat structures are a tool that is being employed to increase habitat complexity in impoverished or naturally structure-less, for fish species and serves as a nutrient source in the littoral zones (Czanercka 2015). Coarse woody habitat serves as a growing surface for periphyton and degrading CWH provides a direct nutrient input in its vicinity for macrophyte and invertebrate communities and ultimately the whole lake through habitat coupling and food web pathways (Nelson et al. 2017; Smokorowski et al. 2006; Vadeboncoeur & Lodge 2000; Ziegler et al. 2017). Thus, given the structural and nutrient effects of CWH structures, it would be expected that CWH would have an impact on macrophyte and benthic macroinvertebrate communities (Czarnecka 2015; Helmus & Sass 2008; Sass et al. 2019; Smokorowski et al. 2006).

Although theoretically feasible in remote locations like the northern boreal due to availability of natural material, CWH structures are often overlooked since vegetation impact on lake condition is generally regarded as low in terms of deadwood input as well as water properties even when considering large scale ecosystem disturbances like fire regimes (Moser et al. 1998; Pienitz et al. 1997a). Macrophyte and macroinvertebrate changes are commonly assessed by species richness or abundance across samples or plots. Diversity and richness measures as well as abundance, even when standardized, only capture either alpha (local diversity) or gamma diversity (total diversity) and population trends. Although serving as an estimator for beneficial effects and effect size, alpha and gamma diversity are not informative enough when evaluating enhancement or restoration projects to determine driving factors in community changes (Cornell & Lawton 1992). Beta diversity bridges the two indicators by providing information on habitat partitioning based on the ratio of local and total species biodiversity turnover (‘turnover’ now ‘replacement’; Schmera et al. 2020; Wilson & Shmida 1984). Beta diversity in its many subsequent iterations captures presence and absence of rare species as well as species nestedness within a community (Baselga 2009). A newer approach takes the total diversity of sample sites or sampling units and splits them into *Similarity* (S), *Replacement* (R) and *Richness Difference* (D). This so-called SDR-simplex method allows identifying important community components, with *Replacement* and *Richness Difference* being considered as beta diversity, thus named beta diversity partitioning, *Replacement* and *Similarit*y indicating richness agreement and *Similarity* and *Richness Difference* describing nestedness (Figure 6b; Baselga 2009; Schmera et al. 2020).

In this study, we demonstrate the utility of said simplex method and its diversity components in evaluating the effectiveness and outcome of habitat enhancement projects that are widely used in aquatic habitat restoration and compensation projects. The main object of our study was to test for the beneficial effects of CWH on aquatic invertebrate and macrophyte communities through introducing CWH in the littoral zone of a northern boreal lake. We carried out a 3-year field study to record and test the response of present macrophyte and invertebrate species to CWH introduction in Lake Steepbank, a structure-less, shallow Lake in northern Alberta, Canada. Our objectives were to (1) test for CWH effects on macrophyte and invertebrate richness and biomass across two treatment types (*Clustered* vs *Spaced* distribution), (2) estimate long-term and potential temporal nature of CWH benefits against *Structural Integrity* (STIN) of

CWH structures exposed to abiotic factors, and (3) assess community changes and split diversity into its SDR-simplex method components to test for relative importance of *Similarity, Richness Difference*, and *Replacement* within and across treatments. We expect that CWH enhanced habitat will produce higher species richness and biomass for macrophyte and invertebrate communities due to habitat provision and increased food and nutrient availability.

4.2.2 Methods

Study System

Our study lake, Lake Steepbank, is a shallow (16-meter maximum depth) northern boreal lake in Lakeland County, Alberta, Canada (55° 28' 35.7" north, 111° 34' 23.8" west) only accessible through a former logging road (Figure 4.2.1). The fish community in the lake is comprised out of northern pike (*Esox lucius*), white sucker (*Catastomus commersonii*), spottail shiner (*Notropis hudsonius*) and brook stickleback (*Culaea Inconstans*). The aquatic invertebrate community in the lake was mostly made up of Molluscs, Amphipods from the *Gammaridae* family, and a few representatives of the *Odonata* order. A total of 12 families were identified in the collected samples and coincides with lake wide reference samples from August 2018 (Figure 4.2.1B; Table S4.2.1; Clifford 1991; reference comparison Lake Goodwin: families = 15; Lake Wappau: families = 19). Six submerged and floating macrophyte species were identified during the sampling process, matching samples from other parts of the lake (August 2018) belonging to the *Polygonaceae, Plantaginaceae, Characeae, Ruppiaceae, Hydrocharitaceae* and *Ceratophyllaceae* families (Figure 4.2.1C; Table S4.2.1; ALMS 2016; reference comparison Lake Goodwin: species = 9; Lake Wappau: species = 13). Lake Steepbank being un-impacted by development projects but naturally degraded in terms of fish population (managed through daily bag limits for Northern Pike) and structural richness compared to other lakes in the area (Lake Goodwin, Lake Wappau) serves as a reference system for artificially constructed lakes that are meant to offset negative environmental impacts due to oil surface mining and thus are structure-less or structure poor by design and can be enhanced through structure addition like CWH.

Enhancements & Treatments

Coarse woody habitat counts were conducted along the shoreline of the lake to identify structure-less areas, suitable for the enhancement treatments. 100 m transects (n = 26) were surveyed around the lake perimeter up to a water depth of 2 meters (Figure 4.2.1C; Table A4.2.1).

Recorded CWH was divided according to length and diameter (small: diameter 10-15 cm & 1.5 – 3 m total length, large: >15 cm diameter & > 3 m total length). We identified the northeastern bay of Lake Steepbank to have the lowest CWH count and corresponding low structural richness (0.2 CWH pieces small and large CWH/100m^2; reference comparison Lake Goodwin: 2.4 ± 1.8; n = 15; Lake Wappau: 2.7 ± 2.1; n = 15; Figure 4.2.1B). We divided the bay into three distinct areas with similar bathymetry, each with a different treatment type (*Spaced, Control,* and *Clustered*) (Figure 4.2.1D). For the corresponding CWH treatments, we used two different types of structures, whole tree drops and CWH bundles. We used local trees under a logging license and deadwood to construct the enhancements out of jack pine (*Pinus banksiana*), white spruce *(Picea glauca*) and aspen poplar (*Populus tremuloides michx*) (Wilkinson 2010). Different local tree species and a mix between felled trees and deadwood was chosen to optimize benefits for invertebrate diversity (Andringa et al. 2019). Coarse woody habitat bundles were made from eight 1.8 m logs with a minimum diameter of 10 cm and sunk at a depth of 1.5 to 2 meters, weighed down by biodegradable burlap sandbags. Each whole tree structure was assembled from two fully crowned trees, 4 to 6 m long trees into a V shape and anchored into the ground, creating half submerged calm water areas (Figure 4.2.1D). The *Spaced* zone received 5 CWH bundles and 5 whole tree structures, spaced 30 m apart, whereas the *Clustered* zone was spaced 15 m between enhancements (Figure 4.2.1D). A large structure-less *Control* area (400 m stretch) was left unenhanced separating the two main treatment areas (Figure 4.2.1D).

Invertebrate and macrophyte sampling and processing

Aquatic macroinvertebrates were sampled during 2018 - 2020 through kick-netting. Three composite kick-net samples, 5 replicates within 10 m radius composited into one sample to account for patchiness in distribution, (D-frame; 500 µm; 2 m^2 composited area in front of kick-net; 30 seconds interval per sample; substrate disturbance to a depth of ~5-10 cm; homogeneous habitat type) were randomly taken in each of the three treatment zones in August each year, except for 2018 in which four composite samples were taken from the whole area, prior to construction (n = 22 composite samples (110 individual samples); based on suggestions by National Aquatic Monitoring Center and Alberta Parks and Environment). Sampling was limited to the summer months due to weather (ice cover November to early May) and the remoteness of the study site as well as to avoid phenology bias (National Aquatic Monitoring Center), as well as acknowledging that detailed invertebrate seasonal dynamics would require higher temporal resolution of sampling. Invertebrates were preserved in 70% isopropyl alcohol (plastic vials with screw caps; Levi 1966) and identified within the laboratory to the family level and weighed to form proportionate biomass contribution per sample as well as presence frequency. Macrophytes were estimated through a sampling plot design, with plot sizes of 1 x 1m. Each plot was chosen randomly within each treatment and 12 plots were sampled per treatment per year with an additional 12 plots, *Pre-enhancement*, in 2018 (n = 84). Macrophytes were identified to the species level in the laboratory based on pulled sample specimens. Proportionate weight (e.g., species A = 35% of total biomass) was estimated through visually counting plant species per plot (digital control in ImageJ) combined with weighing representative sample specimen since clearing whole plots was deemed too high of a disturbance. We estimated species frequency through species presence-absence in a given plot and overall plot cover through a top-down photograph with reference grid using ImageJ according to the Braun-Blanquet method (Wikum & Shanholtzer 1978). Invertebrates as well as macrophytes frequency and biomass were recorded to test for different factors potentially playing into community changes over time since each metric captures a different driver (Moore & O'Sullivan 1970).

Frequency and cover analysis

Statistical analyses were done in *R Software* (4.0.5, R Core Team 2020). Frequency, diversity and richness and biomass dissimilarity and beta diversity were calculated, using the '*vegan*' package for diversity and cluster analysis (Oksanen et al. 2020; Table A4.2.3). Frequency distribution for invertebrates and macrophytes was calculated on sample level and totaled. Results were ranked from highest to lowest and plotted in line graphs to identify and visualize broad trends for invertebrates and macrophytes (Figure 4.2.2). Additionally, macrophyte cover per plot was estimated by field photographs in imageJ according to Braun-Blanquet. Accordingly, each macrophyte plot received a score of +, 1, 2, 3, 4 or 5 corresponding to cover ranges of <5 (few individuals), <5 (numerous individuals), 5-25, 25-50, 50-75 and 75 to 100% (Wikum & Shanholtzer 1978). Differences in cover scores were tested for through an Analysis of Variance (ANOVA;Welch for unequal variances; null hypothesis = no difference in plot cover between enhanced and unenhanced areas), followed by *post-hoc* tests (Games Howell; R Core Team 2020; accepted alpha of < 0.05; Tomarken & Serlin 1986).

Diversity and richness analysis

Changes in alpha diversity, defined as taxa richness at single locations, were quantified through the Shannon Index for invertebrates, using species abundance in combination with frequency, as absolute diversity, compared to the maximum diversity (Peet 1975). Alpha diversity for macrophyte plots was calculated through species richness (Peet 1975). Richness and diversity differences were compared across treatments and years through an ANOVA (ANOVA: Richness/ Diversity ~ Year:Treatment + Year + Treatment) followed by *post-hoc* tests (Games Howell) (R Core Team 2020; Shapiro Wilk test for normality; Levene's test for homogeneity of Variance; accepted alpha of < 0.05; Figure 4.2.3; null hypothesis = no difference in diversity or richness across time and treatment).

Biomass dissimilarity

For community composition analysis, biomass for invertebrates and macrophytes was transformed into distance matrices based on Bray-Curtis dissimilarity. Bray-Curtis dissimilarity is measured on a scale of 0 to 1, where 0 means that two samples have the same proportionate biomass composition (R Core Team 2020; vegan 2020; stats 2020). Average distance between species (macrophytes) and families (invertebrates) was used in a hierarchical cluster analysis and visualized in a heat-map to show biomass differences in proportionate biomass distribution across treatments and years (R Core Team 2020; stats 2020; Figure 4.2.4). Significant dissimilarity differences were tested for by a Permutational Multivariate Analysis of Variance (PERMANOVA; permutations = 999; accepted alpha of < 0.05 for Year and Treatment; null hypothesis = no difference in proportionate biomass across time and treatment) and pairwise multilevel comparison (Anderson 2001; R Core Team 2020; adonis 2016; Arbizu 2020; accepted alpha of < 0.05 for *Pre-enhanced; Clustered, Control* and *Spaced*). Calculated R-squared (R^2) values provide effect sizes for variation in distances based on grouping factors (e.g., Treatment, Year). For instance, in our case study an R-squared value of 0.31 for macrophyte biomass distribution within plots and the factor 'year', means that 31% of variation in distance is explained by year (adonis 2016).

Beta diversity and SDR-simplex components

Beta diversity for invertebrates and macrophytes, defined as taxa differences between communities or sites, was visualized through a Principal Coordinates Analysis (PCoA), based on Jaccard similarity distances (presence-absence data) (Figure 4.2.5; Anderson et al. 2006; R Core

Team 2020; vegan 2020). Centroid distances were used in a permutation test for PCoA dispersion (Permutations = 999, accepted alpha of < 0.05; null hypothesis = no difference in centroid distance between treatments) to test for significant treatment differences in beta diversity based on group variability, between enhanced and unenhanced sites (treatment). Pairwise *t*-tests were used to test for pairwise group dispersion differences (R Core Team 2020; vegan 2020; accepted alpha values 0.05; *Pre-enhancement; Control; Clustered; Spaced*). Beta diversity by itself can reflect two different drivers, species replacement (formerly 'turnover') and nestedness which are important to differentiate (Baselga 2009). Nestedness occurs when the species assemblage of a site represents a subset of the total ecosystem species richness (Ulrich & Gotelli 2007). These subsets may be due to a variety of different factors and represent non-random species loss over time, common in fragmented landscapes (Lourenco-de-Moraes et al. 2018). Replacement of species by others can be linked to biotic or environmental conditions in each habitat. Species sorting is a common phenomenon in patchy habitats with different environmental conditions that allow for dispersion (Leibold et al. 2004).

Beta diversity differences from the PCoA were used in the SDR-simplex analysis (Figure 4.2.6; simplex-SDR; Jaccard similarity; Podani et al. 2013; Schmera et al. 2020). The SDR-simplex approach allows an evaluation of the relative importance of the different beta diversity components, *Similarity* (S), *Richness Difference* (D) and *Replacement* (R), for within and between community changes. In our case we use it after testing for overall beta diversity differences through our PCoA to further test for if habitat enhancement has an impact on S, D or R importance compared to *Pre-enhancement* treatments. The commonly used simplex plot is a triangle with the corners being S, D and R, ranging from 0 to 1 based on Jaccard similarity and corresponding axes of nestedness, similarity agreement and turnover. The S, D and R corners represent extreme situations between sampling units (e.g., site; plot; sample). For instance, a point at the R corner represents sampling units with no species in common, meaning diversity differences within sampling units are 100% due to replacement. The S corner represents sampling units having the exact same species composition and the D corner represents perfect nestedness (Figure 4.2.6; Podani et al. 2013; Schmera et al. 2020). In our case this provides us with the importance of S, D and R between all unenhanced macrophyte plots but also allows us to compare the unenhanced to the enhanced treatment. Consequently, we tested the S, D, R differences for treatments (enhanced vs unenhanced) through an ANOVA with accepted alpha of

< 0.05; null hypothesis = no difference in S, D, R importance between treatments. Significant component differences were identified by paired *t*-tests (accepted alpha of < 0.05; individual S, D, R components across treatments) for macrophytes and invertebrates and converted into percentages, to evaluate the relative importance of *Similarity* (S), *Richness Difference* (D), and *Replacement* (R).

Temporal aspects for CWH benefits

Seasonal and temporal abiotic factors like wind, snow, and waves can influence STIN of CWH enhancement. To control for these influences, we recorded STIN of all enhancements through on shore or snorkel surveys 1 week, 1 month, 1 year and 2 years post construction. CWH degradation and structural changes or failure was recorded in one of five categories, corresponding to a percent score. Scores ranged from full integrity and original location to complete failure and relocation of structure over time (Table A4.2.2). No repairs or maintenance was done to the structures to allow for a natural progression of abiotic and biotic processes and capture the natural life cycle of CWHs in northern boreal lake systems without anthropogenic intervention. The nature of the constructed enhancements as well as the use of natural biodegradable substances was based on research license specifications. Furthermore, the need for estimating enhancement benefit provision over time has increased significantly over the past years in Canada and North America with development project proponents having to adhere to specific compensation targets and meeting perpetuity requirements (e.g., Bull et al 2013; Quigley & Harper 2006). Mean STIN in percent was observed over time (Kruskal Wallis rank sum test; accepted alpha of < 0.05; Figure A4.2.1; null hypotheses = no differences in structural degradation between and within structure types over time) together with richness/ diversity and biomass changes, as well as controlling for 'year' in all previously mentioned analyses steps.

4.2.3 Results

Frequency and Cover

Frequency ranks differed for invertebrates between enhanced and unenhanced sites. Although *Gammaridae*, *Planorbidae* and *Sphaeriidae* were the most frequent families in all samples, *Physidae*, *Aeshnidae*, *Lymnidae* and *Lestidae* became more frequent at enhanced sites. *Aeshnidae*, *Leptoceridae* and *Lestidae* were absent in *Pre-enhancement* samples and *Aeshnidae*, *Leptoceridae* and *Lymnidae* were absent in *Control* samples (Figure 4.2.2A). Macrophyte frequency for both enhanced and unenhanced was dominated by *Chara vulgaris* and *Rupiah cirrhosis*. However, unenhanced plots lacked *Pericardia amphibian Linnaeus* and *Ceratophyllum demersum*, frequently present in enhanced sites (Figure 4.2.2B). Relative macrophyte cover as estimated by the Braun-Blanquet method differed significantly across treatments ($p < 0.05$; $n = 84$; $F = 2.81$; $dfn = 6$). Average plot cover in the *Spaced* treatment was 18 ± 16.9% and significantly higher than in the *Control* and the *Pre-enhancement* treatment with 6.3 ± 8.9% and 6.7 ± 11% of average plot cover (Table S4.2.2; $p < 0.05$). Relative cover was also higher, though non-significant, in the *Clustered* treatment compared the *Control* and the *Pre-enhancement* treatment, with an average plot cover of 16.4 ± 14.9% (Table S4.2.2; $p = 0.057$).

Diversity and richness

Macroinvertebrate diversity, calculated through Shannon's H, changed significantly over time (Figure 4.2.3A; Table S4.2.3; $n = 22$; $df = 2$; $F = 13.209$; $p < 0.001$), with mean diversity increasing significantly from *Pre-enhancement* diversity of 0.41 ± 0.21 in 2018, to 0.95 ± 0.55 in 2019 ($p < 0.05$) and 0.98 ± 0.5 in 2020 (Figure 4.2.3A; Table S4.2.3; $p < 0.001$). Furthermore, Shannon's H for macroinvertebrates differed significantly between treatments (Figure 4.2.3A; Table S4.2.3; $n = 22$; $df = 2$; $F = 52.734$; $p < 0.001$). Mean Shannon's H was 1.28 ± 0.17 for the *Spaced* treatment, marking a significant increase compared to *Pre-enhancement* diversity (0.41 ± 0.21; $p < 0.001$) and *Control* treatment diversity (Figure 4.2.3A; Table S4.2.3; 0.35 ± 0.05; $p < 0.001$). Macroinvertebrate diversity of 1.23 ± 0.13 for the *Clustered* treatment was also significantly higher than *Pre-enhancement* diversity ($p < 0.001$) and *Control* treatment diversity (Figure 4.2.3A; Table S4.2.3; $p < 0.001$). Invertebrate diversity did not differ between *Spaced* and *Control* treatment ($p = 0.966$) as well as between *Pre-enhancement* and *Control* treatment (Figure 4.2.3A; Table S4.2.3; $p = 0.756$). Interactions for Shannon's H between year and treatment were non-significant (Figure 4.2.3A; Table S4.2.3; $n = 22$; $df = 2$; $F = 0.384$; $p = 0.687$).

Macrophyte diversity, calculated through Richness, was significantly different between enhanced (*Spaced* & *Clustered*) and unenhanced (*Pre-enhancement* & *Control*) but not for individual treatments (e.g., *Spaced*, *Clustered*, *Pre-enhancement*, *Control*; Figure 4.2.3B; Table S4.2.4; $n = 84$; $df = 1$; $F = 9.359$; $p < 0.05$). Mean macrophyte richness in enhanced plots was one species higher (3.00 ± 1.58) compared to unenhanced plots with an average macrophyte richness of 2.02 ± 0.56 (Figure 4.2.3B; Table S4.2.4). Interactions for macrophyte richness between year and treatment were non-significant (Figure 4.2.3B; Table S4.2.4; $n = 22$; $df = 1$; $F = 0.186$; $p = 0.668$) as well as year as a factor by itself (Figure 4.2.3B; Table S4.2.4; $n = 22$; $df = 1$; $F = 1.655$; $p = 0.202$).

Biomass

Proportionate biomass distribution for macroinvertebrates was significantly different across treatments with treatment explaining 58% of the variance (Table S4.2.5; $n = 22$; $df = 3$; $F = 8.469$; $p < 0.001$). Biomass distribution for invertebrates was significantly different for both enhanced treatments compared to the two unenhanced ones (Table S4.2.5; $p < 0.05$). Neither enhanced (*Spaced* vs *Clustered*) nor unenhanced treatments (*Pre-enhancement* vs *Control*) differed from each other (Table S4.2.5; $p = 0.363$; *0.568*). Biomass distribution in samples from unenhanced areas was dominated by *Gammaridae,* accounting for 79.63% of biomass in the *Control* area and 55.22% of biomass in *Pre-enhancement* samples from 2018 (Figure 4.2.4A; Table S4.2.5). Other noticeable contributions to biomass in *Control* samples were from the *Gordiidae* (7.99%), *Sphaeriidae* (5.65%) and *Planorbidae* (3.68%) families. Other families like *Baetidae, Helicopsychidae, Physidae, Lestidae* and *Lepidostomatidae* contributed less than < 2% of biomass or were absent (*Aeshnidae, Leptoceridae, Lymnaeidae;* Figure 4.2.4A; Table S4.2.5). *Pre-enhancement* samples were similar in that regard with the exception that *Lymnaeidae* were present, accounting for 22.63 % of the biomass (Figure 4.2.4A; Table S4.2.5). All 12 recorded families were present and contributing to overall biomass in samples from the enhanced areas (*Spaced* and *Clustered*). *Gammaridae* biomass contribution compared to the unenhanced samples was reduced to 16.23% in the *Spaced* treatment and 11.07% in the *Clustered* treatment. Specimens from the *Aeshnidae* family, previously absent in samples, made up 17.94% of biomass in the *Spaced* treatment and 9.65% in the *Clustered* treatment. The biggest increase was observed for *Lymnaeidae* biomass with 42.46% of biomass contribution in the samples from the *Spaced* treatment and 61.33% of biomass contribution in samples from the *Clustered* area. Biomass for other families was comparable to the unenhanced treatments except for a reduction in *Planorbidae* proportionate biomass from around 3% to less than 1.5% and *Gordiidae* from around 8% in unenhanced samples to 3% in samples from the enhanced areas (SI 5). Year as a factor only explained 5% of biomass variance (Table S4.2.5; $n = 22$; $df = 1$; $F = 2.179$; $p <$

0.129) and interactions of treatment and year explained only 3% of biomass variance for macroinvertebrates (Table S4.2.5; n = 22; df = 2; F = 0.737; $p = 0.57$). Overall, *Amphipoda* biomass dominance in the *Control* and *Pre-enhancement* treatments was replaced by a higher biomass contribution by *Odonata* and *Basommatophora* while time in years and time and treatment interactions were negligible drivers for changes in biomass distribution.

Proportionate biomass distribution for macrophytes was significantly different across treatments with treatment explaining 30% of the variance (Table S4.2.6; n = 84; df = 3; F = 9.736; $p < 0.001$). Biomass distribution for macrophytes was significantly different for plots in the *Spaced* and *Clustered* treatment compared to plots in the *Control* area and from *Pre-enhancemen*t assessments in 2018 (Table S4.2.6; $p = 0.001$). Macrophyte plots from the *Spaced* and *Clustered* treatments and their biomass distribution did not differ from each other (Table S4.2.6; $p = 0.096$). Similar results were recorded for comparing biomass distribution in *Pre-enhancement* plots to *Control* plots (Table S4.2.6; $p = 0.093$). Macrophyte biomass proportionate distribution in samples from unenhanced areas was dominated by *Chara vulgaris* and *Ruppia cirrhosa,* with 49.03% and 44.60% of biomass per plot in the *Control* area and 61.19% and 27.16% of biomass per plot contribution in *Pre-enhancement* plots (Figure 4.2.4B; Table S4.2.6). *Elodea canadensis* and *Hippus vulgaris* made up around 3 to 6% of biomass in both unenhanced treatments and *Ceratophyllum demersum* and *Persicaria amphibia L.* were not recorded in any unenhanced plots (Figure Table4.2.4B; Table S4.2.6). All 6 on Lake Steepbank recorded species were present and contributing to overall biomass in samples from the enhanced areas (*Spaced* and *Clustered*). *Chara vulgaris* biomass contribution in comparison to the unenhanced samples was reduced to 17.80% in the *Spaced* treatment and 25.99% in the *Clustered* treatment while *Ruppia cirrhosa* biomass proportions per plot decreased less noticeable to 22.18% in the *Spaced* treatment and 20.56% in the *Clustered* treatment (Figure 4.2.4B; Table S4.2.7). *Persicaria amphibia L.* and *Ceratophyllum demersum* were present in enhanced plots and contributed to the overall biomass distribution with 45.58% for *Persicaria amphibia L.* and 6.41% for *Ceratophyllum demersum* in the *Spaced* treatment and with 43.49% for *Persicaria amphibia L.* and 5.86% for *Ceratophyllum demersum* in the *Clustered* treatment (Figure 4.2.4B; Table S4.2.7). *Elodea canadensis* and its biomass contribution stayed around 3 to 6% across treatments. *Hippus vulgaris* decreased in proportionate biomass across treatments, from around 2.5 to 5% in the unenhanced areas to 0.5 to 2.5% in the enhanced plots (Figure 4.2.4B; Table

S4.2.7). Overall, smaller submerged grass-like plants in the unenhanced areas were replaced in terms of biomass contribution and dominance by a higher biomass presence of floating and emergent species.

Beta diversity and turnover

Beta diversity differed across treatments for macroinvertebrates as measured by the permutation test for dispersion based on differences in centroid distance from the PCoA. Differences in centroid distance separated Beta diversity for the enhanced treatment from Beta diversity for the unenhanced treatments (Figure 4.2.5A; Table S4.2.7; $n = 22$; $df = 3$; $F = 4.01$; $p < 0.05$). Beta diversity based on PCoA centroid distances did not differ significantly across the four macrophyte treatments (Figure 4.2.5B; Table S4.2.7; $n = 74$; $df = 3$; $F = 0.982$; $p = 0.339$). Breaking down these community differences and into their respective SDR components suggested *Replacement* (R) as the main driver for invertebrate communities in the study lake (60.6%; Figure 4.2.6A; Table S4.2.8). *Replacement* (R) was more important in enhanced sites (64.3%) compared to unenhanced ones (56.9%; $n = 22$; $df = 1$; $F = 293.2$; $p < 0.001$). *Similarity* (S) (20.7%) and *Richness Difference* (D) (22.3%) were more important in unenhanced sites compared to enhanced sites (S = 18.2%; D = 17.5%; Figure 4.2.6A; Table S4.2.8; $n = 22$; $df = 1$; $F = 11.43$ & 7.131; $p < 0.05$). Macrophyte differences between plots in Lake Steepbank were mostly driven by replacement (44.3%). *Replacement* (R) importance decreased from unenhanced (47.9%) to enhanced sites (40.7%; $n = 74$; $df = 1$; $F = 33.27$; $p < 0.001$) while *Similarity* (S) and *Richness Difference* (D) increased significantly in terms of relative importance by 5.1 and 2.1% (Figure 4.2.6A; Table S4.2.8; $n = 74$; $df = 1$; $F = 95.72$ & 105; $p < 0.001$). Overall, Beta diversity components and their relative importance differed significantly for invertebrates and macrophytes when comparing enhanced and unenhanced sites as well as within treatments. Increased *Replacement* (R) rates in enhanced areas was the main driver for changes in invertebrate communities when compared to unenhanced communities. Changes in macrophyte communities between enhanced and unenhanced treatments were mainly driven by an increasing *Similarity* (S) between plots and reduced *Replacement* (R) rates.

Structural Integrity of Coarse Woody Habitat over time

The structural assessment showed that the whole tree enhancements kept their STIN longer compared to the CWH bundles (Figure A4.2.1). *Structural Integrity* was high 1 week and 1-

month post construction (over 80% for both enhancement types). However, the bundles lost integrity faster, with a mean integrity score of 65% after 12 months compared to the whole tree structures (90% STIN after 12 months; $n = 10$; $df = 3$; chi-square = 33.936; $p < 0.001$; Table S4.2.9; Figure A4.2.1). Two years post construction, the wood bundles were around or below 30% STIN with the logs spread out across the immediate area (Figure A4.2.1A). Seven out of the 10 whole tree structures were in place after 2 years with some STIN loss becoming only apparent after 2 years (Table S4.2.9; Figure A4.2.1A; $n = 10$; $df = 3$; chi-square = 27.176; $p < 0.001$). *Structural Integrity* between the two enhancement types differed greatly with anchored trees staying structurally intact to a large degree. Structural failure of the enhancements over time did not coincide with a reduction in macrophyte and invertebrate richness, diversity and biomass as controlled for in the models through 'year' as a factor (Figure A4.2.1B). Increases in biomass, richness and diversity were measurable one year post construction but did not differ between 2019 and 2020, having reached a plateau (Figure A4.2.1B). Overall, benefits of CWH structures for macrophyte and invertebrate communities in Lake Steepbank, captured by richness, diversity, and biomass, were retained throughout time while CWH bundles experienced large-scale structural failure and whole tree treatments stayed stable (Figure A4.2.1).

4.2.4 Discussion

Overall, we were able to provide an assessment of the value of CWH in supporting benthic macroinvertebrate and macrophyte communities in what is otherwise a structure-less environment. In particular the utility of CWH as a habitat enhancement was indicated by, (1) CWH enhancements improved macrophyte and invertebrate richness and biomass regardless of CWH spacing (*Clustered* vs. *Spaced*), (2) CWH structures retained improved richness, diversity and biomass, despite the reduction of STIN for log bundles, and (3) using beta diversity components and assessing their relative importance revealed that *Replacement* was more dominant for invertebrates and increasing *Similarity* more important for macrophyte communities post-enhancement. Thus, macrophyte and invertebrate biomass and frequency clearly responded to the introduction of CWH structures over the study period. Due to the similarity of results between the two treatments (*Spaced* and *Clustered*) for here on we aggregate and labelled the two treatments into a single category of 'enhanced' to facilitate discussion. Importantly the effectiveness of CWH as a habitat enhancement to attract organisms at the base of the food web provides promise for this as a potential tool to increase productivity in offsetting practices, like habitat restoration, enhancement, or creation, in lake ecosystems.

Responses of invertebrate and macrophyte communities to CWH

Macrophyte richness and invertebrate diversity increased in the enhanced areas compared to the *Control* and *Pre-enhancement* areas. Higher diversity and species richness support beneficial effects of CWH in providing habitat and essential nutrients. Coarse wood creates habitat for macrophytes and invertebrates, with increased macrophyte cover over time, as measured per plot, further contributes to overall invertebrate diversity and biomass as often observed for sandy and structure-less aquatic ecosystems (e.g., Lusardi et al. 2018; Shupryt & Stelzer 2009). This positive feedback loop emphasizes the various relationships influenced by habitat enhancement and its potential to provide benefits on different ecosystem levels. For invertebrates, unenhanced areas have *Aeshnidae*, *Leptoceridae*, *Lestidae* almost completely absent. These three families covering caddisfly, dragonfly and damselfly larvae all rely on aquatic vegetation and woody habitat as rearing habitat and a food source. For instance, *Polycentropodidae* or tube-making caddisflies, can construct their protective tube from small wood pieces (Wiggins 2005). Another example are dragonfly larvae preying on other invertebrate species or even small-bodied fish species or fish fry, potentially benefiting from the overall increase in invertebrate biomass and observed Spottail Shiner abundance (Benke 1976; Theis et al. unpublished). The presence of *Persicaria amphibia L.,* as an emergent, tall growing macrophyte species, absent in *Control* areas, would support the presence of the three species. Overall greater leaf dissection of submerged aquatic plants can be associated with a higher invertebrate diversity and abundance (Rosine 1955). Increased biomass and frequency of snails from the *Physidae* and *Lymnidae* families*,* compared to the unenhanced areas, supports a similar benefit of the CWH enhancements as mentioned for dragonflies, damselflies, and caddisflies. *Physidae* and *Lymnidae* species prefer to attach their eggs on aquatic macrophytes, preferably leafy plants with larger surface area. Furthermore, periphyton growing on CWH surface areas provides grazing opportunities for both species (Olsen et al. 2007). Similar benefits can be expected for *Planorbidae* and *Gammaridae* despite being equally frequent in *Control* and *Pre-enhancement* samples, likely due to their high tolerance for environmental conditions and generalist nature (Barbosa & Barbosa 1994; Gunnill 1982). *Sphaeriidae* are small generalist filter feeding mussel species that occur in large abundance in various substrates and thus may not rely on habitat enhancements (no difference in frequency and biomass across treatments) but may still benefit from enhancement efforts trough nutrient enrichment of substrate and water through plant detritus and degrading woody material (Watson & Ormerod 2005). Other invertebrate species present in all treatments, like mayflies (*Baetidae*) or snail-case caddisflies (*Helicopsychidae*) increased in their frequency across enhanced treatments, indicating similar benefits for other more generalist species. For instance, mayfly larvae, utilizing a large variety of habitats and being strong swimmers potentially benefited from an increased algae availability on CWH structures, the same way snail-case caddisflies larvae would feed on periphyton (Vaughn 1986). Overall, enhanced, and unenhanced areas feature distinct communities for invertebrates and macrophytes as supported by frequency, richness, diversity, and biomass data and shown in the results of the PCoA for beta-diversity.

Though differences were non-significant for macrophytes in terms of centroid distance, differences in macrophyte frequency and biomass, especially for *Persicaria amphibia L.* seem to suggest community changes pre and post enhancement. CWH enhancements in Lake Steepbank shifted in invertebrate community composition by allowing more specialized species (mainly Arthropods and Molluscs) to populate the area. These changes indicate a greater niche availability through habitat creation and resource diversity, potentially enhanced by the presence of emergent, leafed aquatic vegetation (Sánchez-Hernández et al. 2020). Diversity and biomass changes for both groups hold important food-web implications and could benefit the overall lake, in terms of fish species and terrestrial species (Francis et al. 2011; Schindler & Scheuerell 2002; Theis et al. unpublished). The rapid change in macrophyte communities was likely due to the creation of wave resistant zones in combination with an increased input and retention of organic material along the structures (Horvath 2004). Wind movement patterns on Lake Steepbank often move towards the Northeast section of the lake where the enhancements were placed, representing the maximum fetch length for the lake (Håkanson 1977). The added structures, especially whole tree drops accumulated and retained organic material as early as 1 week post construction. When using CWH structures in the context of offsetting, it is important to acknowledge the versatility of its benefits on an ecosystem scale, where newly created and often structure-less habitat may greatly benefit (Ruppert et al. 2018).

Diversity drivers in invertebrate and macrophyte communities

Invertebrate beta diversity changes based on frequency data were mainly driven by species *Replacement* (R) when partitioning diversity changes into the three components species *Similarity*, species *Replacement* and *Richness Difference* (S, D, R). Species *Replacemen*t became even more prominent in enhanced sites with a relative importance of over 64%. The dominance of species *Replacement* indicates that species substitution takes place in cases of changing or newly colonized ecosystems (Whittaker 1972). These changes reflect the coenocline between enhanced and unenhanced areas (Gauch & Whittaker 1972). As previously discussed, this most likely derived from the newly provided habitat and food sources creating niche diversity. Our findings are important to consider because they highlight the significant ecosystem changes that CWH enhancements can introduce, increasing invertebrate community turnover rather than overall richness within and between areas. Lower *Similarity* values further show that increased habitat heterogeneity in enhanced areas is supporting higher biodiversity (Astorga et al. 2014; Pik et al. 2002). Invertebrate community composition and beta diversity changes have often been attributed to habitat heterogeneity and serve as heterogeneity indicator even for small scale projects like Lake Steepbank (Astorga et al. 2014; Pik et al. 2002). Our results demonstrate that invertebrate beta diversity changes are related to species *Replacement*, supporting findings from other studies that habitat heterogeneity most often drives invertebrate diversity, making CWH enhancements a useful restoration and enhancement tool for increasing biodiversity within a system.

Interestingly, beta diversity changes for macrophyte communities were driven more equally by all three components and became more balanced in enhanced areas with species *Similarity* and *Richness Difference* becoming more important and a decreased contribution of species *Replacement*. According to the literature, macrophyte beta diversity changes usually follow similar patterns as observed for invertebrates, making habitat heterogeneity the most prominent diversity driver (Alahuhta et al. 2017). A reduction in species *Replacement* as the main driver can be explained by the recorded plot frequency of the individual macrophyte species. *Pre-enhancement* and *Control* plots, when vegetated, were largely dominated by *Chara vulgaris* and *Ruppia cirrhosa*, with only a few plots containing *Elodea canadensis* and *Hippus vulgaris*. This observation likely led to larger differences among plots, increasing *Replacement* and reducing *Similarity* while having overall low *Richness Difference* due to an overall low macrophyte richness in the unenhanced areas. *Chara vulgaris* and *Ruppia cirrhosa* were not replaced in enhanced plots but retained their high plot frequency while overall diversity increased through the new occurrence of *Persicaria amphibia L.* and *Ceratophyllum demersum*. Consequently, this increased overall species richness, but limited *Replacement* rates, thus increasing the relative importance of plot *Similarity* and *Richness Difference* for beta diversity changes. Low turnover rates for macrophyte species with increased nutrient input indicates low competition. A low competition scenario is also likely due to the overall low plot cover in *Pre-enhancement* and *Control* sites (< 10%), adding emergent macrophytes to the pre-existing submergent species (Duarte et al. 1986).

Beta diversity in its two main descriptors (directional; *Replacement* across sampling units and non-directional; variation among sampling units) is considered a key metric for assessing community diversity changes, dispersal capabilities, competition as well as environmental heterogeneity (e.g., Bennett & Gilbert 2015; Legendre & De Cáceres 2013). Spatial differentiation makes beta diversity a more insightful measure compared to alpha diversity when it comes to evaluating habitat enhancement and restoration projects in detail. The consensus for alpha diversity (e.g., richness) links back to the perception that more species are desirable in terms of higher biodiversity in a community which does not account for species abundance (Hurlbert 1971; Purvis & Hector 2000). Other metrics like Shannon's H and derived evenness try to account for species dominance and uncertainty (Purvis & Hector 2000). However, both descriptors do not account for species turnover. An enhanced littoral zone could display similar richness values pre- and post-enhancement, while not capturing that species turnover has shifted communities over time. In that regard beta diversity and especially the extension towards SDR-simplex can help inform conservation decision making by measuring community and ecosystem changes over time and their respective drivers, especially when considering changes in habitat heterogeneity (Baselga 2009; Palmer et al. 2010; Socolar et al. 2015). For instance, high beta diversity differences across sites can point to connectivity issues in patchy habitats (Lourenco-de-Moraes et al. 2018; Socolar et al. 2015). Furthermore, beta diversity can link the gap between local-scale alpha diversity and landscape level gamma diversity to better conserve overall global biodiversity (Chisholm et al. 2011).

In our study we were able to capture the invertebrate community shift, post enhancement through beta diversity changes linked to increased *Replacement* rates. Habitat enhancement or restoration generally assumes increased habitat heterogeneity (Cramer & Willig 2005), where the added heterogeneity is reflected in the higher importance of *Replacement* rates over time compared to *Richness Difference* and *Similarity*. This emphasizes the importance of *Replacement* for invertebrates as the desired driver for beta diversity in enhancement and restoration projects (Gauch & Whitaker 1972; Viana et al. 2015; Whitaker 1972).

Replacement should generally be regarded as a similarly desirable driver for macrophyte beta diversity changes in enhancement projects, with macrophytes usually following similar patterns as invertebrates due to habitat heterogeneity (Alahuhta et al. 2017). Our counter intuitive results were likely due to overall increased plot cover and a more even species distribution which in our case of a cover and biomass poor littoral macrophyte community is an actual desired outcome with species and cover slowly filling in. Our findings highlight how beta diversity and its components in combination with cover, biomass or frequency metrics can help identify colonization processes and community changes in better detail.

CWH integrity over time and in a regional context

The temporal nature and structural degradation of CWH enhancements over time did not seem to influence macrophytes and invertebrates two years post construction but could potentially change in the subsequent years. The sustained benefits in the first two years despite structural degradation showcase the difference between species responses to CWH enhancements. Invertebrates and macrophytes more often rely on the overall CWH presence but not necessarily the STIN compared to fish species, requiring proper structures as habitat and shelter (Haase et al. 2012). A decrease in fish abundance in enhanced areas over time in unison with structural failure of enhancements should also have long-term effects on invertebrate community composition (Gilinsky 1984). In most cases, top-down pressure exerted by benthi-planktivorous fish is reduced in shallow lakes with high macrophyte cover, another benefit of CWH enhancements (Jeppesen et al. 1997). Vice versa, grazing pressure e.g., through the increased abundance of white suckers in the area (Theis et al. unpublished) could change community structure over time (Jeppesen et al. 1997). However, anchored tree structures tend to retain their STIN longer than CWH bundles and should be preferred when aiming at long-term increases in spatial heterogeneity for newly created lakes or mitigation projects, especially when considering organic matter retention and the creation of wave resistant zones as well as the regional context of the northern boreal and the potential use of CWH for mitigation efforts (Czanercka 2015; Vadeboncoeur & Lodge 2000).

The wood regime for northern boreal lakes shows significant differences compared to other systems when looking at the individual regime components, wood magnitude (relative or absolute volume of recruited wood material), frequency (how often is wood material recruited), duration (length of wood recruitment events), timing (when wood is recruited), rate (mass or volume of wood per unit of time) and mode (process by which wood is recruited) as part of the wood regime of northern boreal lakes (Gennaretti et al. 2013; Wohl et al. 2019). Northern boreal lakes are generally characterized by low magnitudes of wood influx in infrequent intervals with unpredictable timing due to disturbances mostly delivered through falling of trees along the riparian areas in combination with biotic addition though beavers (Gennaretti et al. 2013; Hood & Larson 2014). Acknowledging differences in natural woody habitat availability in different parts of lake systems due to orientation, wind and geomorphology and woody habitat having a long residence time in northern boreal lakes (spanning 1400 years for black spruce) makes CWH introductions a matter of fixing woody habitat in place rather than just simply introducing it. (Gennaretti et al. 2013; Wohl et al. 2019). Long term STIN becomes even more important regarding the disturbance exposure of northern boreal lakes (Moser et al. 1998; Gennaretti et al. 2013). Many lakes, including our Study Lake and the overall region, experience regular fire regimes, posing large scale disturbances that can impact lake systems for centuries and reduce overall wood availability and influence woody habitat recruitment in major ways (Gennaretti et al. 2013). Furthermore, extreme and changing wind patterns in combination with low riparian structural richness often prevent organic matter or woody habitat retention, especially for medium and smaller structures (Moser et al. 1998). Extreme conditions like fire regimes, long winters and changing wind patterns in combination with low woody habitat recruitment (5.8 pieces per century; Gennaretti et al. 2013) pose significant hurdles for mitigation projects in the northern boreal since specially newly created system cannot rely on natural recruitment timeframes given compliance and project requirements that are often less than 10 years. Sustainability and long-term management of restoration and offsetting projects is a major concern to meet required NNL and equivalency goals in perpetuity (Ruppert et al. 2018). Many offsets in northern boreal systems aiming at fisheries productivity could lose their benefits over time and given that there are structural issues identified here with CWHs, more long-term studies are required to accurately estimate the temporal nature of CWH benefits for invertebrates and macrophytes. Our results underline the importance to identify suitable areas in lake systems maximizing fetch and ensuring long-term structural retention and colonization through on shore anchors, to avoid wood accumulation in deep bays missing their initial target.

4.2.5 Conclusions

In summary, CWH enhancements for shallow northern boreal lakes seem to be beneficial for invertebrate as well as macrophyte diversity and biomass, due to habitat heterogeneity, food/nutrient availability, and organic matter accumulation (Czarnecka 2015; Sass et al. 2019; Smokorowski et al. 2006). More specialized invertebrate species like dragonfly, damselfly and caddisfly larvae or snail species utilizing leaves for egg deposition seem to benefit from the enhancements and the presence of emergent aquatic vegetation (e.g., Benke 1976; Rosine 1955; Wiggins 2005).

Here we find that beta diversity increased for macrophytes and invertebrates in CWH treatment areas compared to reference sites. Beta diversity based on frequency data and associated change across CWH enhancements in Lake Steepbank were mostly driven by species *Replacement* for invertebrates due to habitat heterogeneity while macrophyte community changes could be traced back to an overall increase in species richness due to two previously absent species and plot *Similarity* with low *Replacement* rates. Beta diversity in its many iterations and newer developments (e.g., Zeta diversity) should be more strongly incorporated in restoration and enhancement studies to quantify community shifts that otherwise would not be captured in alpha diversity measurements and help inform conservation decisions (Hui & McGeoch 2014). Here beta diversity captures more information related to underlying community processes and can capture *Replacement*, *Richness Difference* and *Similarity* for sampling units and sites in northern boreal lakes with lower overall diversity and small treatment areas.

Overall, CWH enhancements are a useful tool to restore or enhance structure-less littoral areas in newly created lakes or degraded lakes for macrophyte and invertebrate communities acknowledging the low woody habitat recruitment rates of northern boreal lakes and exposure to extreme environmental regimes (Gennaretti et al. 2013; McCullough & Van Etten 2011). Using long-term monitoring ensures the persistence of benefits after structural degradation, either through natural processes or failure of CWH structures can set in relatively fast. Suitable areas in boreal lake systems maximizing fetch should be identified and anchored whole tree structures should be preferred over wood bundles to ensure long-term structural retention and colonization. Although underutilized in lentic systems, CWH enhancements provide a unique, affordable, and environmentally sustainable way of enhancing littoral areas and should especially be considered for the creation and management process of newly created compensation lakes or converted open pit mines in northern regions (Gammons et al. 2009). Ultimately, the application of CWH enhancements present a viable early-stage tool to aid proponents in achieving outcomes that are outlined in compensation and offsetting policies and regulations.

Chapter 4: Figures and Tables

Chapter 4.1 Figures

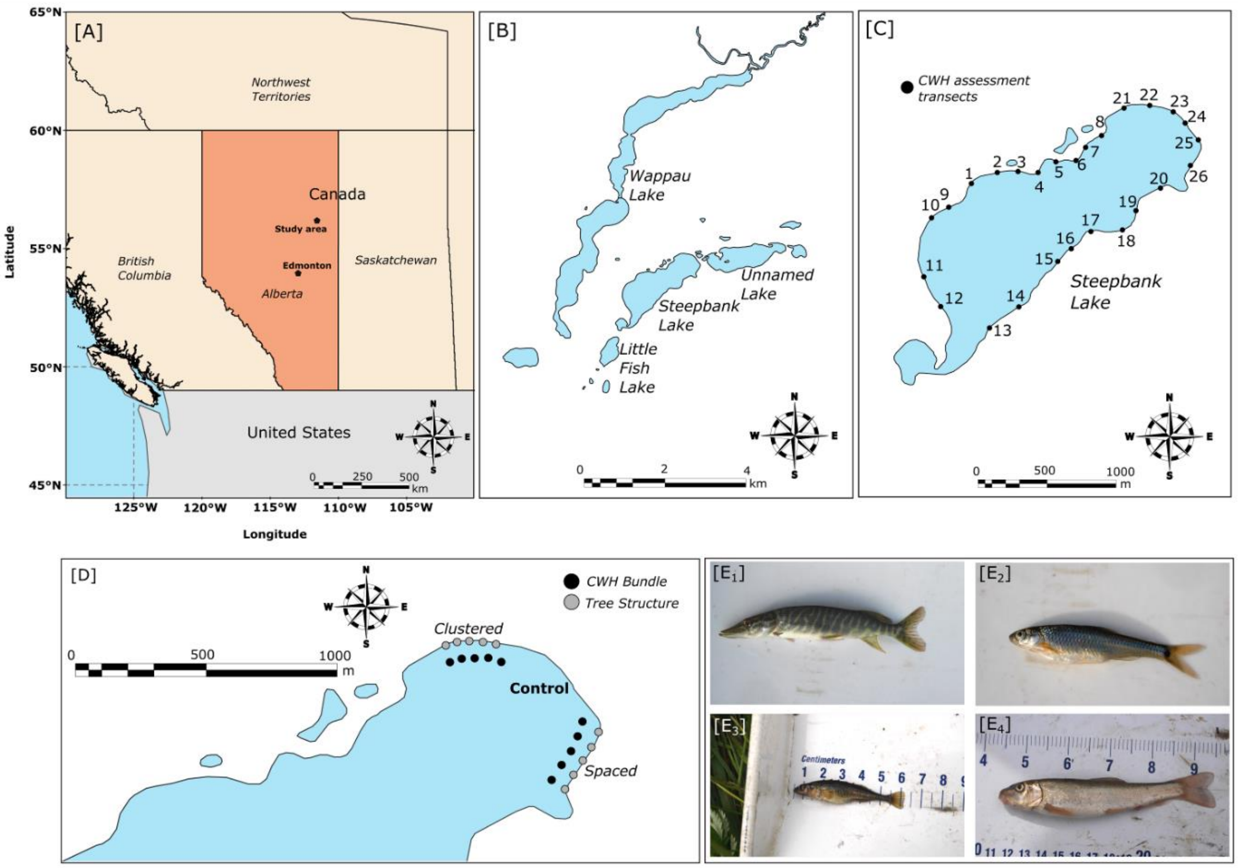

__Figure 4.1.1.__ Study lake (Steepbank Lake) for the coarse woody habitat (CWH) study located in Lakeland County, northern Alberta (A; B). Locations along the shoreline for assessment of structural richness (C, Table A1) as well as placement of CWH enhancements in the two treatments (D) and species composition (spottail shiner, Notropis hudsonius (E2), northern pike, Esox Lucius (E1), white sucker, Catostomus commersonii (E4), brook stickleback, Culaea inconstans (E3)) of Steepbank Lake

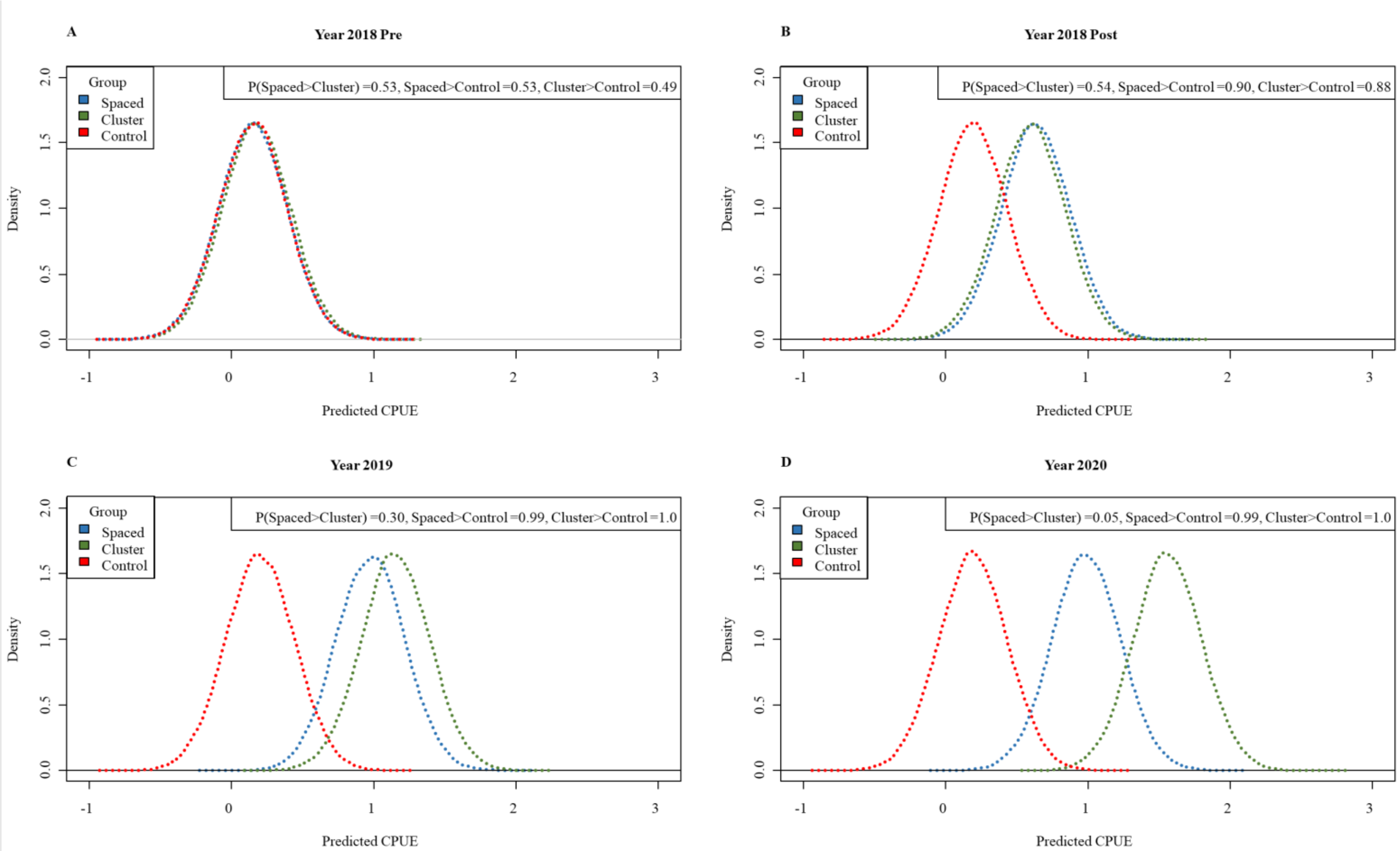

Figure 4.1.2*. Catch per unit effort (CPUE) per 12-hour minnow traps density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication (Table A6) as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 3; Delta = 0.9999.*

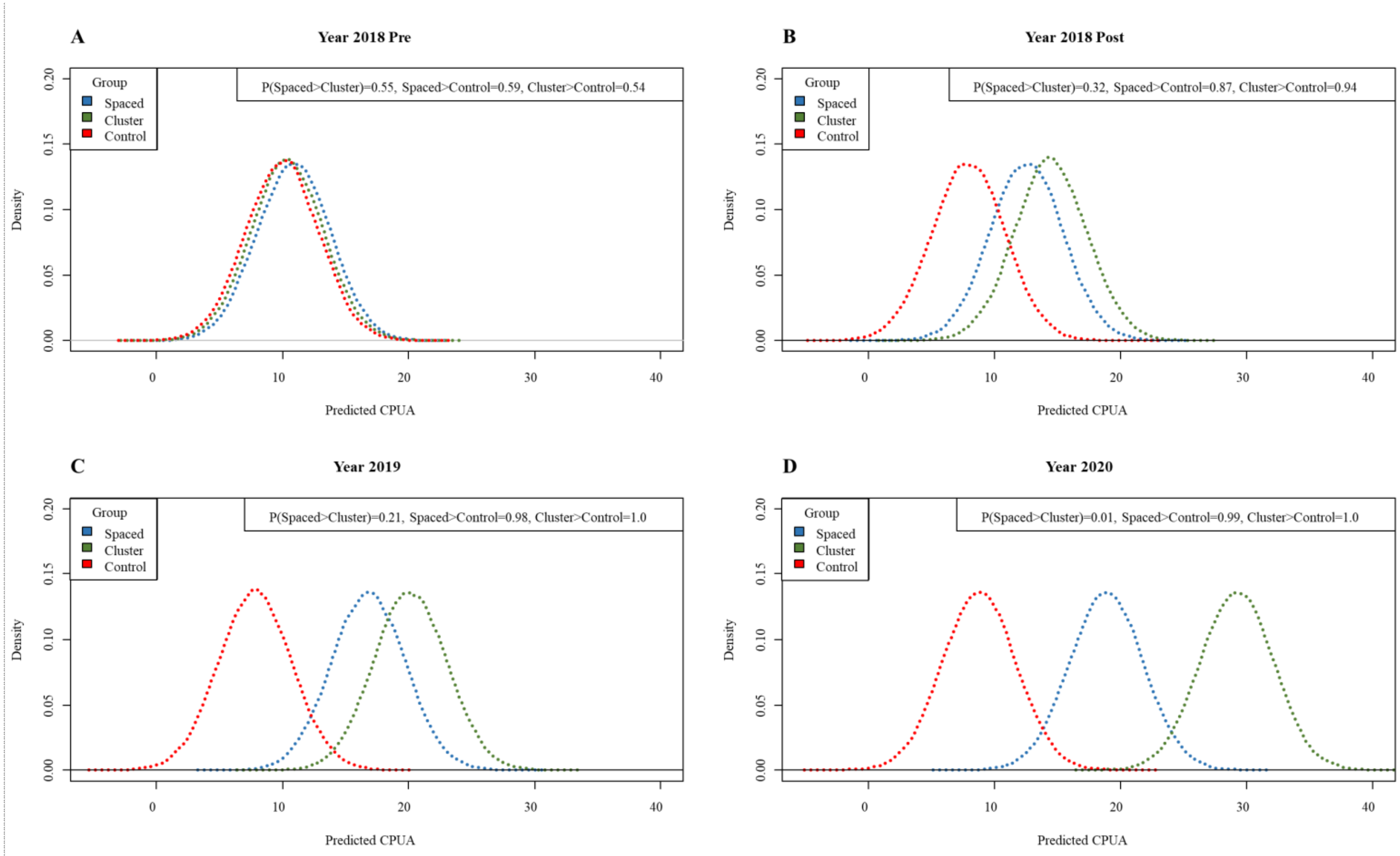

Figure 4.1.3. *Catch per unit area for 50 m seine hauls (100 m^2 standardized) density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication (Table A7) as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 3; Delta = 0.9999.*

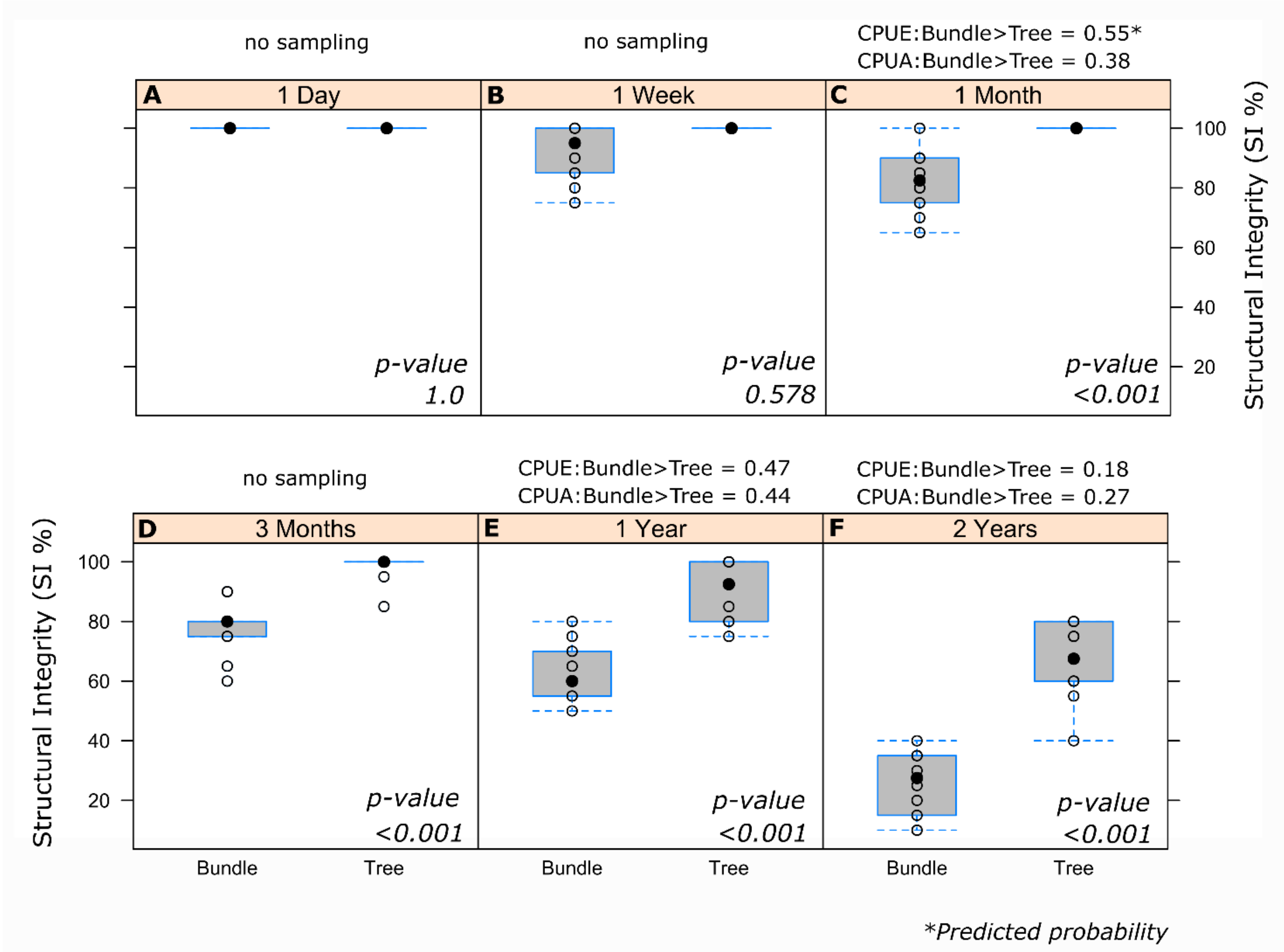

Figure 4.1.4. *Structural integrity (SI) assessment for coarse wood bundles as boxplots as part of the Coarse woody habitat (CWH) study in Lakeland County, northern Alberta, 1 day (A), 1 week (B), 1 month (C), 3 months (D), 1 year (E) and 2 years (F) post construction (Table A2). Significant differences between treatment types are indicated by p-values (Table A4). SI results are related to mean predicted catch per unit area (CPUA) and catch per unit effort (CPUE) values in vicinity of the treatment type and the probability for predicted differences based on the posterior (*; Table A9; A11). CPUE and CPUA estimates have been controlled for spatial autocorrelation (Table A3).*

Chapter 4.1 Tables

Table 4.1.1. *Proportionate stressor presence (%) for coarse woody habitat bundles and tree structures across time (1 day, 1 week, 1 month, 3 months, 1 year and 2 years post construction) covering the three main categories of integrity, placement and anchoring as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta (Table A2). Proportions are based on frequency observations during each sampling period (e.g., how many structures had a leaky sandbag out of total number).*

	Proportionate stressor presence (%)					
CWH Bundles	1 Day	1 Week	1 Month	3 Months	1 Year	2 Years
Rope tear	0	16.7	0	4.3	26.1	28.0
Log detached	0	8.3	23.5	30.4	30.4	36.0
Sandbag detached	0	8.3	41.2	34.8	17.4	0
Sandbag leak	0	16.7	17.6	21.7	8.7	0
Minor move	0	0	11.8	8.7	17.4	12.0
Major move	0	0	0	0	0	24.0
Full integrity	100	50	5.8	0	0	0
Tree structures						
Tree broken	0	0	0	9.1	23.1	5.3
Joint undone	0	0	0	9.1	23.1	10.5
Anchor line ripped	0	0	0	0	0	15.8
Anchor unearthed	0	0	0	0	0	21.1
Tree shifted	0	0	0	0	15.4	31.6
Major move	0	0	0	0	0	15.8
Full integrity	100	100	100	81.8	38.5	0

Chapter 4.2 Figures

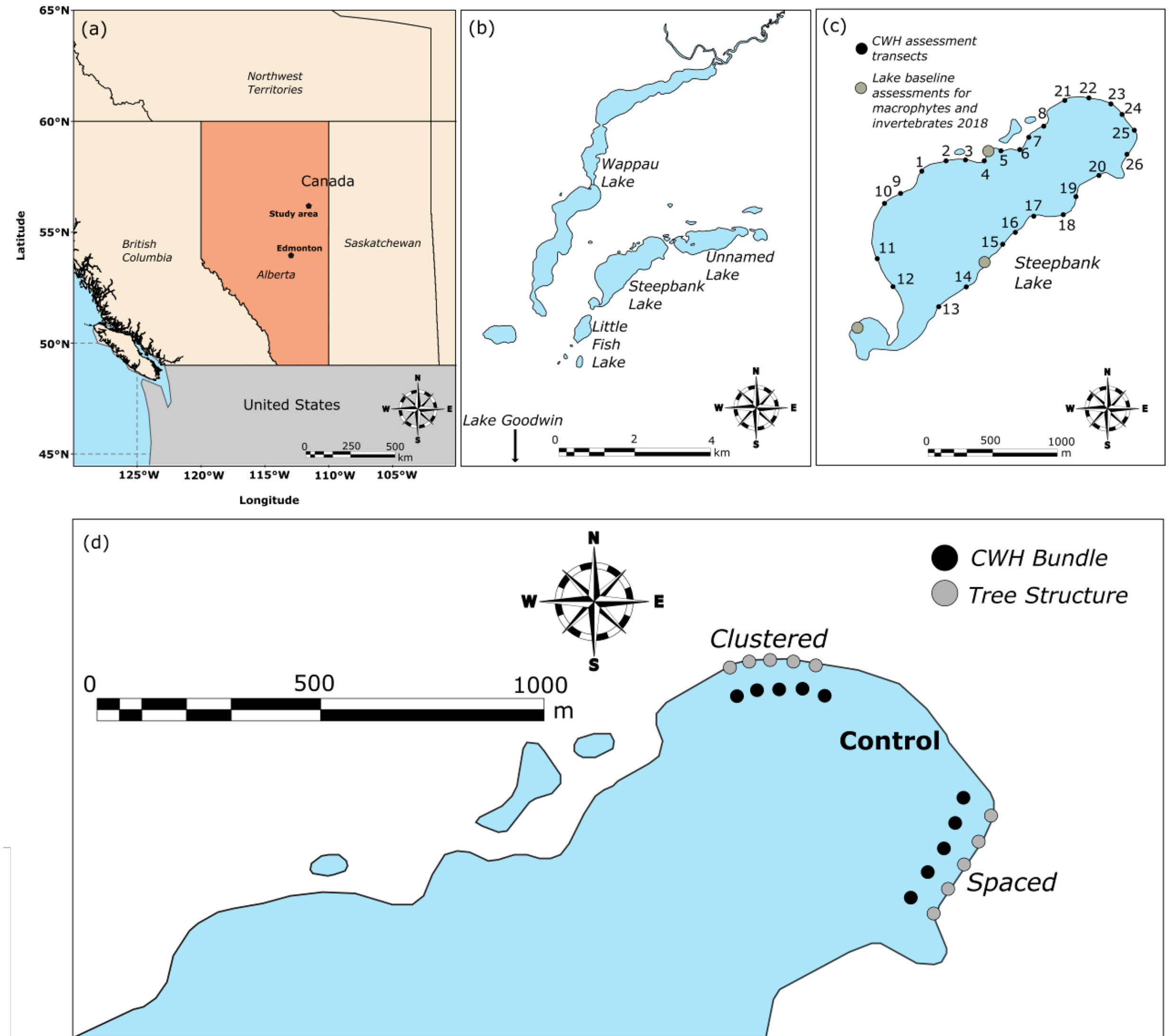

Figure 4.2.1. *Study lake (Lake Steepbank) for the coarse woody habitat (CWH) study located in Lakeland County, northern Alberta (a; b). Locations along the shoreline for assessment of structural richness and lake baseline invertebrate and macrophyte assessments in 2018 (c; Appendix Table 1) as well as placement of CWH enhancements in the two treatments (Spaced & Clustered; d).*

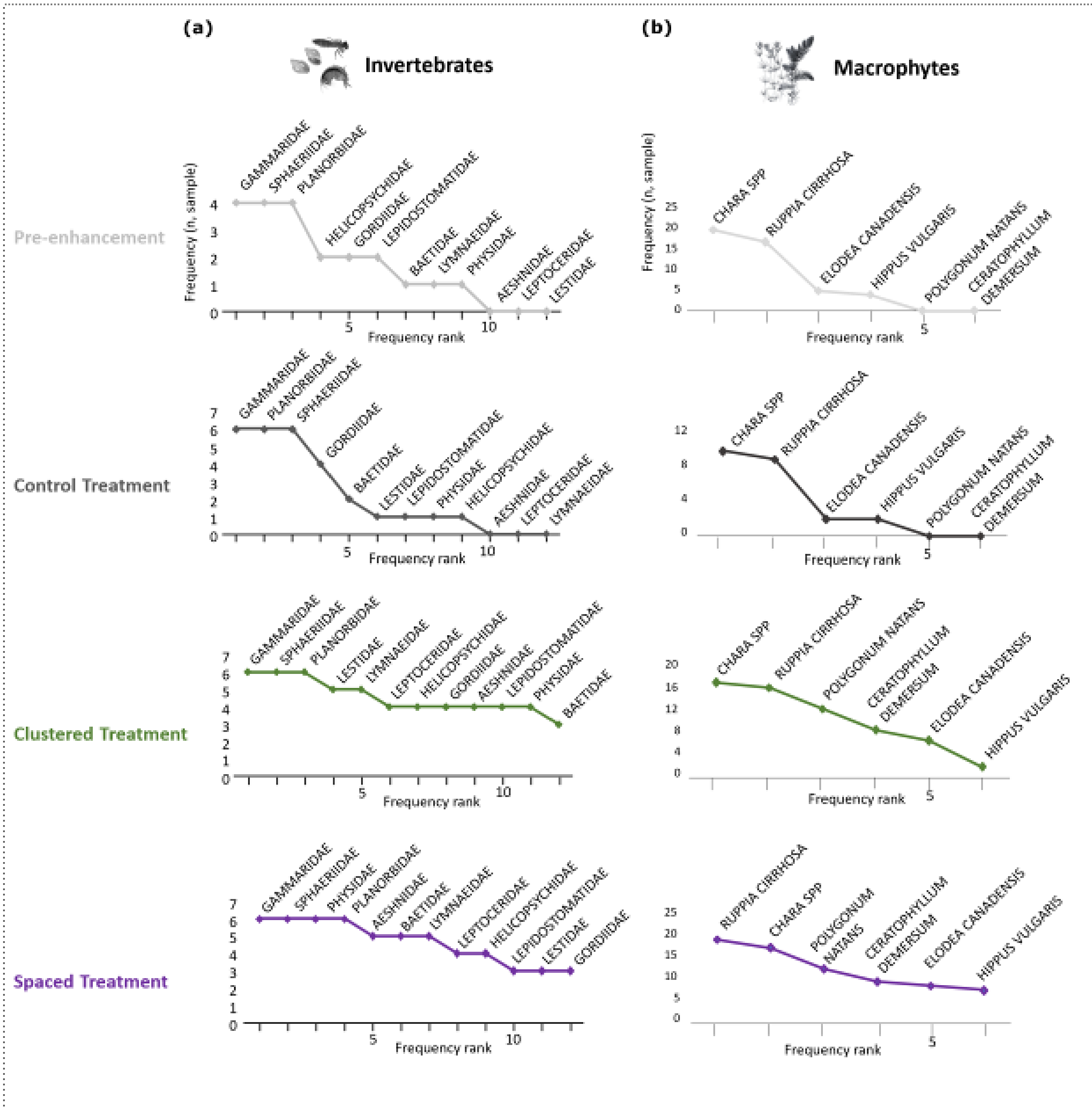

Figure 4.2.2. *Invertebrate family level (a) and Macrophyte species level (b) frequency ranks across samples and the four treatment assessments (Pre-enhancement 2018; Invertebrates n = 4 (4x5 composite samples); Macrophytes n = 12 plots); Control treatment (2019 & 2020; Invertebrates n = 6 (6x5 composite samples); Macrophytes n = 24 plots); Clustered treatment 2019 & 2020; Invertebrates n = 6 (6x5 composite samples); Macrophytes n = 24 plots); Spaced treatment 2019 & 2020; Invertebrates n = 6 (6x5 composite samples); Macrophytes n = 24 plots).*

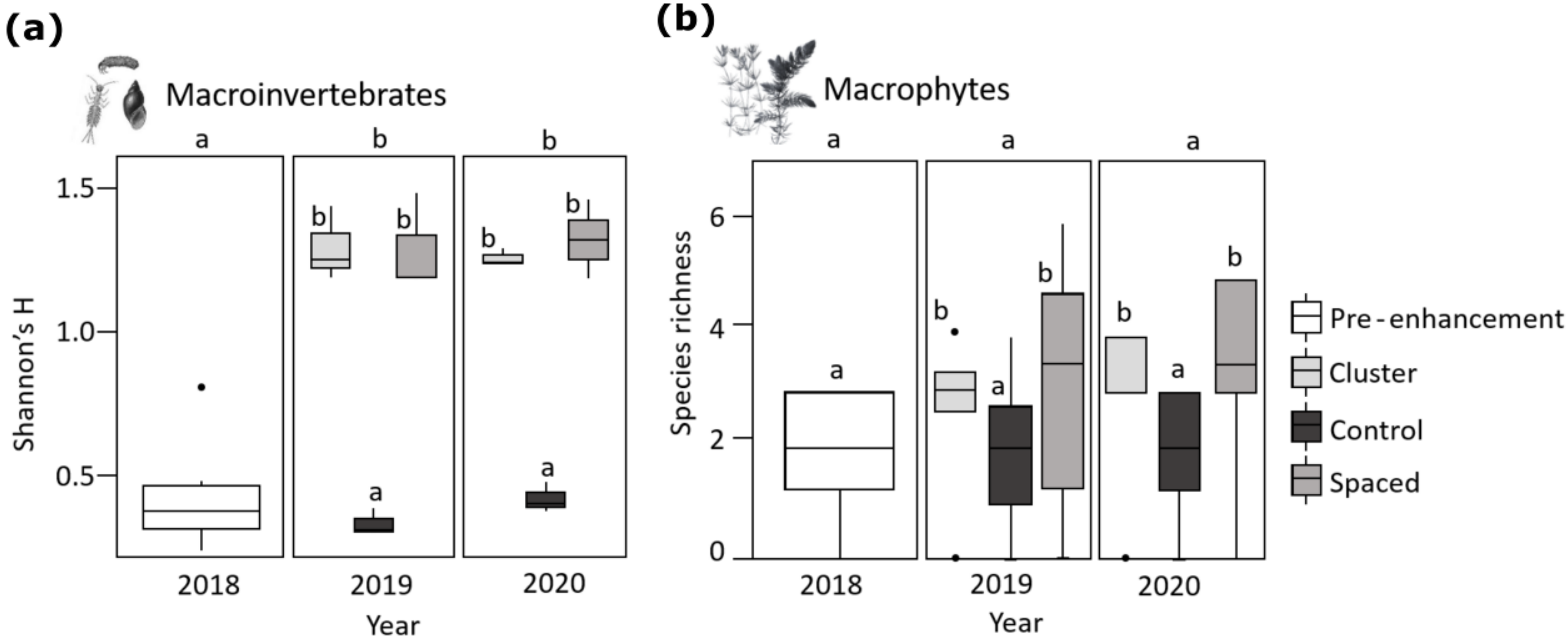

Figure 4.2.3. *Invertebrate diversity measured as Shannon's H (a) and macrophyte richness (b) across treatments (Pre-enhancement, Clustered, Control, Spaced) and time (2018, 2019, 2020) as part of the Coarse Woody Habitat (CWH) study in Lakeland County, Northern Alberta. Significant differences are indicated by letters a, b (Holm corrected p-values).*

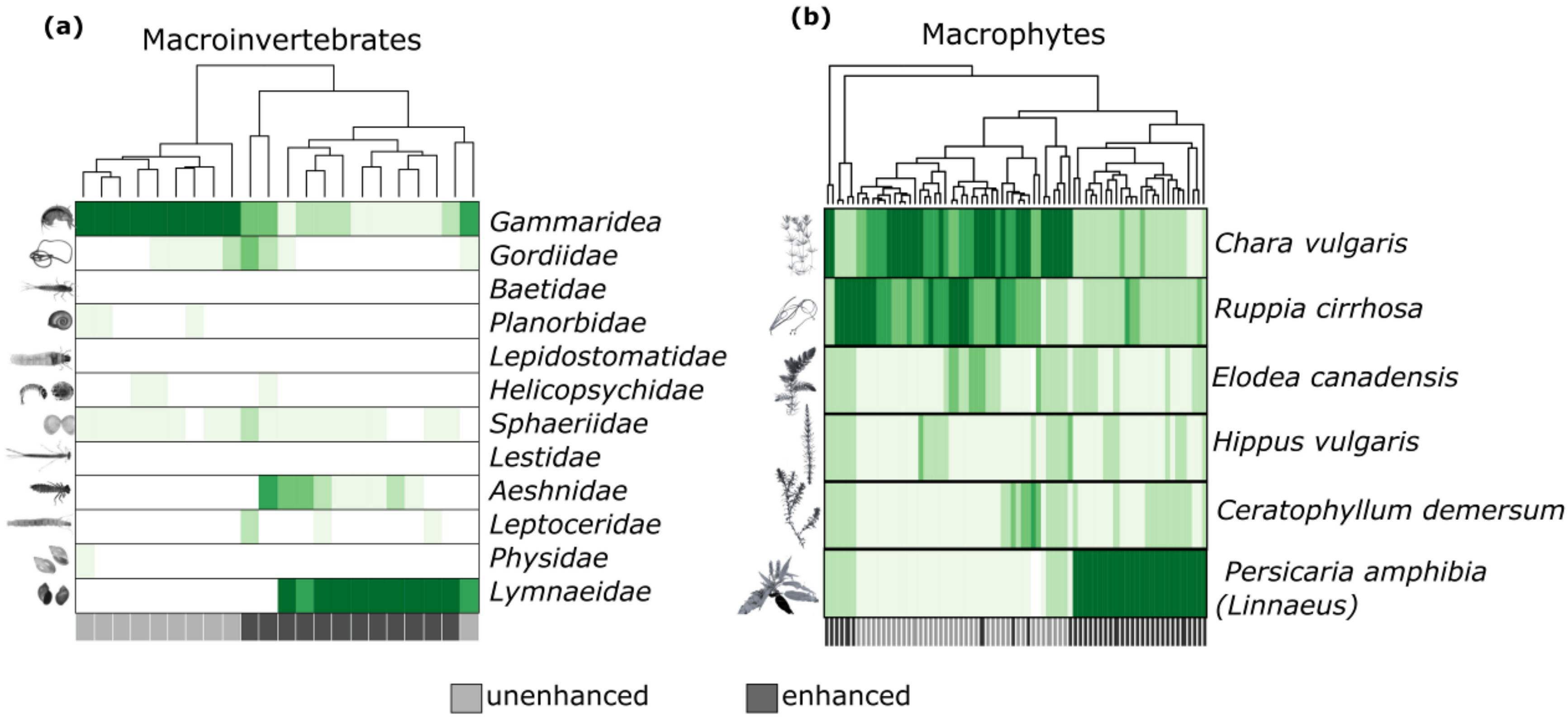

Figure 4.2.4. *Invertebrate (a) and macrophyte (b) cluster analysis (Heat plot, Bray-Curtis dissimilarity) over treatments (Pre-enhancement, Clustered, Control, Spaced) based on sampled biomass proportions. Darker colors indicate a higher biomass contribution to overall sample biomass. Individual samples are grouped into 'enhanced' and 'unenhanced' to showcase differences between treatment and cluster groupings.*

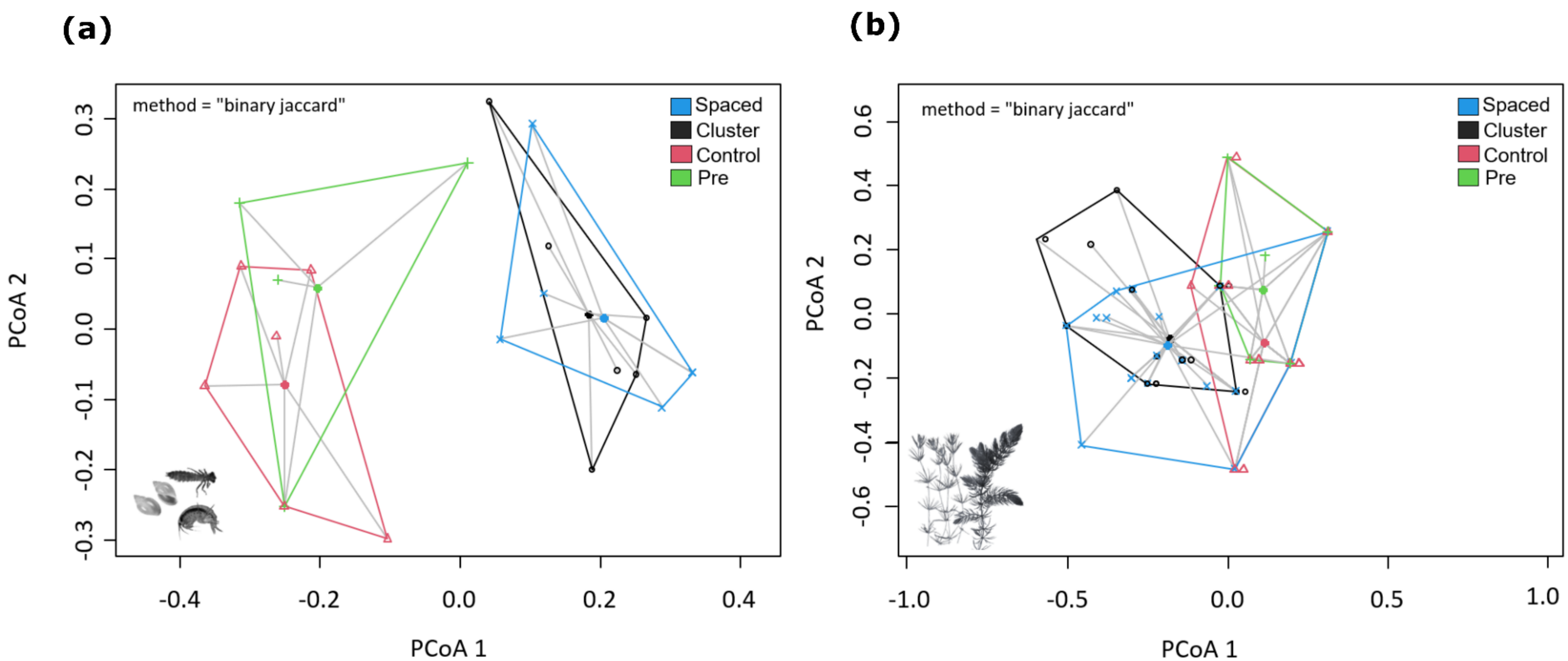

Figure 4.2.5. *Principal coordinate analysis (PCoA) based on beta diversity similarity (similarity: Jaccard, data: presence-absence, metric: centroid distance) for invertebrates (a) and macrophytes (b) across treatments (Pre-enhancement, Clustered, Control, Spaced) as part of the Coarse Woody Habitat (CWH) study in Lakeland County, Northern Alberta*

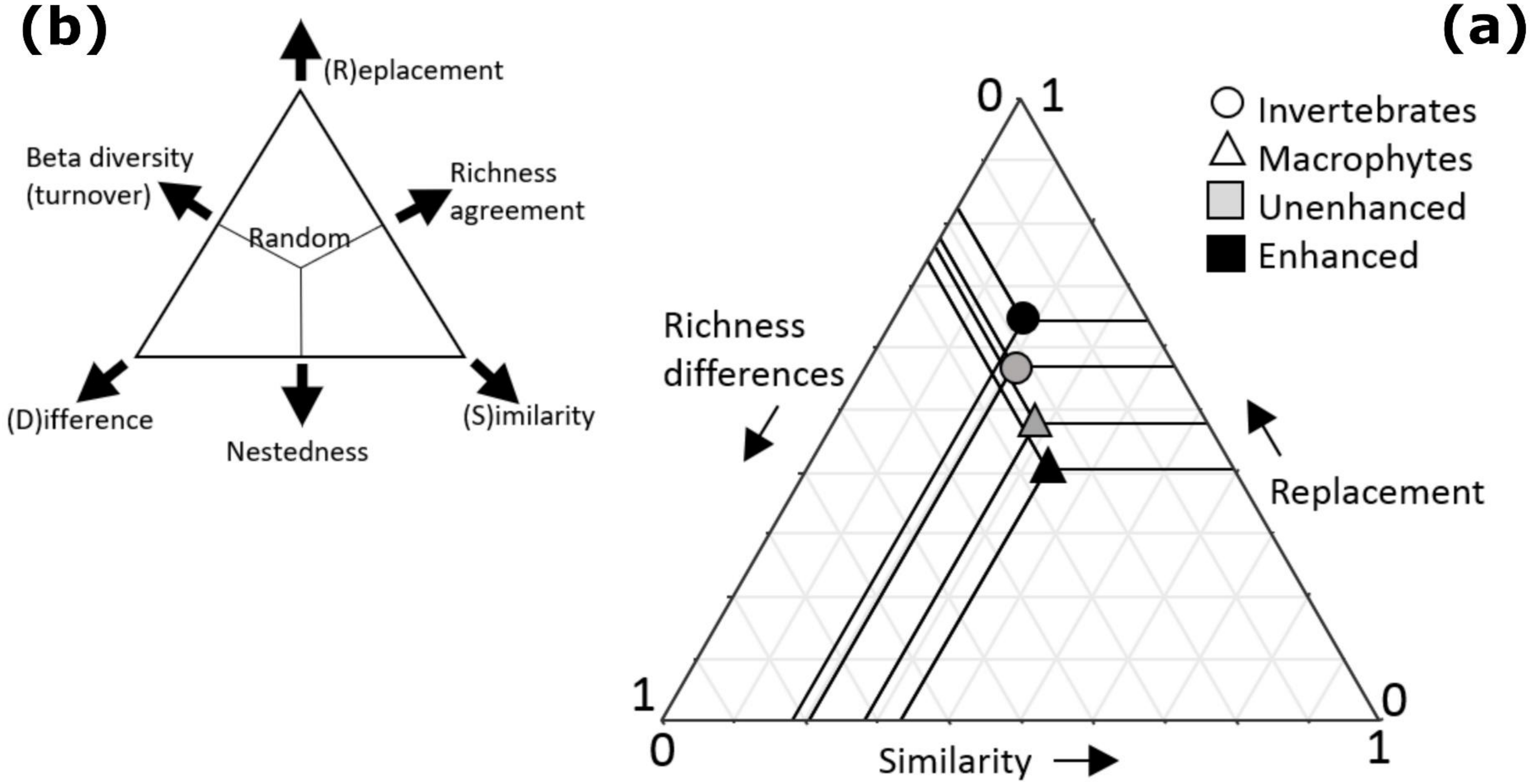

Figure 4.2.6. *SDR-Simplex plot for invertebrate and macrophyte beta diversity component changes between enhanced and unenhanced treatments based on Jaccard similarity (a). Axes indicate relative importance of species Similarity (S), Richness Difference (D) and species Replacement (R), between 0 and 1. Values translate into percentages (0 to 100%), higher values indicate higher importance of component (SDR) in driving beta-diversity changes (b).*

Chapter 5: Conclusions, synthesis, and perspectives for the future

Chapter 5.1 Conclusions

The goals of this thesis were to investigate current offsetting practices and their ability to provide feasible and sustainable benefits, compensating for approved harmful environmental impacts (Chapter 2) as well as evaluating habitat banking and its potential as an alternative offsetting mechanism (Chapter 3) followed by on the ground habitat enhancement through Coarse woody habitat introduction in a northern boreal lake with the application of beta diversity and Bayesian models to address pseudoreplication and drivers in community changes (Chapter 4). Each respective chapter confirmed current knowledge or added new insights into known issues pertaining to offsetting, others identifying new questions, hypotheses, and future research potential to focus on (Figure 5.1).

Chapter 2 shows that for offsetting projects, high regulatory compliance does not necessarily lead to high ecosystem functionality when synthesizing project results from Canada, the United States and Europe. We furthermore show that lakes are an underutilized target system compared to streams or wetlands as well as habitat creation struggling more often in meeting compliance and function requirements compared to alternative methods like enhancement or restoration. Aquatic ecosystems and populations experiencing residual or chronic harm, exerted through a development project post construction can use habitat creation, enhancement and restoration or biological manipulation like nutrient addition or stocking as potential offset options when taking adequate monitoring timeframes, pre-assessments, and unintended consequences into account (Figure 5.1).

Chapter 3 introduces an alternative offsetting mechanism in the form of mitigation and conservation banking. Banks and their credit system which translates banked area into credits which a development project proponent can buy to offset negative residual impacts, is widely developed, and established in the United States. Translation, transformation, and subsequent analyses of banking data from the United States shows that credit transactions and corresponding areas meet impact area to offset area ratio requirements for No Net Loss while missing ecological equivalency in a large proportion of cases. Reasons here are the overuse of wetland preservation which does not add any new ecosystem function or habitat area and stream and river rehabilitation adding ecosystem function in cases of losing habitat area and ecosystem function over the cause of a development project. While re-establishment of ecosystems is part of many banks, the overall poor tendency to meet ecological equivalence is concerning. Current and future demand for banks as part of our modeling framework is predicted to increase, with release schedules potentially not being able to keep up with the demand. Banking reserves furthermore are likely restricted through regional bottlenecks, with a small number of banks holding most capacities and reserves, limited to their respective service areas in the Southeast. The banking framework needs to expand significantly if being considered as a true alternative offsetting mechanisms and considering future land-sue changes and background accelerators like climate change (Figure 5.1).

Assessing offsetting practices in Chapter 2 showcased the widespread use of habitat enhancement and its benefit potential as offsets. Introduction of woody material in lotic systems was among the most applied enhancements. In Chapter 4 we test for the effects of Coarse Woody Habitat introduction in the structure-less littoral area of a northern boreal lake, with having shown that lakes are an underutilized offsetting target in Chapter 2, considering that the northern boreal is subject to large offsetting efforts, mainly due to natural resource extraction like in the Alberta Oil sands. Our results show how tangible benefits in an increased fish density in enhanced areas as well as increased biomass for invertebrates and macrophytes were detectable over the course of three years. Wood bundles degraded faster than whole tree structures, affecting fish densities in those areas but not invertebrates or macrophytes. Structural integrity, fetch distance and the wood regime in northern boreal lakes all contribute to maximizing benefits for Coarse Woody habitat introductions in northern boreal lake systems. Overall study results showcase that Coarse Woody Habitat can be an effective early stage offset in newly constructed compensation or otherwise structurally impoverished lakes. Chapter 4 also adds methodological and theoretical insights into how we usually collect and analyze field data as well as quantify community changes. Our use of Bayesian models and Moran's I shows how to potentially tackle situations of pseudreplication and low treatment sizes. Beta diversity underlines the need to go beyond Alpha diversity when assessing community changes linked to restoration or enhancement projects with beta diversity allowing to identify individual drivers across samples (Figure 5.1).

Chapter 5.2 Synthesis and perspectives for the future

The main contributions of this thesis should be split into four different parts: 1) Offsetting policy and framework; 2) Evaluation of habitat banking over the past 30 years; 3) Monitoring and data collection for offsetting; 4) Theoretical and methodological considerations for application and use cases for habitat enhancement in lakes.

Offsetting policy and framework – Regulatory and ecological drivers in unison? Working towards a positive feedback loop

Results from this thesis, especially Chapters 2 and 3, show that on paper regulatory requirements and ecological targets and goals in offsetting projects should support each other, with offsetting policies and offsetting enabling legislation being strong in many countries, strategic land-planning being more and more incorporated, and restoration or enhancement practices having been studied for decades with well documented gains and benefits (e.g., Arlidge et al. 2018; Bernhardt et al. 2007; Erwin 2008; Pattison et al. 2011; Rachmunder et al. 2011). However, our results from reviewing offsetting projects in terms of compliance and function as well as habitat banking shows how No Net Loss as a declared goal is often not met when aiming for ecological equivalency (e.g., Gardner et al. 2013; Quétier & Lavorel 2011; Turner et al. 2011; Walker et al. 2009; Zu Ermgassen et al. 2019). Our results clearly point towards the regulatory focus towards meeting administrative, financial, or area-based ratios (Bull et al. 2014; Quétier et al. 2014; Turner et al. 2011). In its easiest form a loss of one acre of wetland through road construction should be offset through the construction of an acre of wetland close by (like for like; onsite; Maron et al. 2011; Quétier & Lavorel 2011). Our results show that a) in many cases like for like and onsite offsets are not possible or feasible and b) habitat or ecosystem creation from scratch remains a very difficult task. This leads to a dilemma for the proponent between compliance and long-term project success, often left with insufficient guidance and base ecosystem metrics that do not adequately capture a holistic approach in offsetting negative impacts (Poulton 2016; Quétier et al. 2014; Walker et al. 2009; Zu Ermgassen et al. 2020). For instance, a project disrupting connectivity between downstream and upstream habitat could impede fish movement or potamodromous migration, leading to a reduced overall system productivity for some fish species. The easiest solution would be to stock fish and compensate for the lost productivity or enhance upstream and downstream habitat. However, in many of these cases productivity only captures one tangible ecosystem metric (e.g., Maron et al. 2012; Marshall et al. 2020; Maseyk et al. 2016). Long-term effects of said connectivity disruption could be more variable like increased sedimentation or flow regime changes affecting the food web and whole communities (Binh et al. 2020; Oliveira et al. 2018).

While there is no perfect solution for this dilemma, our results provide concrete evidence that long-term monitoring and multiple targets regarding ecosystem aspects lead to a higher project success and long-term provision of benefits, supported by other studies and current literature pertaining to adaptive management, sustainable conservation practices and minimum viable ecosystem size, with a greater focus on species and habitat metrics (Abdo et al. 2019; Gibbons et al. 2017; Mann 2015; Marshall et al. 2022; Underwood 2010). The latter, going as far back as the SLOSS debate, was a major finding regarding that many small-scale offsets were not monitored and abandoned, leading to project failure. Strategic land-use planning, or umbrella banks hold the potential to address these scale issues to increase connectivity among offset patches and retain functioning ecosystems on an overall landscape level (Laitila et al. 2014; Lindenmayer et al. 2017; Tarabon et al. 2021; Underwood 2010). Other aspects like incorporating uncertainty and unintended impacts as part of our residual and chronic harm meta-analysis underline the benefit of these considerations in for long-term project success. Many of our reviewed studies point towards the dangers of enhancing habitat in favor of one species, leading to detrimental effects for another species, or overestimating restoration benefits through use of improper reference systems or metrics (Gordon et al. 2011; Grimm & Köppel 2019; Laitila et al. 2014; Maron et al. 2012; Simpson et al. 2021). Overall, when taking a step back, our results point to the fact that ecological considerations need to be strengthened in offset planning, approval and implementation processes while backed by an already strong regulatory and legislative framework (Bezombes et al. 2017; Bull et al. 2014; Poulton 2016; Quétier & Lavorel 2011). While strong regulation and policies as well as overall high compliance is very encouraging, ecological shortcomings underline that we still lack knowledge on many ecosystem processes, restoration and creation efforts and translating them into general, feasible and easy to follow proponent guidance while allowing for project specific modifications and adjustments. While conservation policy overall should aim for guidance and information on a national level, conservation practices for offsetting will always need to be handled on a case-by-case basis.

A future challenge regarding the offsetting framework will be on how to incorporate climate change into current and future offsetting requirements. Discussions to make climate change impacts an offsetting requirement are ongoing and seem likely to be implemented at some point (e.g., Anderson et al. 2017; Delgado et al. 2011; Roberts et al. 2020). Offsets are meant to be long-term, ideally in perpetuity and climate change will have a significant effect on provided benefits, especially when considering temperature sensitive species or systems. Offsets of the future will need to consider climate refugia both in terms of the physical offset form as well as location, potentially changing the dynamic of like for unlike and like for like and onsite and offsite offsets (e.g., Coggan et al. 2021; May et al. 2017).

Evaluation of habitat banking over the past 30 years – Current best practice could leave regions, species and ecosystems sidelined

Results from Chapter 3 should not be seen as an endorsement or a critique on habitat banking but rather an assessment of its current state in the United States. Our analyses are of timely importance since the United States has the most well developed and longest standing banking network on a global level with many other countries entering the early development stages of similar banking frameworks (Fox & Nino-Murcia 2005; Robertson 2004; Santos et al. 2015). Drawing knowledge from the US case is invaluable to avoid pitfalls and to provide general guidance on bank establishment and best practice. While done on different regional scales, our results are the first comprehensive approach that attempts to assess banking practice in the Unites States on a national level, using conservation as well as mitigation banks (Lave 2018; Levrel et al. 2017; Poudel et al. 2018; Reiss et al. 2009). Our findings confirm that recorded regional and state-level trends, also persist on a national level. The issue here is like our general findings in our offsetting review that ecological equivalency is often missed in banks (e.g., Levrel et al. 2017; Reiss et al. 2009; Robertson 2004). While many regulatory agencies have already incorporated failsafe's like increasing ratios or preventing the sole use of preservation credits without any other credit type, the proportion of banked area being used as offsets while providing ecosystem function gain versus ecosystem function and habitat area loss is alarming (Levrel et al. 2017). Our findings point towards legislation lacking behind current practices and knowledge. Our national-scale analyses are complemented by a multitude of small-scale studies as well as official reviews conducted by regulatory agencies (e.g., USEPA, USFWS etc.) that all point towards the still prevalent mismatch between impact and offset ecological equivalency (e.g., Levrel et al. 2017; Poudel et al. 2018; Reiss et al. 2009; Robertson 2004). Policies and guidance documents however have not been updated and adjusted accordingly. Other instances show how memorandums e.g., for the Endangered Species Act were withdrawn recently. While changes are happening on a state level, like our example from California shows, other cases like the one of the Greater Sage Grouse shows how easy it is to let competing stakeholder and political interests override safeguards, only compensated for by NGOs and private investors (Catalano et al. 2019; Stoellinger 2014). Results from this thesis and the many other studies evaluating banking practices need to be considered for implementation into updated policies and guidance documents to strengthen the ecological benchmarks and requirements for banking practices in the United States. An updated uniform guidance document, incorporating the collected data since the last memorandum could shift banking practices into a more sustainable and ecological equivalent direction.

Public interest and involvement are strong considering our analyzed data and bank development and demand from the past decades. Previously mentioned changes to guidance documents and policy should also consider future developments in the spatial distribution and extent of the banking network in the United States. Our results indicate quite drastically how certain regions in the United States are not part of the current banking network with most reserves and capacities being limited to the Southeast and a few large banks. While this distribution makes sense looking at land-use in the Southeast in combination with vital wetland and river systems presence, other areas are currently sidelined (Barbier et al. 2013; Hefner & Brown 1984; Lave et al. 2008). For instance, large proportions of land in the Mid- and Northwest and Southwest remain devoid of banks. Banking, being a market driven environment, suggests that the demand is simply not there in these regions. However, our results and ongoing research suggest a different primary driver, ownership. Many of these bank-free areas are managed through federal agencies like the department of forestry (Vincent et al. 2017). Studies have shown how federal land management has often been ineffective with frequent ownership changes between agencies, often resulting into detrimental long-term environmental effects (e.g., Brown 2003; Heisel 1998; Lowell & Kelly 2016). Given the large public interest as well as demand for banking credits suggested by Chapter 3, our results point towards a potential restructuring of land management in these areas. Projected land-use for forestry and agriculture as well as significant impacts on ecosystems and species communities through predicted increase in water-stress by the World Research Institute and their aqueduct tool lets me ask the question "are current federal conservation strategies for these regions sufficient to face these threats?". Banking could provide an alternative way of alleviating financial as well as operational pressure for these large stretches of land, especially when considering that climate change mitigation and offsets are lily to become part of official offsetting policies, e.g., through carbon credits and investments (Bekessy & Wintle 2008; Jantarasami et al. 2010; Mori 2020; Spies et al. 2010; WRI 2022).

Finally, this thesis adds to the highly debated and controversial topic of habitat banking covered in Chapter 3 regarding conservation banks and their currently covered species list (Fox & Nino-Murcia 2005; Poudel et al. 2019; Sonter et al. 2019). Driven by listings on the Endangered Species Act, current conservation banking practices as well as studies are skewed by the fact that over half of the banked area can be attributed to one large habitat bank for the Greater Sage Grouse in Wyoming. Accounting for that bias shows that banked area for endangered species remains rather small as does the number of currently covered species. Our thesis underlines the need for incorporating more endangered species into the banking framework, a task that will become more and more difficult. Thus far, conservation banking largely relies on preserving remaining habitat of endangered species (Fox & Nino-Murcia 2005). The true challenge will be to advance restoration practices and science towards re-establishing and restoring crucial habitat for endangered species while fostering transboundary collaboration and agreements to be able to include migratory or long-range species into the banking network (Lambertucci et al. 2014; Miller & Hobbs 2007; Nordstrom 2000; Trouwborst 2012). While we found early efforts in that regard, current practice seems to be still a long way off. The recently withdrawn memorandum of the Endangered Species Act gives hope that, though withdrawn, amendments will enter policy and guidance documents at some point (Hartl & Owley 2021; Kerkvliet 2021). Our results also support the possibility that conservation banks together with mitigation banks could be incorporated into strategic land planning processes (e.g., early systems in the Netherlands and France) to reduce isolation create positive feedback loops between banks (e.g., Grimm et al. 2019; Janssen et al 2012; Penjor et al. 2021). For instance, a conservation bank preserving habitat patches for endangered species A could benefit from a close by mitigation bank, re-establishing habitat unrelated to the species A. Our results support evidence for landscape level planning and bundling of provided ecosystem services to allow for larger interplay among areas as indicated by other literature and studies (e.g., Deal et al. 2012; Tomer et al. 2013; Tallis et al. 2008; Turner & Daily 2007).

Future research questions would need to address whether it is possible in an ecological as well as legislative manner (different legislation for both types) to connect conservation and mitigation banks better as well as how geospatial models could help identify priority areas suited for banking while using land-use and climate change data. Another necessary assessment would relate to bank and credit prices which is often identified as increasing investor uncertainty and risk and is often labeled as non-transparent. Regional or species related credit price and bank establishment as well as maintenance costs could provide guidance on how to prevent a price driven market, where valuable species and ecosystems will outcompete less valuable ones (Fox & Nino-Murcia 2005; Pawliczek & Sullivan 2011; Poudel et al. 2019).

Monitoring and data collection for offsetting - Feasible and necessary or expensive and impossible?

Another contribution from this thesis is that it confirms how important appropriate monitoring timeframes and pre-impact assessments are as well as how data collection is still a prevalent issue, preventing effective project evaluation. Monitoring is one of the main requirements for offsetting but seemingly almost always listed as one of the main issues relating to offsets (e.g., Bull et al. 2017; Legg & Nagy 2006; Lindenmayer et al. 2017; Maron et al. 2016; Regan et al. 2007). While generally speaking' more equals better' could be said for monitoring, results from all our chapters allow to differentiate this statement more. Our results from Chapter 2 show that minimum monitoring timeframes can help increase project success as they allow for adaptive management over time and to detect and address latent effects, chronic harm, and potential offset failure over time. The general need for increased and more consistent monitoring aligns with other studies and relates to the questions of '*how long*' we are monitoring, *'why'* we are monitoring and *'what'* we are monitoring (Bull et al. 2017; Jones et al. 2013; Lindenmayer et al. 2017; Maron et al. 2016; Regan et al. 2007). Monitoring is expensive and requires long-term resource allocation and sound methods as well as monitoring metrics thus a lot of focus is put on the *'how long'* aspect. However, focusing on the *'why'* and *'what'* should take precedence as shown in the thesis, assuming that proper pre-impact assessment haven been conducted. Having pre-impact assessments or proper reference sites available is a prerequisite for offset success as supported by our findings from Chapter 2. For instance, if a proponent wants to offset a weir construction through stream restoration in an adjacent stream by using reference values from the literature on stream productivity it could introduce a potential over or underestimation of expected benefits. Pre-assessments, both in the impacted stream as well as targeted restoration stream would reduce that bias. Another example was a case study from Chapter 2 with stocking efforts trying to address reduced habitat connectivity in New Zealand. Efforts stayed below expectations due to missing an unrelated bottleneck further downstream (Unwin & Gabrielson 2018). Overall, our review and synthesis results combined with meta-analyses effect sizes underline the need for proper pre-assessments.

Following proper pre-assessments, results from our thesis suggest that the *'why'* and *'what'* should be the next thing to address. A rather simple stocking case as mentioned in Chapter 2, replacing a certain number of lost fish and their lifetime contribution towards the population, or translated to age equivalency, would require regular stock assessments in combination with number of lost fish. We monitor to assess if stocking is an adequate replacement (*'why'*) through situation appropriate metrics (e.g., productivity; survival; biomass; *'what'*). This situation assumes that the fish loss is not due to environmental degradation or other associated impacts that would require other large-scale population or environmental monitoring. We can use our Coarse woody habitat introduction as another example. A sole focus on community response to our enhancements without monitoring structural integrity changes over time would have introduced a bias for recorded fish density and reduction of benefits over time without being able to put it in the proper context. In an actual management context, structural integrity monitoring would allow for fast response and adjustment of enhancement structures. Again, in this case monitoring becomes about the *'what'* and *'why'*, after we were able to look back at multiple years of pre-assessment data from the lake and region, an important question to consider when designing monitoring targets, metrics, and methods. Monitoring habitat enhancements that have already failed or measuring wrong metrics for decades would be a waste of time and money. Our thesis has contributed to this regard as both papers from Chapter 2 being incorporated into science advisory meetings, having the potential to being considered for current and future policy and guidance changes. Overall monitoring is proven to be expensive, but our findings show how pre-assessment and the proper use of *'why'*, *'what'* and *'how long'* can reduce costs, increase efficiency, and reduce offset failure.

Finally monitoring also can be related to project size as specifically shown in Chapter 2.1. Larger projects with more monetary investments and higher stakes so to speak, naturally also invest more time and effort into monitoring or have higher monitoring requirements are per compliance criteria. Future research as well as guidance could focus more on the small-scale projects that are already prone to ecological failure and non-compliance due to scale, with inadequate monitoring adding to the issue (e.g., Dietz et al. 2010; Horwich & Lyon 2007; Legg & Nagy 2006; Pilgrim et al. 2013). As already mentioned, regarding habitat banks and connectivity issues in patchy offsets, monitoring is also in dire need for a more collaborative and overarching framework that would allow smaller projects to receive adequate monitoring as well as also be able to detect potential interactions among projects on a landscape level (Conley & Moote 2003; Pilgrim et al. 2013; Regan et al. 2007; Woellner & Wagner 2019). Exploring the potential for collaborative monitoring and data collection frameworks in combination with the use of new technologies (e.g., acoustic surveys; unmanned aerial vehicles; advanced habitat indices) would be a worthwhile endeavor and focus of future studies.

Application and use cases for habitat enhancement in lakes – documented benefits as part of the age-old ecological discussion of pseudoreplication and inference bias

Our documented findings on benefits of Coarse woody habitat for northern boreal ecosystems are important regarding their temporal nature and suitability as early-stage enrichment for fish species and long-term provision of macroinvertebrate habitat and accumulation of organic material, beneficial for macrophyte communities (e.g., DeBoom & Wahl 2012; Hrodey et al. 2008; Larson et al. 2001; Sass et al. 2006 & 2019). Supported by literature and other case studies for mostly lotic systems, our small-scale experiment contributes greatly to the persistent issue of pseudoreplication. Pseudoreplication can and does often occur in many environmental or ecological study due to inadequate replication of treatments or replicates not being independent from each other. This issue goes as far back as Hurlbert, stating that between 1960 and 1984, out of 176 experimental ecological studies, 27% had pseudoreplication associated with them, 48% of all studies using inferential statistics (Hurlbert 1984). The main issue with pseudoreplication is that it can skew tested treatment effects (wrong error term for hypotheses). Newer studies have shown how biodiversity conservation in tropical ecosystems relies on a body of literature that is laden with unwarranted inferences (68%; Ramage et al. 2013) In our case it could have skewed our results from our Coarse woody habitat introduction either towards a negative or positive inference considering that we only had three treatments with sample size one in each. This becomes crucial when random chance events might be amplified (Hurlbert 1984). For instance, a negative low fish density due to random events could lead to the conclusion that the whole treatment was ineffective. Hurlbert has since divided ecologists with his call for replicated treatments and dispersion of treatments and controls. However, the reality of ecological field experiments is often far more complicated than that. Funds and manpower are limited, timeframes are often dictated by 2-to-4-year degrees and remote areas often do not allow for conducting studies in the best way possible (Hargrove & Pickering 1992). The same applies for sensitive, endangered, or low population species where replication is often impossible by default (Hargrove & Pickering 1992; Zuckerberg et al. 2020). So, should we cease to have small scale ecological studies and what does that mean for conservation monitoring and informing current and future practices on said data? While I see the merit in both arguments, I strongly have to say no.

Small scale studies and experiments hold immense value to collect data, learn valuable methodological and implemental lessons that can be scaled up for larger projects or combined with other similar studies in a broader framework. For instance, we learned valuable lessons on fish detection across life-stages or structural degradation of different enhancement structures over time thus pseudoreplication should not deter us from our small-scale approaches (Davies & Gray 2015; Zuckerberg et al. 2020). Newer studies have fostered the stance that pseudoreplication is a pseudoissue and that replication should not lead to sacrificing temporal and spatial scales. Follow-up experiments, data exchange among scientists and other parties and meta-analyses are suggested to compensate for low treatment sizes (e.g., Hargrove & Pickering 1992; Oksanen 2001; Zuckerberg et al. 2020). Our experimental study shows how utilizing pre-assessments, sampling data from previous years or other areas as well as statistical adjustments can effectively address pseudoreplication or at least lower the likelihood for trapping and amplifying random events in the data. Bayesian models are one approach in a long line of other statistical remedies like mixed-effects modelling, generalized linear mixed models, state-space modeling, and hierarchical modeling approaches (Lazic et al. 2020; Millar & Anderson 2004).

Again, there are pros and cons for every one of these models and our Bayesian approach has garnered critics as well, mainly referring to inherent subjectivity of priori probabilities (e.g., Oksanen 2001). Overall, each method must be individually evaluated in its appropriateness for collected data or sampling design. In our case it was feasible due to the amount of collected data, parallel sampling efforts in other areas and prior knowledge of the system. In my mind our small-scale experiment has shown how a Bayesian approach can reduce pseudoreplication concerns in an easy and feasible fashion given the right circumstances, with many excellent instructional documents available (e.g., Lazic et al. 2020). Even when assuming that pseudoreplication is a pseudoissue, Bayesian modeling frameworks and other mentioned alternatives are easily implemented to double check inferences and doing our due diligence as scientists and researchers to reduce potential bias.

A note on data collection and transfer to centralized databases

The suggested use of data sharing and exchange, meta-analyses and follow up experiments comes full circle when going back to previous chapters. Chapter 2 and 3 clearly show how pooled data can be an incredibly powerful way of evaluating current offsetting practices. However, large-scale data collection in centralized databases suffers from reporting biases that can start as early as with the designated metric for data collection all the way to how data is inserted and transformed in a database (e.g., Bird et al. 2014; Geijzendorffer et al. 2015; Godet & Devictor 2018; Isaac & Pocock 2015). For instance, banking data for Chapter 3 is often limited to an impact area in acres and an offset area in acres (credits) but no other information. This basic ratio-based metric makes it impossible to evaluate ecological equivalency and we only managed to do so for a small proportion of transactions through extensive web scraping and tracking down induvial case files. While remarkable to have a database like RIBITS available, its actual use case does not live up to its potential. Right now, it is a very interesting data repository but very much ineffective in allowing for a proper evaluation of banking effectiveness and practices.

The same goes for assessment methods. Many offsetting projects use a few general assessment approaches based on rapid assessment methods (RAM) or hydrogeomorphology (HGM). A huge focus has been set by the scientific community as well as agencies towards developing newer and better rapid assessment methods, driven by the desire to collect as much data as possible with as little personnel as possible (e.g., Corlett 2017; Erftemeijer 2001; O'Donnell et al. 2012). While feasible and cost efficient, the broad and varied categories assessed by RAMs often reduce their resolution and informative value of the collected data. Rapid assessments have been criticized to not sufficiently capture ecosystem health and functionality, a concern our thesis and my personal experience of reading hundreds of reports and case files can confirm (Gibbons & Freudenberger 2006; Herzog et al. 2002; Quetier & Lavorel 2011). At this point it all comes back to the *'Pre-assessment', 'Why', 'What'* and *'How long'* questions to develop efficient monitoring protocols. Quantity and costs can only outweigh quality issues for so long. This is not to say that they are without merit. In my view RAMs are essential for baseline monitoring or for detecting broad changes in the environment (e.g., climate change; eutrophication). However, as far as our thesis shows, they are overused and certain projects and monitoring programs (e.g., species-specific; certain cases of residual harm and different bank types) would benefit more from focussed monitoring efforts. More focussed and better-defined monitoring programs also often reduce long-term costs and monitoring timeframes (Legg & Nagy 2006; Nichols & Williams 2006). This thesis shows how monitoring and data collection can help inform best practice, evaluate current offsetting approaches as well as reduce experimental biases like pseudoreplication but needs to improve as a whole to better capture project goals and targets, specially when trying to unify needs of scientists, agencies and citizen science at the same time in a more and more collaborative and interconnected conservation and management network with organized and accessible databases (Geijzendorffer et al. 2015; Godet & Devictor 2018).

Chapter 5: Figures and Tables

Chapter 5: Figures

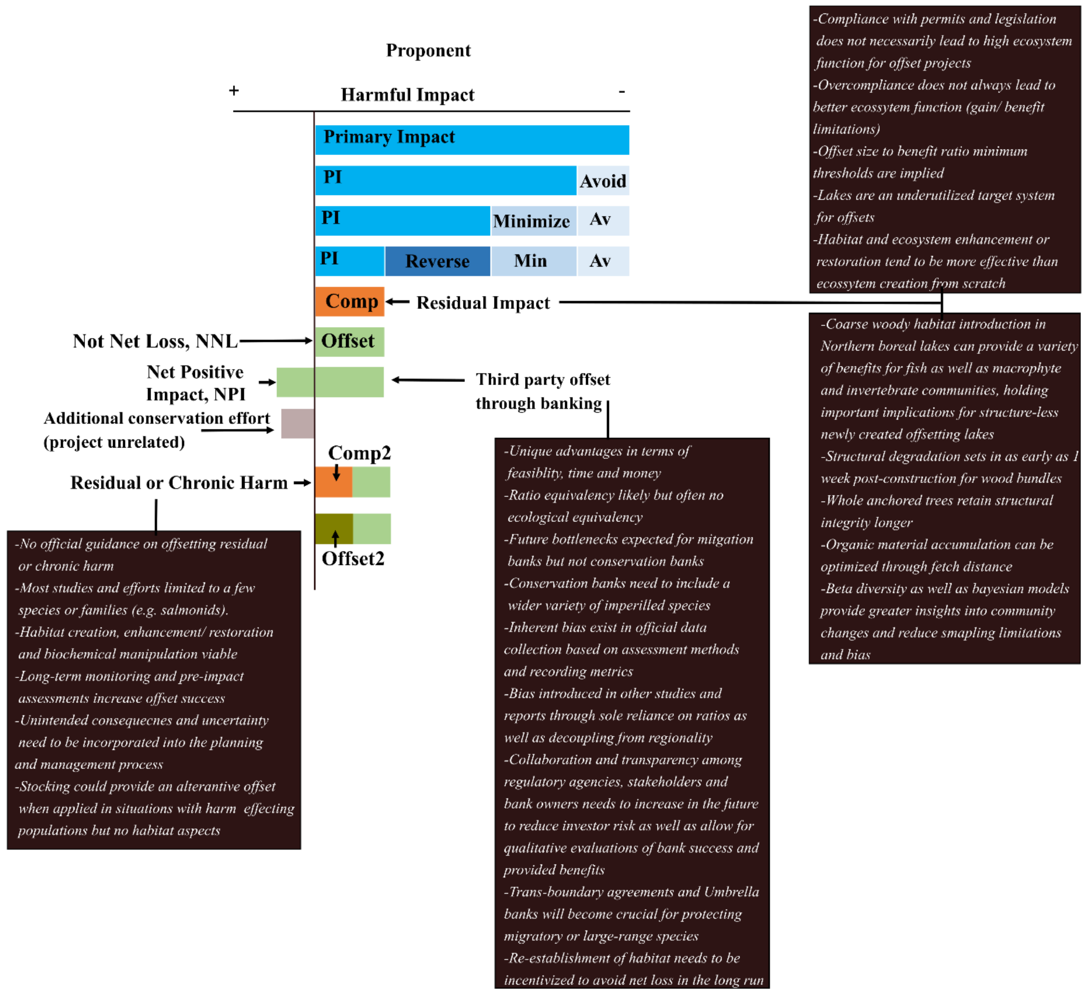

Figure 5.1. *Key findings from all thesis chapters and implications for individual aspects of the mitigation hierarchy and the overall framework (adapted from Kiesecker et al. 2011; Ten Kate et al. 2018).*

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Appendices and Supplements

Chapter 2: Appendices and Supplemental Material

Chapter 2.1 Appendices

Table A.2.1.1. *ANOVA and correlation between Function and Compliance in relation to location.*

Compliance	Df	Sum Sq	Mean Sq	F-value	Pr(>F)
Europe	1	2.473	2.4732	9.33	0.004
Residuals	46	12.193	0.2651		
US	1	10.77	10.770	30.45	<0.001
Residuals	278	98.32	0.354		
Canada	1	2.381	2.3805	19.75	<0.001
Residuals	18	2.170	0.1205		

Table A2.1.2. *ANOVA model statistics, their degrees of freedom, and levels of significance for ecosystem function regarding compliance. Linear permuted model design used for non-linear distribution of data.*

	Df	Sum Sq	Mean Sq	F-value	Pr(>F)
Function - Compliance	1	18.34	18.342	48.72	<0.001
Residuals	347	130.64	0.376		

Table A2.1.3. *Reduced ANOVA-table highlighting positive and negative effects of project targets and methods. P-values based on a permutated linear model and ANOVA, testing for significant effects of methods and project target on compliance and function scores. Significant positive or negative effects are greyed out. Significant effects are based on comparison of factor presence or absence in a project.*

	Function		Compliance	
	p-value	Effect	*p*-value	Effect
Target				
Productivity	<0.05	Positive	<0.001	Positive
Habitat	<0.001	Positive	n.s.	None
Function	<0.05	Negative	0.05	Negative
Method				
Restoration	<0.05	Positive	n.s.	None
Enhancement	n.s.	None	0.05	Positive

Creation	<0.05	Negative	<0.001	Negative

Chapter 2.1 Supplements

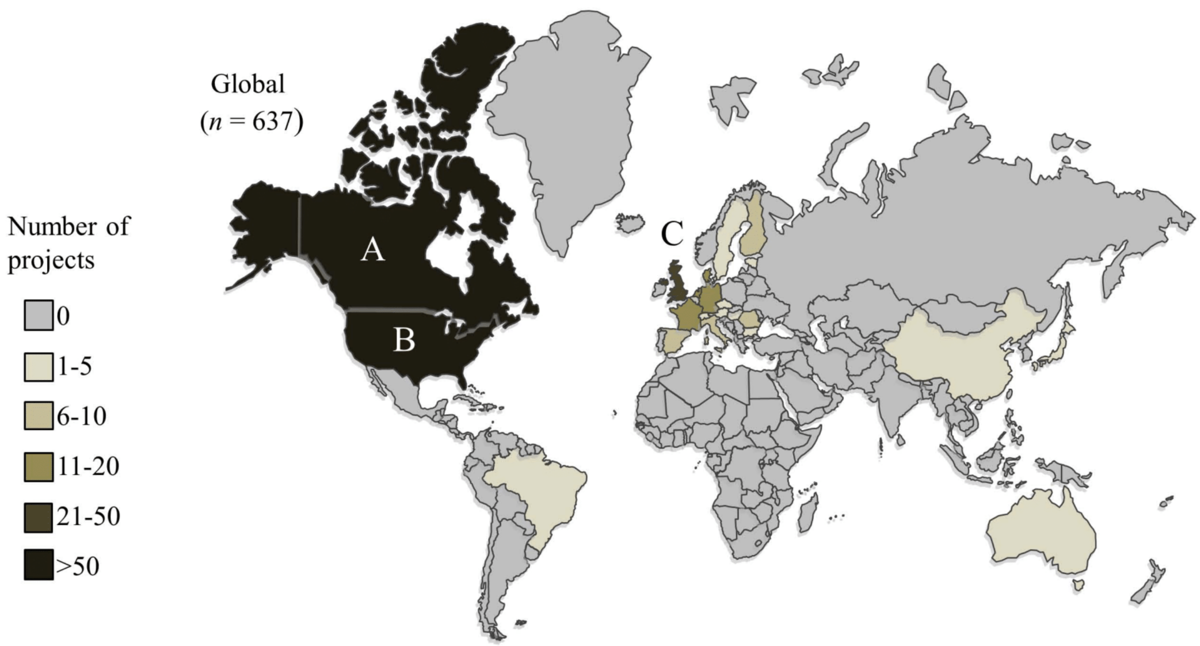


Figure S2.1.1. *Global distribution of all projects including banking (n = 637). Most projects were located in Canada (A), the United States (B) or Europe (C).*

Sample assessment from an official permit:
Cook Creek, North Thompson River, British Columbia.

Expected Impact:
- 2800m² of impaired wetland and stream habitat

Offset requirements in permit:
- Creation of 3500m² of compensatory fish habitat and Coho productivity comparable to reference systems in same drainage.

Compliance results:	**Meaning:**	**Score:**
• Re-creation of secondary channel (335m long and 4-5m width	• Required area met, slightly exceeded (107%)	2
• Creation of pond (1180m²)	• Required habitat restoration fully met	2
• Compensatory channel creation (1040m²)	• Required productivity met	2
• Revegetation complete		
• Productivity 0.3-1.0 fish/m²		Overall 2

Function results:	**Meaning:**	**Score:**
Fish utilization		
• No suitable results do to downstream migration blockage	• Fish utilization not assessable due to constraints,	-
Water Quality (DO)	• Physical features stable (e.g. banks and pools)	2
• DO not comparable to reference, lower	• Water quality not met	1
Physical features	• Flow pattern only partial comparable to reference	1
• Stable and comparable to reference		
Stream flow		
• pattern not fully comparable to pre-construction		Overall 1

**For both compliance and function an error margin of 10% was included which is officially applied in most permits to account for variability and assessment difficulties. (e.g. meeting 90% of the permit requirements or eco-system assessment criteria was accepted as full compliance/full functionality).*

Sample assessment from an official permit & report:
Highway Construction, Newfoundland, Canada.

Expected Impact:
- Loss of 162 meters of fluvial salmonid habitat after mitigation was deemed impossible

Offset requirements in permit:
- Replacement of stream section (162 m), No Net loss for salmonid species trough:

"Provision of meanders, creation of pools, removal of vegetation, planting of streamside vegetation, substrate placement (boulders, rubble,…), ensure bank and feature stabilization"

Compliance results:	**Meaning:**	**Score:**
• Creation of 195.2 m of stream (20% gain)	• Required area surpassed significantly (120%)	3
• Riffle area 5.16 m³ (6% gain)		
• Pool area 1.69 m³ (134% gain)	• Required habitat constructed and habitat ratios adapted to better suit different age classes of salmonids	2
• Pool:Riffle ratio 1:3 (209% gain)		
• Undercut bank 9.75 m³ (100% gain)		
• Fish biomass: (salmonid biomass as surrogate for 'total productive capacity': 634.9 g, 2.58 fold increase post-construction	• No net loss achieved, 2.58 fold increase	3
		Overall 3

Function results:	**Meaning:**	**Score:**
• Assessment of bank stability: all structures stable after readjustment over the course of 3 years	• All assessed ecosystem aspects were functioning as expected after readjustment	2
• (Benefits to one species at the expense of another (preference given to brook trout considering their importance for recreational fisheries), increase in brook trout – decrease in Atlantic salmon	• Long term effects of species equilibrium not predictable over the 3 years of monitoring	-
• Vegetation established	• Riparian zone and vegetation functioning	2
• Flow regime comparable to pre-construction	• Flow regime as expected	2
		Overall 2

Figure S2.1.2. *Sample design on translating project results into a common metric for compliance and function.*

Synthesis protocol

1. Rationale of a qualitative synthesis

A Scientific synthesis is normally defined as "The inferential process whereby new models are developed from analysis of multiple data sets to explain observed patterns across a range of time and space scales" (Kemp and Boynton 2011). A synthesis allows combining information from various quantitative and qualitative sources and can include peer-reviewed articles, books but also lectures, reports and other forms of grey literature. Furthermore, a scientific synthesis provides the researcher with the ability to not only combine different sources of data but also to harmonize them by drawing connections between the sources and tackle larger overarching issues or arguments due to the variety of observations and perspectives (Seers 2012).

2. Beginning and Ending Dates of review

October 2017 to July 2018

3. Objectives and questions

Main study questions:

Is high compliance directly linked to high ecosystem function in aquatic offsetting projects?

Record project evaluations:

Assess compliance and function outcomes regarding the respective evaluation method and record project scores or official verdicts on success/ non-success.

Main components of the PICO search:

Population: Offsetting projects in aquatic freshwater systems.

Intervention: Offsetting through creation, enhancement, or restoration.

Control: Presence of clearly stated goals or requirements for the respective project.

Outcome: Monitoring or Evaluation of project success regarding official permit or stated goals.

A secondary question was included in the review to a) complement the primary goal by taking a deeper look into what factors potentially influence project and compliance outcomes and b) to gather supplementary demographic information to further help understand offsetting practices and scope.

Secondary question:

What are potential drivers for compliance and function/ project outcome or 'success?'

Record project demographics and additional information:

- Where? (Project location, target ecosystem)
- When? (Endpoint of construction)
- What? (Project focus and utilized methods)
- How big? (Project size)

4. Criteria for inclusion of studies in the review

Types of Studies

This review did consider studies and databases providing information on compliance and ecosystem function in aquatic offsetting projects. Search period was October and November 2017. Study types included the following:

Scientific databases:

Web of science

Scientific literature like journal articles, abstracts, and conference proceedings. Main advantage is the ability to scan varied data and multidisciplinary research topics in a timely manner using ontology (webofknowledge.com 2018).

Science direct

Large database for peer-reviewed scientific and medical publications and books. Highly customizable search settings and freely available abstracts. Limited information availability through pay-per-view purchases (sciencedirect.com 2018)

Federal Science Library formerly ‘WAVES’

> Database for full text Fisheries and Oceans Canada and Canadian Coast Guard publications from all departments (science-libraries.canada 2018).

Search engines:

Google Scholar – the first 200 hits were screened

Search engine for free academic resources with Customizable search options. Fast processing time but limited by its lack of authority control. Advised to be used for supplementary results but should not be the exclusive source of literature for this review (Notess 2005).

Science.gov – the first 200 hits were screened

Government science information search engine from the United States government. Engine provides access to scientific and technical information from over 60 databases and over 2200 websites collected by government agencies and gives users the ability to target their search through topic clustering (Science.gov 2018).

Other databases and websites:

1. Restore-Rivers
2. Parks Canada
3. ‘DFO’ Fisheries and Oceans Canada
4. United States Army Corps of Engineers
5. ‘EPA’ United States Environmental Protection Agency
6. The international Union of Conservation of Nature
7. NSW Government
8. The Nature Conservancy
9. Fisheries Science Service
10. Natura 2000 EUNIS
11. Bundesamt für Naturschutz
12. European Environmental Agency
13. European Commission

Manual search extension

Accepted articles, books and other publications provided an additional source of subsequent information by going through their reference sections and bibliographies manually.

Types of Participants/ population

Our target populations for this review were aquatic freshwater systems. Marine systems due to their mostly open nature and difficulty to assess outcomes reliably were excluded. Only exceptions were wetlands which were exposed to sometimes brackish intertidal conditions. As for freshwater systems we considered all lentic and lotic forms, natural and anthropogenic engineered.

Types of Intervention/Area of Interest

Goal of this review was to assess the results of offsetting projects in aquatic ecosystems in terms of compensating negative impacts by modifying, enhancing, restoring, or creating habitat, biodiversity, or productivity.

Types of Control

Controlling for measurable effects we decided to focus on projects with a measurable before and after effect. Ecosystem effects like a change in productivity or biomass due to implemented offsetting strategies had to be directly compared to previous assessment or a comparison to natural reference systems in the same area or of the same making and had to be framed by clearly stated goals and targets.

Types of Outcomes

Project outcomes regarding success or failure had to be clearly stated in terms of project compliance and/or resulting overall ecosystem function. Function and compliance were a result of a multitude of different metrics ranging from permit requirements to measured biomass for a target species. Did the proponent fulfill enough permit requirements? Did the implemented methods achieve compliance with the stated goals? Did the methods also provide good ecosystem function? How were other areas affected by this project? We also wanted to collect an array of general metrics for projects when available like project size or construction date.

5. Search Strategy

Different search strategies were implemented to optimize project diversity and prevent potential database, search engine or institutional bias and errors. Using complementary approaches should minimize the chance to overlook or omit relevant information and data. Based on the previously stated inclusion criteria for studies, search terms and Boolean operators were developed.

The first strategy was to assess the mentioned databases (web of science, science direct and federal science library). The search in these databases was conducted on October 3rd 2017. This first sweep was based on the developed PICO search string (Table 1). Search strings were developed after an initial search attempt with a limited set of search terms (Offset, Function, Comply, Compliance, Aquatic) to browse relevant studies and identify potential other relevant phrases and terms.

Table 1. *PICO search String.*

Population	Reservoir OR Lake OR Stream OR River OR Wetland OR Freshwater OR Aquatic OR Pond OR Channel OR Canal OR Creek OR Water*
AND	
Intervention	Compensat* OR Offset* OR Replace* OR Creat* OR Enhance* OR Restor* OR Augment* OR Improve* OR Mitigat* OR Modif*
AND	
Control	Reference OR Control OR Permit OR Goal OR Require* OR Object* OR Focus OR Target OR Compar* OR Guide* OR Natur* OR Pre* OR Post* OR Pristine
AND	
Outcome	Function* OR Comply OR Compliance OR Eco* OR Effect* OR Assess* OR Monitor* OR Stability OR Stable OR Product* OR Biomass OR Habitat OR Comply OR No net loss OR NNL OR State OR Abundan* OR Densit* OR Evaluat* OR Success OR Fail* OR Report*

The second search strategy (October 4th to October 8th) focused on the two big search engines Google Scholar and Science.gov. Timeframe limitations were adopted as to the ones present for the engine. For instance, Science.gov only goes back as far as 1990. However, since offsetting is a rather young practice, we did not see these search limitations as a restriction of major concern. Search terms were used for both google scholar and science.gov. The first 200 hits for each engine were sorted by relevance and screened for their appropriateness regarding the primary and secondary review questions.

A third sweep was done utilizing the additional databases and websites mentioned in section 5. These sites were screened on October 9th and October 11th, 2017. For this search strategy the same inclusion and exclusion criteria applied as for the other sources and were applied by using scrapy.org, an open source and collaborative framework for extracting data and search terms (Offset, Function, Compliance). Where web scraping was not possible, or the site offered an easy to access database on it is own manual searches were conducted. There was no geographic restriction on project location since the aim was to do an across-country assessment, however we limited search results to information available in English and German due to language constraints of the reviewing personnel. The first 30 search page results for each term were screened (90 total), to see if they provide either information or responding links to appropriate studies, reports, and publications.

The fourth search strategy was a manual search extension, which was done between October 3rd and 11th parallel to the 3 prior sweeps and was comprised of hand checking references, links, and bibliographies of the first 3 searches for additional sources or linked studies and reports.

The overall screening process can be found in the flow diagram, depicting the main 3 phases of data acquisition. The overall process can be divided, as seen, into three main parts. Part one was to identify potential sources like relevant databases (N = 3), search engines (N = 2), specialist websites (N = 13) and later subsequent material. Part two included sweeping those sources in the previously described fashion. Potential records underwent an abstract and title screening through EPPI Reviewer (n = 252). Grey literature information and any other records which could not be submitted to EPPI screening were checked manually either through a quick check of reading summary statements or file descriptions or underwent a full text review if the required inclusion or exclusion criteria were not easily assessable. All Manually and EPPI Reviewer included records underwent full text screening to ensure the data provided was usable for our primary and potentially secondary questions (n = 104). Consistency checks were done on October 14th and 15th on 11 relevant records to ensure a uniform understanding of the agreed upon inclusion and exclusion criteria. A full list of all excluded records can be found in at the end of this protocol in Table 3, as well as the reason for their exclusion. A total number of 57 records was included in the final synthesis and split to identify individual projects n = 637.

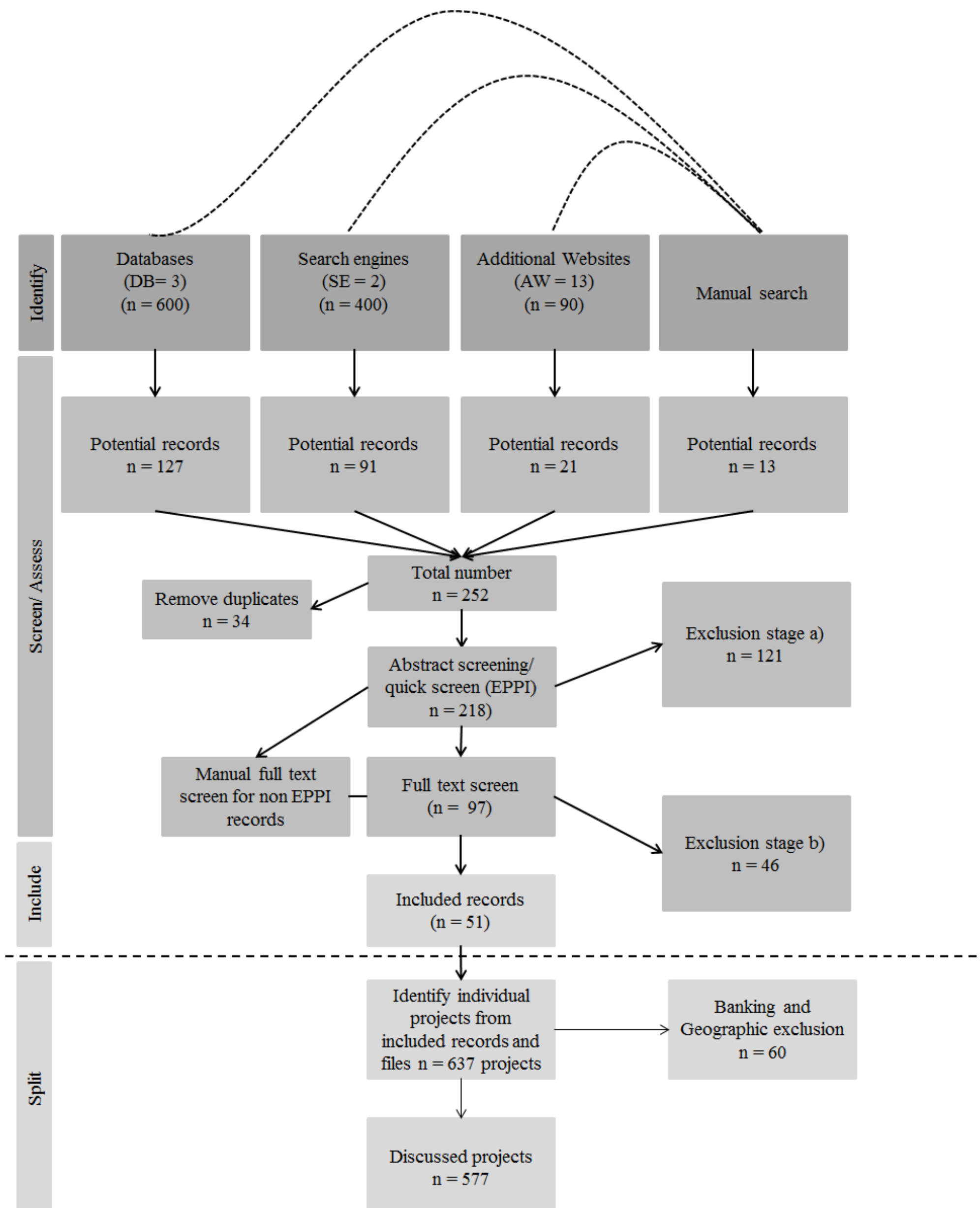


Figure 5. *Search protocol flow diagram, including the four major data acquisition steps.*

6. Assessment of Methodological Quality

Critical Appraisal

Critical appraisal was used to systematically evaluate the included sources and identify strengths and weaknesses of the provided data and results and whether they were useful for this review.

Key points of the critical appraisal were a) appropriateness of the study design in regard to our asked questions and target population and overall credibility, b) methodology/ assessment used in the study and c) if the outcome could be clearly related to compliance or function. Source appraisal verdicts followed the scheme in Table 2, were transferred into the final excel sheet to accompany the included records are noted in the exclusion table. The critical appraisal procedure was based on the AACODS checklist developed at the Flinders University (Tyndall 2010). This approach was chosen and adapted due to the fair amount of present grey literature.

Table 2. *Modified version of the AACODS approach to appraise grey literature.*

AACODS		Yes	No	?
	Identifying who is responsible for the intellectual content. Individual author: • Associated with a reputable organization? • Cited by others? (use Google Scholar as a quick check) Organization or group: • Is the organization reputable? (e.g., W.H.O) • Is the organization an authority in the field? In all cases: • Does the item have a detailed reference list or bibliography?			
Accuracy	• Does the item have a clearly stated aim or brief? • Does it have a stated methodology? • Has it been peer-reviewed? • Has it been edited by a reputable authority? • Supported by authoritative, documented references or credible sources? • Is it representative of work in the field? • If No, is it a valid counterbalance? • Is any data collection explicit and appropriate for the research? If item is secondary material (e.g., a policy brief of a technical report) refers to the original. Is it an accurate, unbiased interpretation or analysis?			
Coverage	All items have parameters which define their content coverage. These limits might mean that a work refers to a particular population group, or that it excluded certain types of publication. A report could be designed to answer a particular question or be based on statistics from a particular survey.			

	Are any limits clearly stated?			
Date	For the item to inform your research, it needs to have a date that confirms relevance • Does the item have a clearly stated date related to content? No easily discernible date is a strong concern. • If no date is given, but can be closely ascertained, is there a valid reason for its absence?			
Significance	This is a value judgment of the item, in the context of the relevant research area • Is the item meaningful? (this incorporates feasibility, utility and relevance) • Does it add context? • Is it integral, representative, typical? • Does it have impact? (in the sense of influencing the work or behavior of others)			

7. Method to Extract Data and synthesis

All records underwent data extraction after their full text review and were filled into an excel form. Extracted and recorded information are explained in Figure 1. General record info included the main author of the record or if not available the corresponding authority, type of record, e.g., paper, government report etc. and the date when it was assessed. The second large part of extracted data was comprised out of project demographics. We noted where the project was located at in terms of country, region, or city and when construction was officially finished. Project size was also included. To estimate a generalize scale effect, all measurements were converted to either ha or km (mostly present for riverine projects). Standardized sizes were consequently converted into 3 different categories, dividing the included offsetting projects in to small, medium, and large ones (small: <0.5 ha/km, medium: 0.5-5 ha/km, large: >5 ha/km). A similar approach was chosen for project system. Project system refers to the aquatic system in which the offsetting measure was undertaken. Lakes and Reservoirs were combined to just ‘Lakes’ as well as brooks and creeks and streams to ‘Streams’. As there is no single scientific definition on how to define these systems, we went with the information given in the records to determine the target ecosystem. In cases, warranted for doubt, we referred to the stream order ($\leq$6 = stream, >6 = river) (Koschitzki 2004; Shreve 1966). Furthermore, a short description was included in the excel file for each individual project to capture its main goal or target. Based on the info found in the literature each project was assigned one or multiple focuses/ targets. These targets were a result of considering a variety of different projects and information on offsetting. 4 main focuses were identified. Projects could either try to reach Productivity goals (measured in weight/area or biomass) for a target species or a whole community. Secondly an offsetting measure could target a specific habitat, trying to create, enhance or restore physical habitat (e.g., pool and riffle creation, spawning shelter, riparian modification). A different focus was the basic function or feature replacement which focus was not to enhance habitat or biomass/ productivity but to replace or restore lost features like ‘creating a 10ha wetland without species or habitat specifications’ or ‘changing flow regime of a river’. Lastly projects could also focus on a preservation aspect in the form of biodiversity or habitat banking, moving the responsibility away from the proponent towards a bank by buying the appropriate number of credits. Banking as a non-traditional approach regarding the offsetting principle was later excluded from the analysis. From the project information provided in the included record we also discerned the methods utilized to achieve the method. As with project focus, after scanning multiple studies and reports, we grouped methods into four main measures: Restoration, enhancement, creation, and banking. Definitions for these methods were based on engineering reports. Restoration was defined as restoring a presently degraded aquatic ecosystem or a feature of it to its original condition (e.g., meander reestablishment in a stream). Enhancements alter ecosystem characteristics to augment conditions mostly in the favor of a specific target species while often at the expense of another species or ecosystem function (e.g., increase water flow in a river section to prevent sedimentation of fine material in spawning areas/ benefiting salmonid species). Creation generally refers to the creation of a completely new ecosystem on a site that did not feature that system before, like creating a whole new wetland or bridging 2 river stretches by a new canal. Creation as per definition is often associated with having only one main goal or major function due to its high costs, intensive management requirements and overall scale. Banking as a method was like

'Project focus' later excluded from the analysis due to its deviation from traditional offsetting measures and the inability to properly evaluate banking projects and their state.

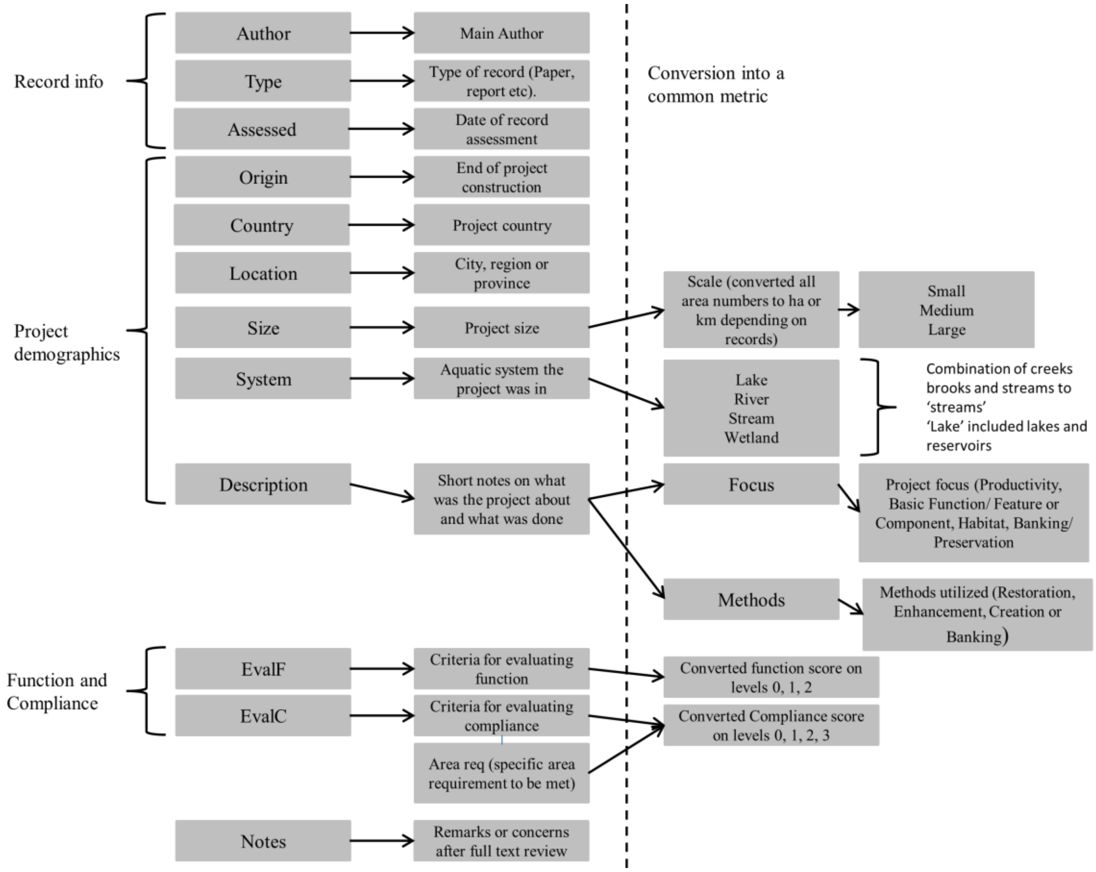


Figure 6. *Extracted project information and processing.*

To evaluate project compliance and ecosystem function we used a two-step approach. First step was to identify which method was used to assess compliance or function. Evaluations included a wide array of methods from official permit requirements to a specific biomass for a target fish species to rapid assessment methods. In both cases results could be presented in a qualitative as well as quantitative way. Qualitative outcomes for compliance or function were double checked with their respective sources and records to determine if they were based on reliable assessments and reports if actual numerical data was not readily available, following the procedure shown in Figure 3. Step two was to transform function and compliance results into numerical values to allow for a synthesized evaluation later. For instance, if a project had 8 permit requirements and met 5 out of 8 according to the report than it was noted down as 62.5% for complying with those. The goal was to ultimately develop levels of compliance and function as shown in Figure 2. Specific compliance area requirements were evaluated as a separate score as shown in Figure 2, as they were also handled separately in permits and reports besides other permit requirements.

Function	**Compliance**
2 *Full functionality:* Ecosystem is functioning as desired.	**3** *Exceeded compliance:* Exceeds requirements, e.g. suggested project size (>110%).
1 *Partial functionality*: Meets some of the assessment criteria.	**2** *Full compliance:* All requirements are fulfilled (>90%).
0 *No functionality:* Little to none of the desired ecosystem functions are met.	**1** *Partial compliance:* Some requirements are met, others are lacking (25-90%).
	0 *Non-compliant:* Little to none of the requirements are met (0-25%).

Compliance and function criteria in the different studies:

Function criteria are based on assessed ecosystem functions related to reference systems or values:

- Rapid assessment methods
- Water quality
- Hydrogeomorphology
- Chemical processes
- Target species (Biomass, NNL)
- Habitat scores

Compliance criteria are based on permit/ project requirements:

- Size
- Habitat specifications
- Chemical processes
- Target Species (Biomass, NNL)
- Reports
- Monitoring
- Invasive species

Figure 7. *Function and compliance common metric and scoring system plus criteria in different studies.*

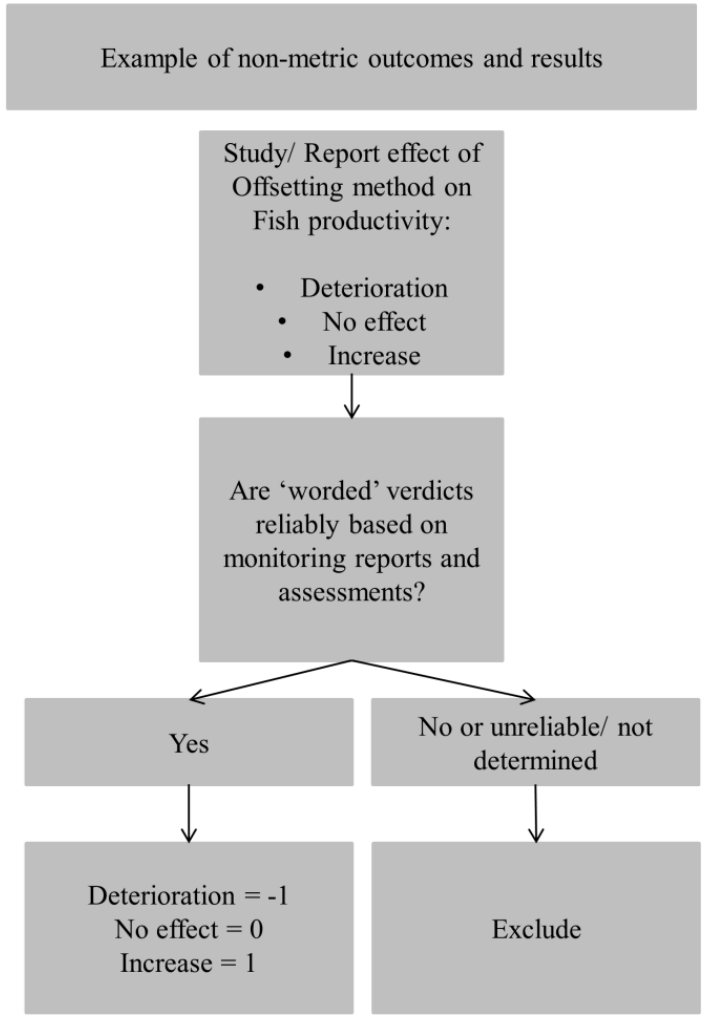


Figure 8. *Inclusion or exclusion of qualitative data for later analysis.*

Chapter 2.2 Appendices

Table A2.2.1. Types of offsets used in cases of residual or chronic harm in aquatic ecosystems. Results and classifications are based on information from the systematic review and meta-analysis as indicated by the 'sources; column.

Type	Subtype	Measure	Associated benefits/ goals	Sources
Habitat creation	Off channel habitat creation	Side-channel creation Overwintering pond creation	Spawning habitat provision, rearing habitat provision, overwintering habitat	Giannico & Hinch 2003; Henning et al. 2006; McMichael et al. 2005

				Morley et al. 2005; Roni et al. 2006; Rosenfeld et al. 2008; Scruton et al. 2006; Ward et al. 1999
Habitat restoration and enhancement	R-Restoration	(Riparian) restoration, Rehabilitation	Buffer zone creation, reduction of environmental impacts, food availability, habitat coupling	Auer et al. 2017; Baletto & Teal 2011; Barnthouse et al. 2019; Keeley et al. 1996; May & Lee 2004; Morley et al. 2005; Roni et al. 2010; Taylor et al. 2019
	Structure and Cover	Bank stabilization Riparian heterogeneity LWD & Logjams Boulders Pools & Riffles	In-stream habitat provision (shelter, food availability), changes in flow regime, and peak flow times/ intervals	
	Connectivity	Dam and barrier removal Fish Passage Reconnection (Floodplain…)	Lateral & longitudinal habitat connection, migration corridors, nutrient and sediment exchange, and transport, flow regime	
Bio and Chemical Manipulation	Stocking	Stocking Re-introduction Translocation	Direct addition of individuals and biomass, Potential increase in Productivity	Antonio Agostino et al. 2010; Barnthouse et al. 2019; Brown et al. 2013; Holmes 2018; Koenings et al. 2000; Kohler et al. 2012; Margenau 1992; Michaud 2000; Naiman et al. 2002; Sierp et al. 2009; Unwin & Gabrielson 2018; Wipfli et al. 2010
	Nutrients	Nutrient enrichment	Productivity boost for biotic production, compensation for nutrient loss through lack of anadromous fish/ carcasses	

Table A2.2.2. Fixed effect model (Jackson Method) for estimating cumulated pooled effect size for habitat creation measures and study heterogeneity (accepted Alpha < 0.05). Between study variance estimated through 'DerSiminian-laird' estimator for groups with low heterogeneity.

Habitat creation	SMD	95% CI	z-value	p-value
Fixed effect model	0.7959	[0.4728; 1.1191]	4.83	<0.0001

Heterogeneity	Q	df	p-value
	6.47	8	0.5950

Table A2.2.3. Random effect model (Hartung-Knapp adjustment) for estimating cumulated pooled effect size for habitat restoration measures and study heterogeneity (accepted Alpha < 0.05). Between study variance estimated through 'restricted maximum likelihood estimator for groups with low heterogeneity.

Habitat restor	SMD	95% CI	t-value	p-value
Random effect model	0.6595	[0.2990; 1.0200]	3.95	0.0017
Heterogeneity	Q	df	p-value	
	31.61	13	0.0027	

Table A2.2.4. Random effect model (Hartung-Knapp adjustment) for estimating cumulated pooled effect size for biological and chemical manipulation measures and study heterogeneity (accepted Alpha < 0.05). Between study variance estimated through 'restricted maximum likelihood estimator for groups with low heterogeneity.

Bio Man	SMD	95% CI	t-value	p-value
Random effect model	0.5146	[-0.1790; 1.2083]	1.82	0.1194
Heterogeneity	Q	df	p-value	
	26.12	6	0.0002	

Table A2.2.5. General Success Score (GSS) corresponding to fully meeting project goals and targets (Score = 2), partially meeting stated goals and targets (Score = 1) or not meeting offset goals and targets (Score = 0) tested for homogeneity of variance regarding GSS to Pre-assessment, Years monitored and Site. Alpha values < 0.05 indicate unequal variances.

Levene	Df	F-value	Pr (>F)
Group	13 16	0.8889	0.5794

Table A2.2.6. General Success Score (GSS) corresponding to fully meeting project goals and targets (Score = 2), partially meeting stated goals and targets (Score = 1) or not meeting offset goals and targets (Score = 0) tested normality through Shapiro Wilk test. Alpha values < 0.05 indicate non-normal distribution.

Shapiro Wilk	W	p-value
	0.79831	6.108e-05

Table A2.2.7. Kruskal-Wallis test accounting for non-normal data and unequal variances and pairwise comparison for General Success Score (GSS) and offset location (onsite; offsite; both). Accepted alpha of < 0.05.

Success Score	Chi-squared	df	p-value
Site (on/off)	0.67163	2	0.7148

	Both	Offsite
Offsite	0.72	-
Onsite	0.72	0.72

Table A2.2.8. Kruskal-Wallis test accounting for non-normal data and unequal variances and pairwise comparison for General Success Score (GSS) and whether studies had a pre-assessment associated with them (yes; no). Accepted alpha of < 0.05.

Success Score	Chi-squared	df	p-value
Pre-assessment	7.2548	1	0.007071

Table A2.2.9. Kruskal-Wallis test accounting for non-normal data and unequal variances and pairwise comparison for General Success Score (GSS) and monitoring timeframes (< 4 years; 4 to 6 years; > 6 years). Accepted alpha of < 0.05.

Success Score	Chi-squared	df	p-value
Years	7.5859	2	0.02253

	Less than 4 years	4 to 6 years
4 to 6 years	0.043	-
More than 6 years	0.059	0.702

Chapter 2.2 Supplements

Table S2.2.1. Critical appraisal to assess project validity based on study design and assessment bias.

Category no/ bias/ data quality feature	Specific data feature	Study design	Score	Validity
1. Selection and performance bias: study design	Design	BACI	NA	High
		BA, CI, or Incomplete BACI	NA	Medium
		BA comparison (> 3 before, > 3 after)	25	NA
	Temporal repetition	BA comparison (< 3 before, > 3 after)	20	NA
		BA comparison (> 3 before, < 3 after)	15	NA
		BA comparison (< 3 before, < 3 after)	10	NA
		Deficient BA	5	NA
		No BA	0	NA
	Spatial repetition	Site comparison/CI > 2 control and impact)	25	NA
		Site comparison/CI < 2 control, > 2 impact)	20	NA
		Site comparison/CI > 2 control, < 2 impact)	15	NA
		Site comparison/CI < 2 control, < 2 impact)	10	NA
		Deficient CI	5	NA
		No CI	0	NA
2. Assessment bias	Measured Outcome	Quantitative	NA	High
		Quantitative estimate	NA	Medium
		Semi quantitative	NA	Low
	Monitoring	Frequent Mon	NA	High
		1 time Mon	NA	Medium
		No Mon	NA	Low

Temp + Spat score		
	30 to 50	High
	20 to < 30	Medium

0 to < 20	Low

Table S2.2.2. Systematic review and meta-analysis protocol.

1. Search Strategy for meta-analysis PI(E)CO search criteria were used to define the important aspects (James et al. 2016)

Early screening articles were specifically referring to the concrete negative impacts on fish populations or the causation e.g., flow alteration. However, the key focus of the review is that residual fish mortality is present on a temporal scale, happens in certain intervals or persists after construction of the development project is done. Thus:
P – Fish populations I (E) - must lead to mortality or serious harm C – pre-assessment comparing offset to O – negative impact must be offset plus recorded method and outcome put in terms of productivity, abundance, condition, diversity, or biomass

2. Search terms Search terms were based on the results of screening and extracting keywords from several scientific and grey literature documents covering the topic of fish mortality in regard to human development projects.
Fish, Fisheries, Productivity, Habitat, Offset, Measures, Report, Residual, Mortality, Canada, Monitoring, No net loss, Ecosystem, Aquatic, Effort, Development, Creation, Restoration, Temporal, Nutrient, Addition, Chemical, Restoration, Alter, Increase, Policy, Net, Effective, Commercial, Recreational, Mitigation, Banking, Avoid, Practice, Negative, Mitigate, Outcome, Maintaining, Priority, Reducing, Relocation, Ocean, Unavoidable, Ongoing, Manage, Sustainable, Techniques, Stocking, Dam, Passage, Downstream, Oxygen, Discharge, Electrical, Power, Turbine, Energy, Disturbance, Salmon, Young, Juvenile, Size, Chronic, abundance, Fishway, Spawner, Shutdown, Turbine, Growth, Species, Preservation, Food, Nutrients, Insects, Invertebrates, Resources, Population, Abundance, Water, Flow, Discharge, Speed, Velocity
Keywords were extracted using R and the packages ‘slowraker’ ‘udpipe’ and ‘textrank’ and the ‘rake’ command which part of ‘rakeR’ R (Jones 2017).

3. Boolean Search String
Boolean search terms were formulated and used for Web of Science, Google Scholar and to some extent the web-scraper. All searches were streamlined through publish or perish software. (Fish* OR Spawn* OR Juvenile OR Salmon* OR Young OR Species) AND (Aquatic OR Stream OR River* OR Lake OR Impound* OR Pond OR Reserv* OR Ocean OR Coast* OR Eco* OR Lentic OR Lotic OR Marine OR Freshwater)

AND
(Mortality OR Kill* OR Harm OR Injur*)
AND
(Tempor* OR Residual OR Remain* OR Continu*)
AND
(Develop* OR Anthropo* OR Industr* OR Farm* OR Construct* OR Power OR Turbine OR Electric*)
AND
(Offset* OR Compensat* OR Mitigat* OR Reduc* OR Reverse OR NNL OR No Net Loss)
AND
(Creat* OR Restor* OR Enhanc* OR Preserv* OR Bank* OR Credit)
AND (Product* OR Biomass OR Abundan* OR Biodiversity OR Divers*)
AND (Method* OR Polic* OR Outcome OR Report* OR Manag* OR Monitor* OR Practice)

4. Search output
Google Scholar First 200 search results screened sorted by relevance August 17th Web of Science Full search using the Boolean search terms – 181 results August 19th Grey literature Any other listed websites, parsed through the web-scraper in combination with Boolean search terms 1. Alberta Hydro – 2654 parsed sites, 0 hits 2. US Corps of Engineers – split into regional divisions – 25199 parsed sites, 79 collected pages 3. Fisheries and Oceans Canada – Waves – 86722 parsed sites, 28 collected pages 4. NOAA – 12872 parsed sites, 321 collected pages 5. Google Scholar extended – 78798 parsed sites, 140 collected pages

5. Literature assessment and scan
Title and abstract screening Google Scholar: 29 documents and papers saved as pdfs out of 200 Web of Science: 181 – 35 documents and papers saved as pdfs Combined other websites: 34 papers and documents saved as pdfs Body of literature Total: 98 documents - 30 with usable data – validity assessment (Table S1)

BARNTHOUSE ET AL.	2019	MORLEY ET AL.	2005
CLARKE ET AL.	2008	PATRICK ET AL.	2015
GIBEAU ET AL.	2020	RAYMOND	1988
GREENWOOD	2008	RONI ET AL.	2006
HADDERINGH & JAGER	2002	RONI ET AL.	2010
HANSEN ET AL.	2017	ROSENFELD ET AL.	2008
HARVEY ET AL.	1998	SCRIVENER & BROWNLEE	1988
HIGGINS & BRADFORD	1996	SCRUTON ET AL.	2005
HITT ET AL.	2012	SKALSKI ET AL.	2016
HORNE ET AL.	2004	STANTEC	2017
HUNT ET AL.	2012	STOCKNER & MACISAAC	1996
KEELEY ET AL.	1996	THOMAS ET AL.	2013
KNIGHT PIÉSOLD LTD.	2015	TONALLA ET AL.	2017
LEMLY	2010	UNWIN & GABRIELSON	2018
MAES ET AL.	2004	YOUNG ET AL.	2011

Table S2.2.3. Project information divided into offsets, methods, common metric, and study validity based on table S1.

ID	Offset	Method	Metric	Study Validity	Effect size	Species	Sites Reference/ Treatment
C1	Habitat Creation	Off channel construction	Density	High	0.76137	Coho	7/7
C2	Habitat Creation	Off channel construction	Density	High	0.87095	Coho, Chum, Chinook	11/11
C3	Habitat Creation	Off channel construction	Density	High	0.22857	Coho	9/10
C4	Habitat Creation	Pond & F-plain	Density	Medium	1.65517	Coho, Steelhead	3/5
C5	Habitat Creation	Pond & F-plain	Density	Medium	0.81937	Coho, Steelhead	10/10
C6	Habitat Creation	Channel & Flow	Biomass	High	2.35467	Brook Trout	3/3
C7	Habitat Creation	Off channel construction	Density	Medium	0.52446	Chum, Coho	24/24

C8	Habitat Creation	Off channel construction	Survival	Medium	1.56256	Coho	24/24
C9	Habitat Creation	Floodplain & Channel	Density (smolt production)	Low	0.66032	Coho	5/11
R1	Restoration and Enhancement	Culvert removal	Density	Medium	0.64027	Community	6/6
R2	Restoration and Enhancement	LWD introduction	Density	Medium	0.39474	Salmonids	58/58
R3	Restoration and Enhancement	Boulder Weir	Density	Medium	1.38985	Salmonids	12/12
R4	Restoration and Enhancement	Logjam	Density	Medium	0.59831	Salmonids	24/24
R5	Restoration and Enhancement	Substrate	Density	Medium	0.69442	Salmonids	3/13
R6	Restoration and Enhancement	Dam removal	Density	Low	-0.6022	Community	3/13
R7	Restoration and Enhancement	Dam removal	Biomass	Low	-0.0157	Community	3/3
R8	Restoration and Enhancement	Dam removal	Density	Medium	0.59615	Eel	16/25
R9	Restoration and Enhancement	Dam removal	Density	Medium	1.21918	Eel	15/15
R10	Restoration and Enhancement	Dam removal	Density	Medium	1.32235	Eel	15/15
R11	Restoration and Enhancement	Dam removal	Density	Medium	0.85944	Eel	15/15
R12	Restoration and Enhancement	Dam removal	Richness	Medium	-0.2812	Community	15/15
R13	Restoration and Enhancement	Dam removal	Richness	Medium	0.71623	Community	5/5

R14	Restoration and Enhancement	Riverbank enrichment	Temperature	Low	1.47122	Small-bodied fish	3/3
B1	Bio and Chemical Manipulation	Model Stocking	Survival	Medium	0.73521	Salmonids	40/40
B2	Bio and Chemical Manipulation	Stock vs Wild	Survival	Medium	-0.54608	Salmonids	21/19
B3	Bio and Chemical Manipulation	Stock vs Wild	Survival	Medium	-0.40901	Salmonids	21/18
B4	Bio and Chemical Manipulation	Nutrient enrichment	Biomass	High	1.12581	Salmonids	10/10
B5	Bio and Chemical Manipulation	Model Stocking	Survival	Medium	1.18182	Salmonids	8/8
B6	Bio and Chemical Manipulation	Stock vs Wild	Survival	Low	0.69298	Salmonids	3/3
B7	Bio and Chemical Manipulation	Nutrient enrichment	Biomass/ environmental variables	High	0.94533	Communit y	5/5

Table S2.2.4. Full document list of all 98 reviewed documents used for the review, meta-analysis, and for informational value.

ID	TITLE	AUTHOR	YEAR
1	EFFECTS OF FLUSHING SPENCER HYDRO ON WATER QUALITY, FISH, AND INSECT FAUNA IN THE NIOBRARA RIVER, NEBRASKA	LARRY W. HESSE AND BRAD A. NEWCOMB	1982
2	POWER PLANT IMPACT ASSESSMENT A SIMPLE FISHERY PRODUCTION MODEL APPROACH	ALEC D . MACCALL, KEITH R . PARKER, RONALD LEITHISERAND BILL JESSEE	1983
3	PRODUCTION FORGONE: AN ALTERNATIVE METHOD FOR ASSESSING THE CONSEQUENCES OF FISH ENTRAINMENT AND IMPINGEMENT LOSSES AT POWER PLANTS AND OTHER WATER INTAKES	PAUL J. RAGO	1984
4	POPULATION DYNAMICS OF LAKE WHITEFISH (COREGONUS CLUPEAFORMIS) DURING AND AFTER	K. H. MILLS AND S. M.CHALANCHUK	1986

	THE FERTILIZATION OF LAKE 226, THE EXPERIMENTAL LAKES AREA		
5	MORTALITY OF FISH PASSING THROUGH TIDAL, LOW-HEAD HYDROPOWER TURBINES AND POSSIBLE MITIGATION STRATEGIES	ROGER A. RULIFSON' AND MICHAEL J. DADSWELL'	1987
6	EFFECTS OF HYDROELECTRIC DEVELOPMENT AND FISHERIES ENHANCEMENT ON SPRING AND SUMMER CHINOOK SALMON AND STEELHEAD IN THE COLUMBIA RIVER BASIN	HOWARD L. RAYMOND	1988
7	GAS SUPERSATURATION AND GAS BUBBLE TRAUMA IN FISH DOWNSTREAM FROM A MIDWESTERN RESERVOIR	DONNA SCHULZE LUTZ	1995
8	AN EVALUATION OF INTRODUCED WOODY DEBRIS BUNDLES TO INCREASE SUMMER DENSITIES OF JUVENILE COHO SALMON IN THE MAINSTEAM CLEARWATER RIVER	R. PETERS, E. KNUDSEN, G. PAULEY AND J. CEDERHOLM	1996
9	ESTIMATES OF PRODUCTION BENEFITS FOR SALMONID FISHES FROM STREAM RESTORATION INITIATIVES	E.R. KEELEY, P.A. SLANEY AND D. ZALDOKAS	1996
10	BRITISH COLUMBIA LAKE ENRICHMENT PROGRAMME: TWO DECADES OF HABITAT ENHANCEMENT FOR SOCKEYE SALMON	JOHN G. STOCKNER AND ERLAND A. MACISAAC	1996
11	EVALUATION OF THE CONSTRUCTION OF ARTIFICIAL FLUVIAL SALMONID HABITAT IN A HABITAT COMPENSATION PROJECT, NEWFOUNDLAND, CANADA	DAVID SCRUTON	1996
12	FISH HABITAT REHABILITATION PROCEDURES	P.A. SLANEY AND D. ZALDOKAS	1997
13	TECHNIQUES TO EVALUATE THE EFFECTIVENESS OF FISH HABITAT RESTORATION WORKS IN STREAMS IMPACTED BY LOGGING ACTIVITIES	C. WENDELL KONING, MARC N. GABOURY, MICHAEL D. FEDUK, AND PAT A. SLANEYR	1998
14	IMPOUNDMENT AND INTRODUCTIONS: THEIR IMPACTS ON NATIVE FISH OF THE UPPER WAIPORI RIVER, NEW ZEALAND	RICHARD M. ALLIBONE	1999
15	ASSESSMENT OF TECHNIQUES FOR RAINBOW TROUT TRANSPLANTING AND HABITAT MANAGEMENT IN BRITISH COLUMBIA	GORDON HARTMAN AND MICHAEL MILES	2001
16	FISH PASSAGE THROUGH CULVERTS, ROCK WEIRS AND ESTUARINE OBSTRUCTIONS	M. LARINIER	2002

17	GOOD DAMS AND BAD DAMS: ENVIRONMENTAL CRITERIA FOR SITE SELECTION OF HYDROELECTRIC PROJECTS	GEORGE LEDEC, JUAN DAVID QUINTERO	2003
18	SALMON ON THE EDGE	DEREK MILLS	2003
19	IMPACTS OF HYDRO-DAMS, IRRIGATION SCHEMES AND RIVER CONTROL WORKS	ROGER YOUNG, GRAEME SMART AND JON HARDING	2003
20	STREAMBANK PROTECTION WITH RIP-RAP: AN EVALUATION OF THE EFFECTS ON FISH AND FISH HABITAT	J.T. QUIGLEY AND D.J. HARPER	2004
21	SIMULATING EFFECTS OF HYDRO-DAM ALTERATION ON THERMAL REGIME AND WILD STEELHEAD RECRUITMENT IN A STABLE-FLOW LAKE MICHIGAN TRIBUTARY	BRAD D. HORNE, EDWARD. S. RUTHERFORD AND KEVIN E. WEHRLY	2004
22	HOW FINE SEDIMENT IN RIVERBEDS IMPAIRS GROWTH AND SURVIVAL OF JUVENILE SALMONIDS	KENWYN B. SUTTLE, MARY E. POWER, JONATHAN M. LEVINE, AND CAMILLE MCNEELY	2004
23	PREDICTING BENEFITS OF SPAWNING-HABITAT REHABILITATION TO SALMONID (ONCORHYNCHUS SPP.) FRY PRODUCTION IN A REGULATED CALIFORNIA RIVER	JOSEPH E. MERZ, JOSE D. SETKA, GREGORY B. PASTERNACK, AND JOSEPH M. WHEATON	2004
24	EFFECTS OF THE FEDERAL COLUMBIA RIVER POWER SYSTEM ON SALMONID POPULATIONS	JOHN G. WILLIAMS, STEVEN G. SMITH, RICHARD W. ZABEL, WILLIAM D. MUIR, MARK D. SCHEUERELL, BENJAMIN P. SANDFORD, DOUGLAS M. MARSH, REGAN A. MCNATT, AND STEPHEN ACHORD	2005
25	JUVENILE SALMONID (ONCORHYNCHUS SPP.) USE OF CONSTRUCTED AND NATURAL SIDE CHANNELS IN PACIFIC NORTHWEST RIVERS	SARAH A. MORLEY, PATRICIA S. GARCIA, TODD R. BENNETT, AND PHILIP RONI	2005
26	A CASE STUDY OF HABITAT COMPENSATION TO AMELIORATE IMPACTS OF HYDROELECTRIC DEVELOPMENT: EFFECTIVENESS OF RE-WATERING AND HABITAT ENHANCEMENT OF AN INTERMITTENT FLOOD OVERFLOW CHANNEL	D. A. SCRUTON, K. D. CLARKE, M. M. ROBERGE, J. F. KELLY AND M. B. DAWE	2005
27	BLASTING EFFECTS ON INCUBATING SALMONID EGGS	S. FAULKNER	2006
28	NUCLEAR POWER IN CANADA: AN EXAMINATION OF RISKS, IMPACTS AND SUSTAINABILITY	MARK WINFIELD, ALISON JAMISON, RICH WONG, PAULINA CZAJKOWSKI	2006

29	COHO SALMON SMOLT PRODUCTION FROM CONSTRUCTED AND NATURAL FLOODPLAIN HABITATS	PHIL RONI,* SARAH A. MORLEY, AND PATSY GARCIA	2006
30	STATUS AFTER 5 YEARS OF SURVIVAL COMPLIANCE TESTING IN THE FEDERAL COLUMBIA RIVER POWER SYSTEM (FCRPS)	JOHN R. SKALSKI, MARK A. WEILAND, KENNETH D. HAM, GENE R. PLOSKEY, GEOFFREY A. MCMICHAEL, ALISON H. COLOTELO, THOMAS J. CARLSON, CHRISTA M. WOODLEY, M. BRAD EPPARD & ERIC E. HOCKERSMITH	2016
31	EFFECT OF OPERATIONAL CHANGES IN REDUCING FISH IMPINGEMENT AT A POWER PLANT IN OHIO, USA	PAUL HENRY PATRICK, ELAINE MASON, DARLENE AGER AND SCOTT BROWN	2015
32	COMPARISON OF FISH IMPINGEMENT BY A THERMAL POWER STATION WITH FISH POPULATIONS IN THE EMS ESTUARY	R. H. HADDERINGH & Z. JAGER	2002
33	FIELD EVALUATION OF A SOUND SYSTEM TO REDUCE ESTUARINE FISH INTAKE RATES AT A POWER PLANT COOLING WATER INLET	J. MAES, A. W. H. TURNPENN, D. R. LAMBERT, J. R. NEDWELL, A. PARMENTIER AND F. OLLEVIER	2004
34	EVALUATION OF A LARGE-SCALE FISH SALVAGE TO REDUCE THE IMPACTS OF CONTROLLED FLOW REDUCTION IN A REGULATED RIVER	P. S. HIGGINS & M. J. BRADFORD	1996
35	ASSESSING DEMOGRAPHIC EFFECTS OF DAMS ON DIADROMOUS FISH: A CASE STUDY FOR ATLANTIC SALMON IN THE PENOBSCOT RIVER, MAINE	JULIE L. NIELAND, TIMOTHY F. SHEEHAN, AND RORY SAUNDERS	2015
36	A FRAMEWORK FOR ASSESSING FISHERIES PRODUCTIVITY FOR THE FISHERIES PROTECTION PROGRAM	M.J. BRADFORD, R.G. RANDALL, AND K.S. SMOKOROWSKI, B.E. KEATLEY AND K.D. CLARKE	2013
37	METRICS FOR ASSESSING FISHERIES PRODUCTIVITY AND OFFSETTING STRATEGIES UNDER CANADA'S NEW FISHERIES ACT	KAREN K. CHRISTENSEN-DALSGAARD, R. NILOSHINI SINNATAMBY AND MARK POESCH	2014
38	EVOLUTIONARY RESPONSES BY NATIVE SPECIES TO MAJOR ANTHROPOGENIC CHANGES TO THEIR ECOSYSTEMS: PACIFIC SALMON IN THE COLUMBIA RIVER HYDROPOWER SYSTEM	ROBIN S. WAPLES, RICHARD W. ZABEL MARK D. SCHEUERELL AND BETH L. SANDERSON	2007
39	SURFACE FLOW OUTLETS TO PROTECT JUVENILE SALMONIDS PASSING THROUGH HYDROPOWER DAMS	GARY E. JOHNSON & DENNIS D. DAUBLE	2007
40	EFFECTS OF IMPOUNDMENT ON NUTRIENT AVAILABILITY AND PRODUCTIVITY IN LAKES	ANDREAS MATZINGER, ROGER PIETERS, KEN I. ASHLEY, GREGORY A. LAWRENCE, ALFRED WUEST	2007

41	REPRODUCTIVE SUCCESS OF CAPTIVE-BRED STEELHEAD TROUT IN THE WILD: EVALUATION OF THREE HATCHERY PROGRAMS IN THE HOOD RIVER	HITOSHI ARAKI, WILLIAM R. ARDREN, ERIK OLSEN, BECKY COOPER, AND MICHAEL S. BLOUIN	2007
42	VALIDATION OF THE FLOW MANAGEMENT PATHWAY: EFFECTS OF ALTERED FLOW ON FISH HABITAT AND FISHES DOWNSTREAM FROM A HYDROPOWER DAM.	KEITH D. CLARKE, THOMAS C. PRATT, ROBERT G. RANDALL, DAVE A. SCRUTON, KAREN E. SMOKOROWSKI	2008
43	EFFECTS OF SIDE CHANNEL STRUCTURE ON PRODUCTIVITY OF FLOODPLAIN HABITATS FOR JUVENILE COHO SALMON	JORDAN S. ROSENFELD, ELIZABETH RAEBURN, PATRICK C. CARRIER AND RACHEL JOHNSON	2008
44	THE USE OF ADVANCED HYDROELECTRIC TURBINES TO IMPROVE WATER QUALITY AND FISH POPULATIONS	G.F. CADA1, P.A. BROOKSHIER2, J.V. FLYNN3, B.N. RINEHART4, G.L. SOMMERS4, AND M.J. SALE1	2008
45	VALIDATION OF THE FLOW MANAGEMENT PATHWAY: EFFECTS OF ALTERED FLOW ON FISH HABITAT AND FISHES DOWNSTREAM FROM A HYDROPOWER DAM	KEITH D. CLARKE, THOMAS C. PRATT, ROBERT G. RANDALL, DAVE A. SCRUTON, KAREN E. SMOKOROWSKI	2008
46	COMBINING TURBINE BLADE-STRIKE AND LIFE CYCLE MODELS TO ASSESS MITIGATION STRATEGIES FOR FISH PASSING DAMS	JOHN W. FERGUSON, GENE R. PLOSKEY, KJELL LEONARDSSON, RICHARD W. ZABEL, AND HANS LUNDQVIST	2008
47	FISH MORTALITY BY IMPINGEMENT ON THE COOLING-WATER INTAKE SCREENS OF BRITAIN'S LARGEST DIRECT-COOLED POWER STATION	M.F.D. GREENWOOD	2008
48	HYDROELECTRICITY AND FISH: A SYNOPSIS OF COMPREHENSIVE STUDIES OF UPSTREAM AND DOWNSTREAM PASSAGE OF ANADROMOUS WILD ATLANTIC SALMON, SALMO SALAR, ON THE EXPLOITS RIVER, CANADA	D. A. SCRUTON, C. J. PENNELL, C. E. BOURGEOIS, R. F. GOOSNEY, L. KING, R. K. BOOTH, W. EDDY, T. R. PORTER, L. M. N. OLLERHEAD, K. D. CLARKE	2008
49	LONG-TERM BROWN TROUT POPULATIONS RESPONSES TO FLOW MANIPULATION	CATHERINE SABATON, YVES SOUCHON, HERVE CAPRA, VERONIQUE GOURAUD, JEAN-MARC LASCAUXAND LAURENCE TISSOT	2008
50	LONG-TERM EFFECTS OF HYDROPOWER INSTALLATIONS AND ASSOCIATED RIVER REGULATION ON RIVER SHANNON EEL POPULATIONS: MITIGATION AND MANAGEMENT	T. K. MCCARTHY, P. FRANKIEWICZ, P. CULLEN, M. BLASZKOWSKI, W. O'CONNOR , D. DOHERTY	2008
51	RATE OF BIOTIC COLONIZATION FOLLOWING FLOW RESTORATION BELOWA DIVERSION DAM IN THE BRIDGE RIVER, BRITISH COLUMBIA	A. SCOTT DECKER, MICHAEL J. BRADFORD AND PAUL S. HIGGINS	2008

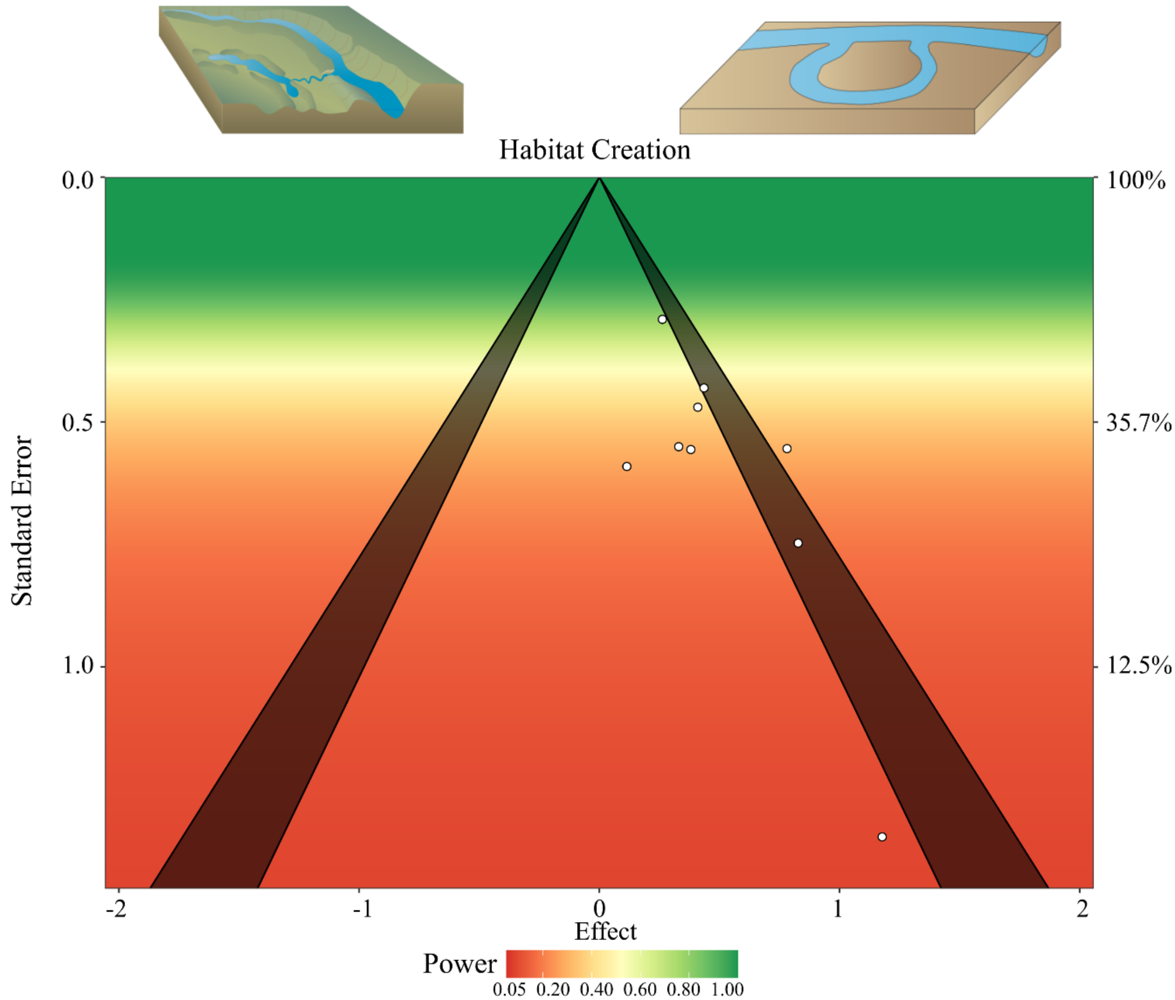
52	COUNTERINTUITIVE RESPONSES OF FISH POPULATIONS TO MANAGEMENT ACTIONS: SOME COMMON CAUSES AND IMPLICATIONS FOR PREDICTIONS BASED ON ECOSYSTEM MODELING	WILLIAM E. PINE, III STEVEN J. D. MARTELL, CARL J. WALTERS, AND JAMES F. KITCHELL	2009
53	CREATING LAKES FROM OPEN PIT MINES: PROCESSES AND CONSIDERATIONS, EMPHASIS ON NORTHERN ENVIRONMENTS	CHRISTOPHER H. GAMMONS, LES N. HARRIS JAMES M. CASTRO PETER A. COTT BRUCE W. HANNA	2009
54	USING AN UNPLANNED EXPERIMENT TO EVALUATE THE EFFECTS OF HATCHERIES AND ENVIRONMENTAL VARIATION ON THREATENED POPULATIONS OF WILD SALMON	ERIC R. BUHLE, KIRSTIN K. HOLSMAN, MARK D. SCHEUERELL, ANDREW ALBAUGH	2009
55	A WHITE PAPER ON ENVIRONMENTAL DAMAGE FROM COAL COMBUSTION WASTE: THE COST OF POISONED FISH AND WILDLIFE	A. DENNIS LEMLY	2010
56	HYDRAULIC ASSESSMENT OF ENVIRONMENTAL FLOW REGIMES TO FACILITATE FISH PASSAGE THROUGH NATURAL RIFFLES: SHOALHAVEN RIVER BELOW TALLOWA DAM, NEW SOUTH WALES, AUSTRALIA	IVARS REINFELDS, MARCUS LINCOLN-SMITH, TIM HAEUSLER, DAVID RYAN AND IVOR GROWNS	2010
57	ESTIMATING CHANGES IN COHO SALMON AND STEELHEAD ABUNDANCE FROM WATERSHED RESTORATION: HOW MUCH RESTORATION IS NEEDED TO MEASURABLY INCREASE SMOLT PRODUCTION?	PHILIP RONI, GEORGE PESS, TIM BEECHIE, AND SARAH MORLEY	2010
58	FACTORS REGULATING THE DOWNSTREAM MIGRATION OF MATURE EELS (ANGUILLA SPP.) AT ANIWHENUA DAM, BAY OF PLENTY, NEW ZEALAND	JACQUES A. BOUBÉE , CHARLES P. MITCHELL , BENJAMIN L. CHISNALL , DAVE W. WEST , EDDIE J. BOWMAN & ALEX HARO	2010
59	DIRECT AND INDIRECT ESTIMATES OF THE PRODUCTIVE CAPACITY OF FISH HABITAT UNDER CANADA'S POLICY FOR THE MANAGEMENT OF FISH HABITAT: WHERE HAVE WE BEEN, WHERE ARE WE NOW, AND WHERE ARE WE GOING?	CHARLES K. MINNS, ROBERT G. RANDALL, KAREN E. SMOKOROWSKI, KEITH D. CLARKE, ANTONIO VÉLEZ-ESPINO, ROBERT S. GREGORY, SIMON COURTENAY, AND PATRICE LEBLANC	2010
60	BIOLOGICAL RESPONSES TO LIMING IN BOREAL LAKES: AN ASSESSMENT USING PLANKTON, MACROINVERTEBRATE AND FISH COMMUNITIES	DAVID G. ANGELER* AND WILLEM GOEDKOOP	2010
61	FIRST RECORD OF MIGRATING SILVER AMERICAN EELS (ANGUILLA ROSTRATA) IN THE ST. LAWRENCE ESTUARY ORIGINATING FROM A STOCKING PROGRAM	GUY VERREAULT , PIERRE DUMONT , JOHANNE DUSSUREAULT, RÉMI TARDIF	2010

62	Gravel Addition as a Habitat Restoration Technique for Tailwaters	RYAN A. MCMANAMAY AND D. J. ORTH, CHARLES A. DOLLOF, MARK A. CANTRELL	2010
63	Comparison of the British Columbia and Federal Environmental Assessments for the Prosperity Mine	M. Haddock	2011
64	Hydropower-related pulsed-flow impacts on stream fishes: a brief review, conceptual model, knowledge gaps, and research needs	Paciencia S. Young, Joseph J. Cech Jr., Lisa C. Thompson	2011
65	Survival of Migrating Atlantic Salmon Smolts through the Penobscot River, Maine: a Prerestoration Assessment	Christopher M. Holbrook , Michael T. Kinnison & Joseph Zydlewski	2011
66	Comparing the Fish and Benthic Macroinvertebrate Diversity of Restored Urban Streams to Reference Streams	Scott A. Stranko, Robert H. Hilderbrand, and Margaret A. Palmer	2011
67	Relationships between habitat characteristics and breeding population densities in sockeye salmon (Oncorhynchus nerka)	Douglas C. Braun and John D. Reynolds	2011
68	Preliminary Evaluation of a Large-Scale American Eel Conservation Stocking Experiment	Thomas C. Pratt & Ron W. Threader	2011
69	Environmental Effects of Hydrokinetic Turbines on Fish: Desktop and Laboratory Flume Studies	Paul T. Jacobson, Stephen V. Amaral, Theodore Castro-Santos, Dan Giza, Alexander J. Haro, George Hecker, Brian McMahon, Norman Perkins, Nick Pioppi	2012
70	Fish Habitat Banking in Canada: Opportunities and Challenges	Kyle Hunt, Paul Patrick, Michael Connell	2012
71	PASSAGE EFFICIENCY OF OFFSET AND STRAIGHT ORIFICES FOR UPSTREAM MOVEMENTS OF IBERIAN BARBEL IN A POOL-TYPE FISHWAY	A. T. SILVA, J. M. SANTOS, M. T. FERREIRA, A. N. PINHEIRO and C. KATOPODIS	2012
72	Development of a National Fish Passage Database for Canada (CanFishPass): Rationale, Approach, Utility, and Potential Applicability to Other Regions	Charles Hatry, Thomas R. Binder, Caleb T. Hasler, Keith D. Clarke, Christos Katopodis, Karen E. Smokorowski and Steven J. Cooke	2012

73	FISH STRANDING IN FRESHWATER SYSTEMS: SOURCES, CONSEQUENCES, AND MITIGATION	ALEXANDER NAGRODSKI*, GRAHAM D. RABY, CALEB T. HASLER, MARK K. TAYLOR, STEVEN J. COOKE	2012
74	REPRODUCTIVE SUCCESS OF CAPTIVELY BRED AND NATURALLY SPAWNED CHINOOK SALMON COLONIZING NEWLY ACCESSIBLE HABITAT	JOSEPH H. ANDERSON, PAUL L. FAULDS, WILLIAM I. ATLAS, AND THOMAS P. QUINN	2012
75	CHARACTERIZATION OF FISH PASSAGE CONDITIONS THROUGH THE FISH WEIR AND TURBINE UNIT 1 AT FOSTER DAM, OREGON, USING SENSOR FISH, 2012	J. DUNCAN	2013
76	KEEYASK GENERATION PROJECT - FISH HABITAT COMPENSATION PLAN	KEEYASK HYDROPOWER	2013
77	CYANIDE AND REMOVAL OPTIONS FROM EFFLUENTS IN GOLD MINING AND METALLURGICAL PROCESSES	NURAL KUYUCAK , ATA AKCIL	2013
78	DISTRIBUTION AND ABUNDANCE OF STREAM FISHES IN RELATION TO BARRIERS: IMPLICATIONS FOR MONITORING STREAM RECOVERY AFTER BARRIER REMOVAL	C. GARDNER, S. M. COGHLAN JR, J. ZYDLEWSKI, AND R. SAUNDERS	2013
79	PHOSPHORUS MITIGATION TO CONTROL RIVER EUTROPHICATION: MURKY WATERS, INCONVENIENT TRUTHS, AND "POSTNORMAL" SCIENCE	HELEN P. JARVIE,* ANDREW N. SHARPLEY, PAUL J. A. WITHERS, J. THAD SCOTT, BRIAN E. HAGGARD, AND COLIN NEAL	2013
80	DESIGN AND IMPLEMENTATION OF A NEW AUTONOMOUS SENSOR FISH TO SUPPORT ADVANCED HYDROPOWER DEVELOPMENT	Z. D. DENG, J. LU, M. J. MYJAK, J. J. MARTINEZ, C. TIAN, S. J. MORRIS, T. J. CARLSON, D. ZHOU, AND H. HOU	2014
81	UNDERSTANDING BAROTRAUMA IN FISH PASSING HYDRO STRUCTURES: A GLOBAL STRATEGY FOR SUSTAINABLE DEVELOPMENT OF WATER RESOURCES	RICHARD S. BROWN, ALISON H. COLOTELO, BRETT D. PFLUGRATH, CRAIG A. BOYS, LEE J. BAUMGARTNER, Z. DANIEL DENG, LUIZ G. M. SILVA, COLIN J. BRAUNER, MARTIN MALLEN-COOPER, OUDOM PHONEKHAMPENG, GARRY THORNCRAFT & DOUANGKHAM SINGHANOUVONG	2014
82	CONCEPTUAL FISH HABITAT OFFSETTING PLAN	KNIGHT PIÉSOLD LTD.	2015
83	THE ROLE OF OFFSETS IN COMPENSATING FOR DAMAGE IN THE COASTAL AND MARINE ENVIRONMENTS	SAMANTHA JANE PAREDES	2015

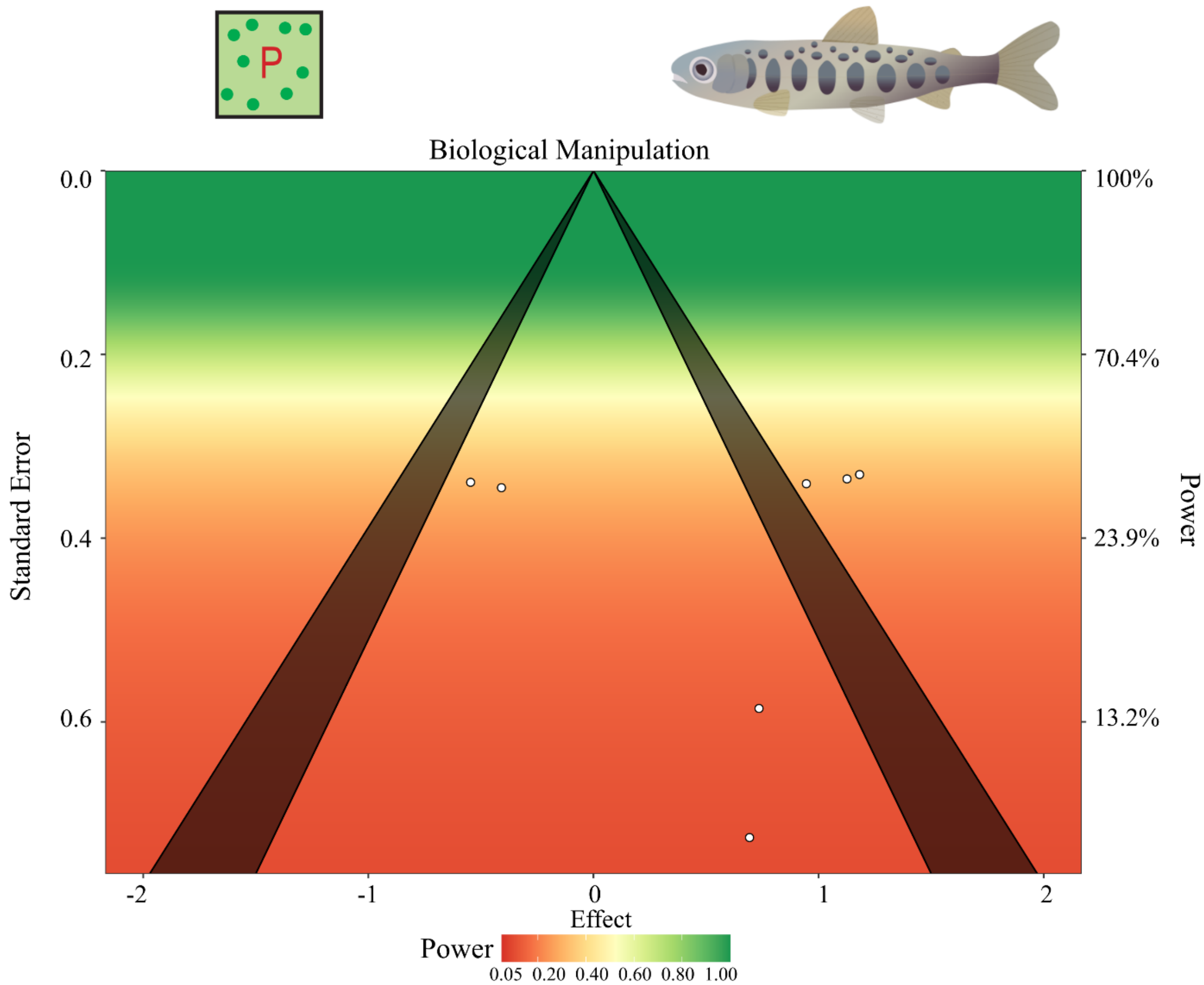
84	Evaluating Habitat Compensation in Insular Newfoundland Rivers: What have we Learned?	K. Clarke	2016
85	Managing flow ramping operations at run-of-river hydroelectric projects	Adam Lewis, Katie Healey	2016
86	Strategies for mitigating the impact of hydropower plants on the stocks of diadromous species in the Daugava River	Alona Bolonina, Claudio Comoglio, Olle Calles, Maris Kunickis	2016
87	Methods used in evaluating the effectiveness of creating/enhancing stream habitats in Newfoundland and Labrador	M.M. Roberge and T.N. Warren	2017
88	Monitoring fish habitat compensation in the Pacific region: lessons from the past 30 years	Michael J. Bradford, J. Stevenson Macdonald, and Colin D. Levings	2017
89	Monitoring to determine the efficacy of utilizing fishless lakes as fish habitat compensation in Labrador	M. M. Roberge and T. N. Warren	2017
90	Synthesis of Downstream Fish Passage Information at Projects Owned by the U.S. Army Corps of Engineers in the Willamette River Basin, Oregon	Amy C. Hansen, Tobias J. Kock, and Gabriel S. Hansen	2017
91	WESTRIDGE MARINE TERMINAL UPGRADE AND EXPANSION PROJECT APPLICATION TO VANCOUVER FRASER PORT AUTHORITY	Stantec	2017
92	Economic and policy considerations regarding hydropower and migratory fish	Emmi Nieminen, Kari Hyyti€ainen & Marko Lindroos	2017
93	REVIEW OF HISTORICAL HATCHERY RELEASES OF CHINOOK SALMON IN NEW ZEALAND	MARTIN UNWIN, RASMUS GABRIELSSON	2018
94	Fishfriendly Innovative Technologies for Hydropower	Christian Wolter, Richard A. Noble, Ruben van Treeck, Johannes Radinger	2019
95	Managing dams for energy and fish tradeoffs: What does a win-win solution take?	Cuihong Song a, Andrew Omalley b, Samuel G. Roy c, Betsy L. Barber b, Joseph Zydlewski d,b,Weiwei Mo a,*	2019
96	Quantifying Restoration Offsets at a Nuclear Power Plant in Canada	Lawrence W. Barnthouse, Cherie-Lee Fietsch, David Snider	2019
97	Ecological consequences of flow regulation by Run-of-River hydropower on salmonids	P. Gibeau	2020

98	REALIZING BENEFICIAL END USES FROM ABANDONED PIT LAKES	CHERIE D. MCCULLOUGH, MARTIN SCHULTZE AND JERRY VANDENBERG	2020



$\alpha = 0.05$; $\delta = 0.8$ | $med_{power} = 30\%$; $d_{33\%} = 0.84$; $d_{66\%} = 1.32$ | E = 3.09; O = 3; $p_{TES} = 0.952$; R-Index = 26.8%

Figure S2.2.1. *Funnel plot indicating publication and other bias for habitat creation studies related to residual or chronic harm. Centered around 0 of effect size (Hedge's g) and indicating power (right vertical axis) and standard error (SE; left vertical axis). Included information: alpha (0.05); true effect size (0.8); median power of all tests (30%); true effect size needed for reaching 33% and 66% of median power; test of significance (p = 0.952) for expected significant studies (E: 3.09 studies) and observed (O: 3); R-index for probability of replication (26.8%). Image attribution to: Joe Edgerton; Tracey Saxby; Integration and Application Network (ian.umces.edu/media-library).*



$\alpha = 0.05$; $\delta = 0.5$ | $med_{power} = 31.1\%$; $d_{33\%} = 0.52$; $d_{66\%} = 0.81$ | E = 1.81; O = 3; $p_{TES} = 0.307$; R-Index = 19.2%

Figure S2.2.2. *Funnel plot indicating publication and other bias for biochemical manipulation studies related to residual or chronic harm. Centered around 0 of effect size (Hedge's g) and indicating power (right vertical axis) and standard error (SE; left vertical axis). Included information: alpha (0.05); true effect size (0.5); median power of all tests (31.1%); true effect size needed for reaching 33% and 66% of median power; test of significance (p = 0.307) for expected significant studies (E: 1.81 studies) and observed (O: 3); R-index for probability of replication (19.2%). Image attribution to: Joe Edgerton; Tracey Saxby; Integration and Application Network (ian.umces.edu/media-library).*

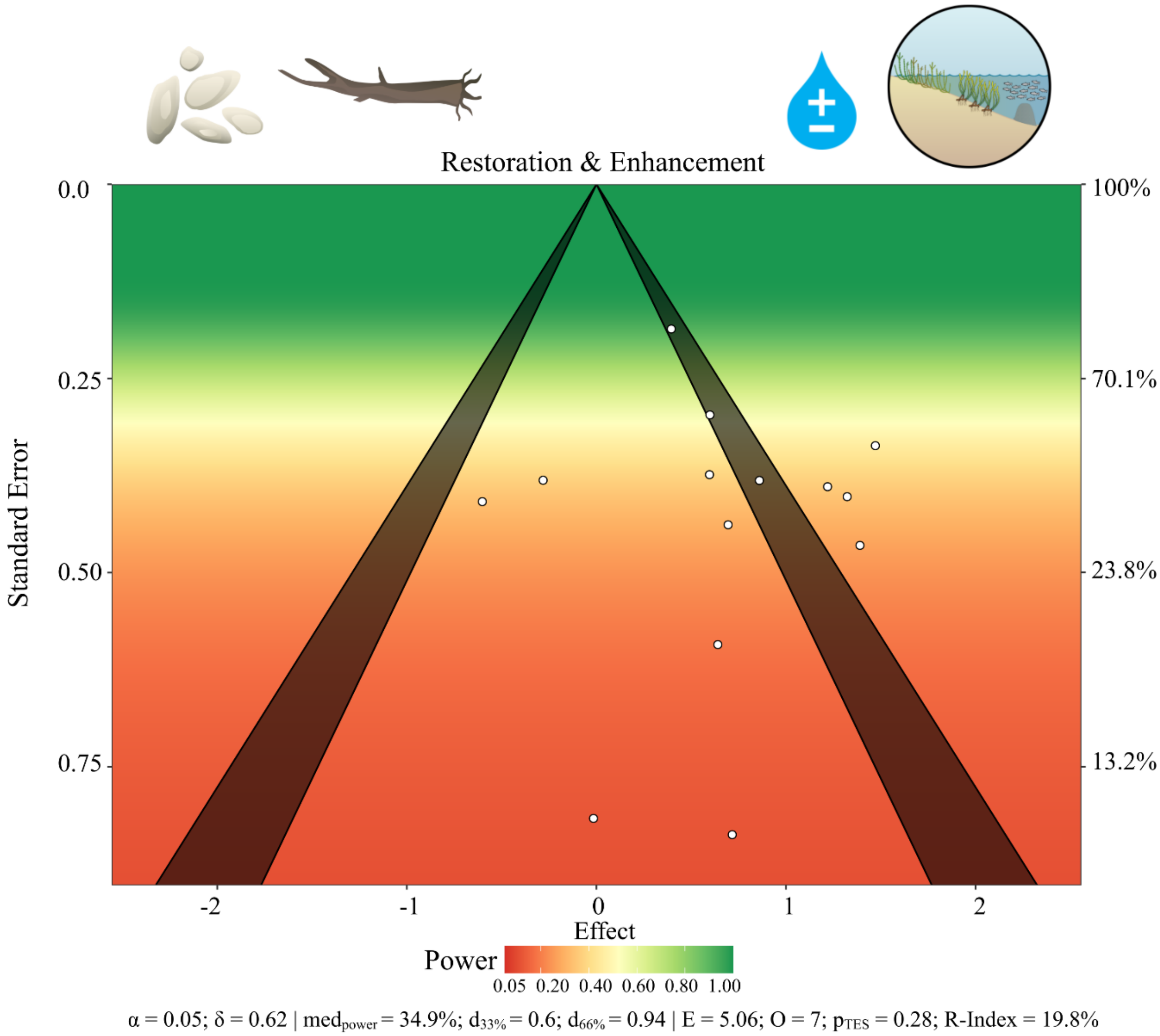


Figure S2.2.3. *Funnel plot indicating publication and other bias for habitat restoration studies related to residual or chronic harm. Centered around 0 of effect size (Hedge's g) and indicating power (right vertical axis) and standard error (SE; left vertical axis). Included information: alpha (0.05); true effect size (0.5); median power of all tests (34.9%); true effect size needed for reaching 33% and 66% of median power; test of significance (p = 0.28) for expected significant studies (E: 5.06 studies) and observed (O: 7); R-index for probability of replication (19.8%). Jane Hawkey; Joe Edgerton; Tracey Saxby; Integration and Application Network (ian.umces.edu/media-library).*

Chapter 3: Appendices and Supplements

Chapter 3.1 Appendices

Table A3.1.1. *Chi-squared test of independence for frequency table analyses. Categorical variables are comprised out of Bank-Type (Wetland; Stream; Species; Group; Multi-ecosystem; Multi-species) and Region (Northeast; Southeast; Midwest; West; Southwest). Positive residuals indicate a positive correlation between two categories and magnitude (e.g., 3.313 for Species Bank and West means that Species Banks are associated with the West). Post-hoc analysis for Chi-squared test based on standardized residuals with Bonferroni correction indicating significance of positive or negative correlations.*

Pearson's chi-squared test for Credit-Type

X-squared = 208.8; df = 24; p-value < 2.2e-16

Residuals	Wetland	Stream	Species	Group	Multi-Spe	Multi-Eco
Northeast	2.268	-0.815	-2.208	-1.732	-1.794	1.360
Southeast	-0.797	3.261	-2.208	-1.732	-1.237	1.360
Midwest	3.323	0.055	-2.555	-1.733	-1.739	-2.371
West	-2.173	-2.561	3.313	6.929	6.566	-1.929
Southwest	-1.623	0.059	3.658	-1.732	-1.794	1.580

p-values	Wetland	Stream	Species	Group	Multi-Spe	Multi-Eco
Northeast	0.005	1.000	0.408	1.000	1.000	1.000
Southeast	1.000	0.005	0.408	1.000	1.000	1.000
Midwest	<0.001	1.000	0.123	1.000	1.000	1.000
West	0.161	0.110	0.005	<0.001	<0.001	0.795
Southwest	1.000	1.000	0.001	1.000	1.000	1.000

Table A3.1.2. *Linear model and associated p-values for the six Mitigation-Methods and transactions per method (Establishment; Preservation; Rehabilitation; Re-establishment; Enhancement) over time in years from 1995 to 2020.*

Linear model, establishment transactions over time (R2: 0.37)	Estimate	Std. Error	t-value	Pr(>|t|)
(Intercept)	1154.178	290.673	3.971	<0.001
Year	-0.570	0.144	-3.940	<0.001

Linear model, preservation transactions over time (R2: 0.03)	Estimate	Std. Error	t-value	Pr(>|t|)
(Intercept)	-742.578	585.502	-1.268	0.217

Year	0.384	0.291	1.318	0.200

Linear model, rehabilitation transactions over time (R2: 0.74)	Estimate	Std. Error	t-value	Pr(>|t|)
(Intercept)	-1.222e+03	1.512e+02	-8.079	<0.001
Year	6.160e-01	7.534e-02	8.176	<0.001

Linear model, re-establishment transactions over time (R2: 0.25)	Estimate	Std. Error	t-value	Pr(>|t|)
(Intercept)	1123.837	353.778	3.177	<0.05
Year	-0.544	0.176	-3.089	<0.05

Linear model, enhancement transactions over time (R2: 0.01)	Estimate	Std. Error	t-value	Pr(>|t|)
(Intercept)	-213.475	274.658	-0.777	0.445
Year	0.114	0.136	0.836	0.411

***Table A3.2.3**. Chi-squared test of independence for frequency table analyses. Categorical variables are comprised out of Mitigation-Target (Wetland; Stream; Species; Group) and NNL category (Loss; Partial, NNL; Gain). Positive residuals indicate a positive correlation between two categories and magnitude (e.g., -4.417 for Wetland and partial NNL means that Wetland transactions are not associated with partial NNL). Post-hoc analysis for Chi-squared test based on standardized residuals with Bonferroni correction indicating significance of positive or negative correlations.*

Pearson's chi-squared test for Mitigation-Target

X-squared = 419.02; df = 9; p-value < 2.2e-16

Residuals	Wetland	Stream	Species	Group
Loss	-3.317	8.967	-1.334	-0.307
Partial	-4.417	12.533	-0.169	-2.817
NNL	0.966	-4.235	1.728	2.810
Gain	4.015	-9.701	-1.008	-0.428

p-values	Wetland	Stream	Species	Group
Loss	<0.001	0.001	1.000	1.000
Partial	<0.001	<0.001	1.000	0.02
NNL	0.05	<0.001	0.049	0.006
Gain	<0.001	<0.001	1.000	1.000

Table A3.2.4. *Chi-squared test of independence for frequency table analyses. Categorical variables are comprised out of Mitigation-Method (Establishment; Preservation; Rehabilitation; Re-establishment; Enhancement). Positive residuals indicate a positive correlation between two categories and magnitude (e.g., -15.310 for Preservation and partial NNL means that Preservation transactions are not associated with partial NNL). Post-hoc analysis for Chi-squared test based on standardized residuals with Bonferroni correction of positive or negative correlations.*

Pearson's chi-squared test for Mitigation-Method

X-squared = 5234.5; df = 12; p-value < 2.2e-16

Residuals	Preservation	Enhancement	Rehabilitation	Reestab	Establish
Loss	-1.145	-6.694	8.556	1.544	4.434
Partial	-23.975	11.694	16.032	6.083	2.398
NNL	-22.621	23.465	-3.821	5.247	-0.01
Gain	42.304	-29.910	-12.720	-10.790	-3.849

p-values	Preservation	Enhancement	Rehabilitation	Reestab	Establish
Loss	1.000	<0.001	<0.001	1.000	<0.001
Partial	<0.001	<0.001	<0.001	<0.001	0.111
NNL	<0.001	<0.001	<0.001	<0.001	1.000
Gain	<0.001	<0.001	<0.001	<0.001	<0.001

Table A3.2.5. *Chi-squared test of independence for frequency table analyses. Categorical variables are comprised out of Mitigation-Target (Wetland; Stream; Species; Group) and NNL category (Match (Impact type matches offset type); Miss (Impact type = Area and function loss through habitat loss; Offset type = Function gain through Preservation, Enhancement; Rehabilitation or Buffer but no new area gain); Overcompensate (Impact type = function loss through degradation; Offset type = function and area gain through establishment or re-establishment). Positive residuals indicate a positive correlation between two categories and magnitude (e.g., -2.668 for Wetland and Miss means that Wetland credits transactions are not associated with mismatching impact and offset type). Post-hoc analysis for Chi-squared test based on standardized residuals with Bonferroni correction of positive or negative correlations.*

Pearson's chi-squared test for Impact to offset type match or mismatch

X-squared = 98.005; df = 6; p-value < 2.2e-16

Residuals	Wetland	Stream	Species	Group
Miss	-1.120	3.816	0.093	-1.869
Match	1.214	-3.374	0.042	0.147
Overcompensate	-2.971	6.749	-0.387	3.359

p-values	Wetland	Stream	Species	Group
Miss	0.0138	<0.001	1.000	0.039

Match	<0.001	<0.001	1.000	1.000
Overcompensate	<0.001	<0.001	1.000	0.006

Chapter 3.2 Appendices

Table A3.2.1. Model selection for Wetland Banks.
Wetland banks – Unreleased Capacity (UCA)

Dickey-Fuller	Lag order	p-value
-2.0828	2	0.5408
Corrected (x2 diff)		
-3.7032	2	0.04263

p	D	q	AIC	AICc	BIC
1	2	1	525.98	527.18	529.51
Sigma^2	Log likelihood				
145757775	-259.99				

Best model: ARIMA (1,2,1)

Residuals	Ljung-Box test	
Q*	df	p-value
2.0829	3	0.5554

Wetland banks – Available Unwithdrawn Reserve (AUR)

Dickey-Fuller	Lag order	p-value
-1.8715	2	0.6213
Corrected (x2 diff)		
-7.283	2	<0.01

p	D	q	AIC	AICc	BIC
2	2	3	475.59	480.53	482.66
Sigma^2	Log likelihood				
15053991	-231.79				

Best model: ARIMA (2,2,3)

Residuals	Ljung-Box test	
Q*	df	p-value
1.9207	3	0.589

Wetland banks – Yearly added capacity (YAC)

Dickey-Fuller	Lag order	p-value
-2.2605	2	0.4731
Corrected (x1 diff)		
-3.9124	2	0.02768

p	D	q	AIC	AICc	BIC
1	1	1	552.65	553.79	556.31
Sigma^2	Log likelihood				
189939726	-273.33				

Best model: ARIMA (1,1,1)

Residuals	Ljung-Box test	
Q*	Df	p-value
0.32184	3	0.9559

Table A3.2.2. Model selection for Stream Banks.
Stream banks – Available Unwithdrawn Reserve (AUR)

Dickey-Fuller	Lag order	p-value
-1.9546	2	0.5897
Corrected (x2 diff)		
-7.4338	2	0.01

p	D	q	AIC	AICc	BIC
2	2	3	663.53	668.47	670.6
Sigma^2	Log likelihood				
4.53e+10	-328.32				

Best model: ARIMA (2,2,3)

Residuals	Ljung-Box test	
Q*	df	p-value
6.2568	3	0.09976

Stream banks – Yearly added Capacity (YAC)

Dickey-Fuller	Lag order	p-value
-1.4538	2	0.7804
Corrected (x1 diff)		
-4.1009	2	0.01973

p	D	q	AIC	AICc	BIC
0	1	1	767.29	767.83	769.73
Sigma^2	Log likelihood				
1.086e+12	-381.64				

Best model: ARIMA (0,1,1)

Residuals	Ljung-Box test	
Q*	df	p-value
6.664	4	0.1547

Stream banks – Unreleased Capacity (UCA)

Dickey-Fuller	Lag order	p-value
-1.8179	2	0.6418
Corrected (x2 diff)		
-4.5987	2	0.01

p	D	q	AIC	AICc	BIC
1	2	3	721.41	724.74	727.3

Sigma^2	Log likelihood
1491520	-196.69

Best model: ARIMA (1,2,3)

Residuals	Ljung-Box test	
Q*	Df	p-value
2.7668	3	0.429

Table 3.2.3. Model selection for Multi-Ecosystem Banks.

Multi-Ecosystem banks – Unreleased Capacity (UCA)

Dickey-Fuller	Lag order	p-value
-1.464	2	0.7766
Corrected (x2 diff)		
-4.9658	2	<0.01

p	D	q	AIC	AICc	BIC
0	2	1	442.49	443.06	444.84
Sigma^2	Log likelihood				
5108744	-219.24				

Best model: ARIMA (0,2,1)

Residuals	Ljung-Box test	
Q*	df	p-value
1.601	3	0.6592

Multi-Ecosystem banks – Available Unwithdrawn Reserve (AUR)

Dickey-Fuller	Lag order	p-value
0.56518	2	0.99
Corrected (x2 diff)		
-4.3775	2	0.01009

p	D	q	AIC	AICc	BIC
1	2	3	431.18	431.75	433.53
Sigma^2	Log likelihood				
3191277	-213.59				

Best model: ARIMA (1,2,3)

Residuals	Ljung-Box test	
Q*	df	p-value
5.6349	3	0.095419

Multi-Ecosystem banks – Yearly added capacity (YAC)

Dickey-Fuller	Lag order	p-value
-1.5815	2	0.7318
Corrected (x1 diff)		
-3.9949	2	0.02343

p	D	q	AIC	AICc	BIC
1	1	0	486.35	486.9	488.79
Sigma^2	Log likelihood				
14354171	-241.18				

Best model: ARIMA (1,1,0)

Residuals	Ljung-Box test	
Q*	df	p-value
2.7618	4	0.5984

Table 3.2.4. Model selection for Species Banks/ Conservation Banks.

Species banks – Available Unwithdrawn Reserve (AUR)

Dickey-Fuller	Lag order	p-value
-0.77888	2	0.9518
Corrected (x3 diff)		
-5.9106	2	<0.01

p	D	q	AIC	AICc	BIC
1	3	3	470.64	474.17	476.32
$Sigma^2$	Log likelihood				
23955185	-230.32				

Best model: ARIMA (1,3,3)

Residuals	Ljung-Box test	
Q*	df	p-value
4.5479	3	0.2081

Species banks – Yearly added Capacity (YAC)

Dickey-Fuller	Lag order	p-value
-2.8005	2	0.2674
Corrected (x1 diff)		
-3.945	2	0.02536

p	D	q	AIC	AICc	BIC
0	1	0	585.37	585.54	586.59
$Sigma^2$	Log likelihood				
797389698	-291.68				

Best model: ARIMA (0,1,0)

Residuals	Ljung-Box test	
Q*	df	p-value
4.2864	5	0.509

Species banks – Unreleased Capacity (UCA)

Dickey-Fuller	Lag order	p-value
-2.3168	2	0.4517
Corrected (x2 diff)		
-3.8863	2	0.02955

p	D	q	AIC	AICc	BIC
0	2	0	540.81	540.99	541.99
$Sigma^2$	Log likelihood				
329331206	-269.41				

Best model: ARIMA (0,2,0)

Residuals	Ljung-Box test	
Q*	Df	p-value

4.3514	5	0.5

Table A3.2.5. Linear model output over time for Wetland banks and associated YAC; UCA and AUR.

Wetl-YAC	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-775289.7	676767.5	-1.146	-0.263	
Year	394.2	337.1	1.169	0.254	0.015

Wetl-UCA	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-1.358e+07	9.892e+05	-13.73	7.34e-13	
Year	6.798e+03	4.928e+02	13.79	6.62e-13	0.883

Wetl-AUR	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-8928539.8	348249.4	-25.64	<2e-16	
Year	4479.6	173.5	25.82	<2e-16	0.964

Table A3.2.6. Linear model output over time for Stream banks and associated YAC; UCA and AUR.

Stream-YAC	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-161817957	51235416	-3.158	0.00425	
Year	81038	25522	3.175	0.00408	0.267

Stream-UCA	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-611187917	67120151	-9.106	2.96e-09	
Year	305488	33435	9.137	2.78e-09	0.767

Stream-AUR	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-135235216	12222257	-11.06	6.58e-11	
Year	67651	6088	11.11	6.04e-11	0.831

Table A3.2.7. Linear model output over time for Multi-Ecosystem banks and associated YAC; UCA and AUR.

Multi-YAC	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-548216.6	194927.4	-2.812	0.00965	
Year	275.6	97.1	2.838	0.00908	0.22

Multi-UCA	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-4063671	349334	-11.63	2.37e-11	
Year	2032	174	11.68	2.20e-11	0.844

Multi-AUR	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-1.596e+06	1.886e+05	-8.460	1.16e-08	
Year	8.002e+02	9.395e+01	8.517	1.02e-08	0.741

Table A3.2.8. Linear model output over time for Species banks and associated YAC; UCA and AUR

Species-YAC	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-1323611.7	1163508.8	-1.138	0.267	
Year	663.4	579.6	1.145	0.264	0.012

Species-UCA	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted

Intercept	-5262185.4	1026555.3	-5.126	3.02e-05	
Year	2631.7	511.4	5.146	2.87e-05	0.505

Species-AUR	Estimate	Std. Error	t-value	Pr(>|t|)	R2-adjusted
Intercept	-3605437.6	555221.6	-6.494	1.03e-06	
Year	1802.1	276.6	6.516	9.72e-07	0.624

Chapter 3.2 Supplements

***S3.2.1** – Model selection*

Model selection process Example based on Multi-Ecosystem banks and their Unreleased Capacity (UCA):

Main model components for univariate time series prediction, selecting candidate Auto Regressive Moving Average (ARMA)

p = is the number of autoregressive terms (AR)

d = is the number of non-seasonal differences needed for stationarity

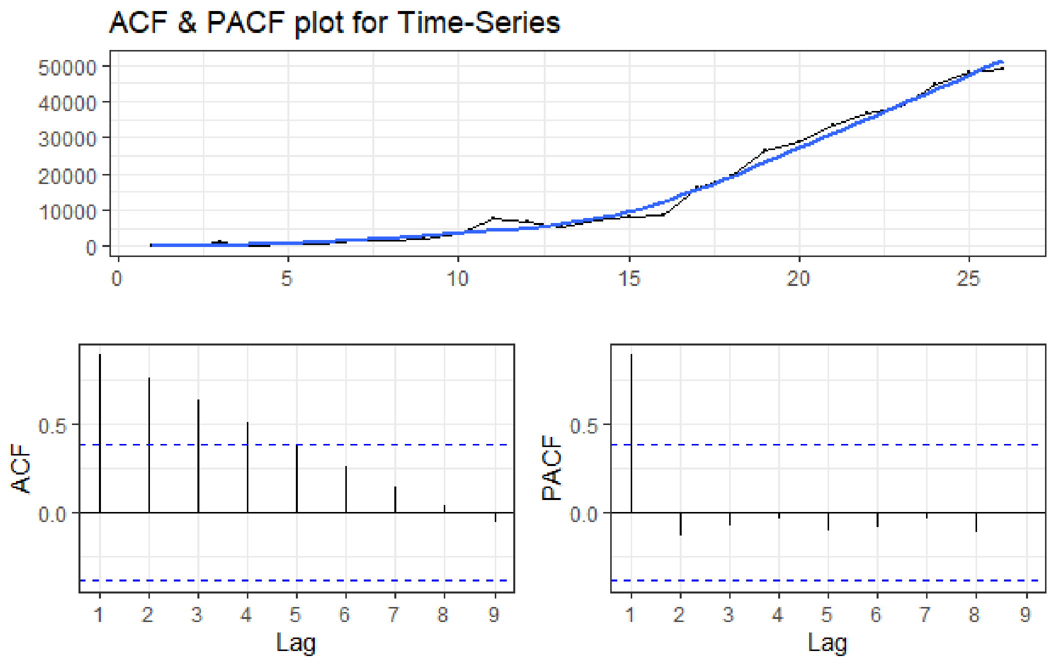
q = is the number of lagged forecast errors in the prediction equation (MA)

Show up in the **ARIMA model as: order = c(p, d, q)**

Base model order = c(p, d, q)

1. Load data; convert into time series and test for stationarity of time series (d)

Dickey-Fuller = -2.2605; Lag order = 2; p-value = 0.4731

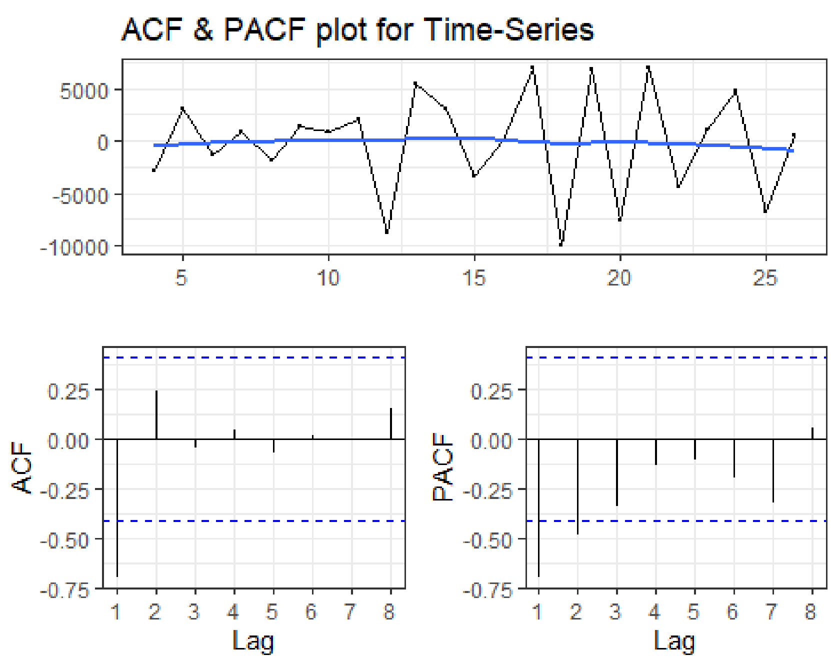


➔ ACF and PACF plots indicated p and q BUT cannot be investigated yet before establishing stationarity which is not given as indicated by the Dickey-Fuller test.

2.	Correct for non-stationarity

➔ Correction for non-stationarity; 3x diff; d = 3

Dickey-Fuller after correction= -4.9658; Lag order = 2; p-value <0.01



➔ Stationarity given

Base model order = c(p, 3, q)

3. ACF & PACF plots to determine p and q or alternatively use the auto.arima suggestions

Best suggestions for:

p = 1 (AR) – 1 significant ACF lag

q = 2 (MA) – 2 significant PACF lags

Base model order = c(1, 3, 2)

AIC (0,2,1): 442.49; AICc: 443.06; BIC: 444.84

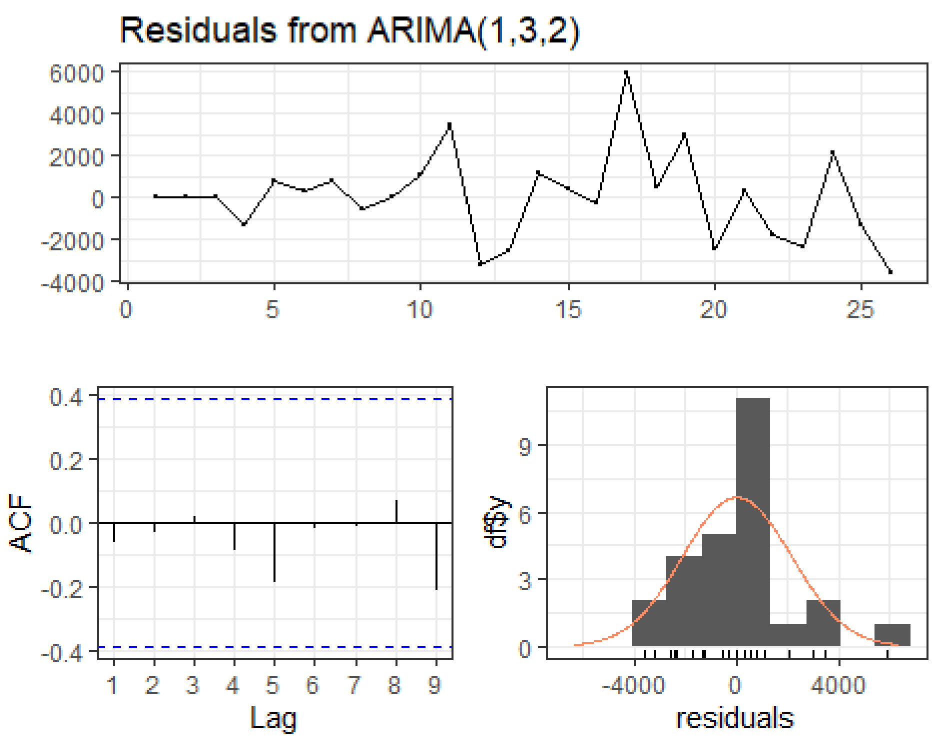
➔ Better fit than any other model option based on AIC

4. Check model residuals

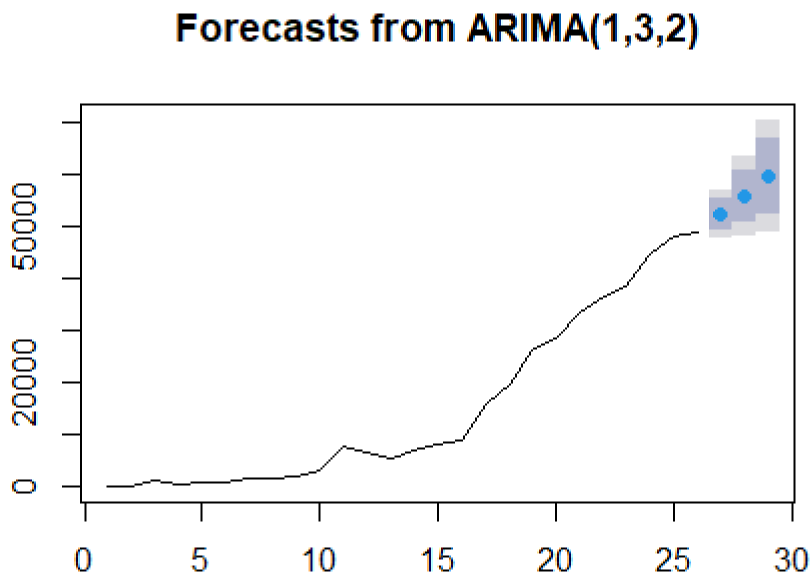
Q* = 1.601, df = 3, p-value = 0.6592 (Ljung-Box test)

Model df: 3. Total lags used: 6

➔ Non-significant p-value indicates that autocorrelations of the residuals is not given.



5. Forecast, example to predict the next points based on the selected model + 80 and 95% CI



Chapter 4: Appendices and Supplements

Chapter 4.1 Appendices

Table A4.1.1. *Structural assessment of the littoral zone of the study lake (Lake Steepbank, Lakeland County, Alberta). Coarse woody habitat (CWH) was counted in 100 m sections up to a depth of 2 m (n = 26). CWH were divided into small and large categories (small = diameter 10-15 cm and 1.5 – 3 m length, large = >15 cm diameter and >3 m). Overall structural complexity and CWH richness was presented through the number of CWH in both size classes per surveyed area in m$^{2.}$ Treatment area to be enhanced and was chosen from all surveyed areas (treatment T= experimental treatment area).*

ID	Section Length m	Section Width m	CWH Count	Section Area m^2	Structural complexity (CWH/100 m^2)	Small CWH	Large CWH	Treatment
1	99.8	3.2	7	319.36	2.191884	6	1	-
2	95.3	2.8	5	266.84	1.873782	5	0	-
3	94.2	2.7	8	254.34	3.145396	6	2	-
4	99	3	10	297	3.367003	9	1	-
5	100	2	5	200	2.5	4	1	-
6	98.4	2.5	8	246	3.252033	8	0	-

7	97.4	2.7	4	262.98	1.521028	4	0	-
8	93.4	4	6	373.6	1.605996	6	0	-
9	97	4.5	3	436.5	0.687285	2	1	-
10	91.7	5	2	458.5	0.436205	2	0	-
11	93.3	5.3	8	494.49	1.617828	6	2	-
12	94.6	6	3	567.6	0.528541	3	0	-
13	96.3	9	6	866.7	0.692281	5	1	-
14	89.3	9.4	8	839.42	0.953039	7	1	-
15	100	12	4	1200	0.333333	4	0	-
16	100	11.7	5	1170	0.42735	3	2	-
17	99.1	12.1	13	1199.11	1.084137	10	3	-
18	94.7	10.4	9	984.88	0.913817	8	1	-
19	95	8.9	12	845.5	1.419279	10	2	-
20	88.5	10.3	8	911.55	0.877626	7	1	-
21	100	13	4	1300	0.307692	4	0	T
22	99.5	14.4	6	1432.8	0.41876	6	0	T
23	98.7	13.8	2	1362.06	0.146836	2	0	T
24	96	14.7	3	1411.2	0.212585	3	1	T
25	93.9	15	2	1408.5	0.141995	2	0	T
26	94	14	3	1316	0.227964	3	0	T

Table A4.1.2. *Structural integrity (SI) assessment scores for coarse wood bundles and whole tree structures as part of the coarse woody habitat (CWH) study on Lake Steepbank, Lakeland County, Alberta. Percent (%) score gets subtracted from the original state of 100%, monitoring over the span of 2 years.*

Category	Characteristic	Structure Aspect	Structural Integrity %
Bundles			
1	Log detached (up to 60% for all 6)	Integrity	-10
2	Sandbag detached (up to 40% for all 4)	Anchor	-10
3	Structural move minor	Placement	-15
4	Structural move major	Placement	-30
5	Rope tear (up to 60% for 3 sides)	Integrity	-20
6	Sandbag leaking (up to 20% for all 4)	Anchor	-5
Trees			
7	Tree broken (up to 50% for both)	Integrity	-25
8	Tree shifted (up to 20% for both)	Placement	-10
9	Anchor line ripped (up to 40% for both)	Anchor	-20
10	Anchor point unearthed	Anchor	-50
11	Tree moved significantly (up 50% for both)	Placement	-25
12	Tree joint point undone	Integrity	-10

Table A4.1.3. *Moran's I based on Longitude and Latitude coordinates for CPUE and CPUA samples (Model <- Moran.I (dat$CPUE, CPUE.dists.inv) and inverse sample distance. Accounts for potential spatial autocorrelation within treatments (e.g. are CPUE near a bundle related to CPUE of a nearby Tree structure?) Significant p-values with an accepted Alpha < 0.05 indicate spatial autocorrelation. Calculations done through the "forecast" package.*

Treatment	observed	expected	sd	p-value
CPUE Cluster	0.0005946956	-0.02564103	0.02206682	0.2344706
CPUE Spaced	-0.01718295	-0.02564103	0.021506	0.694106
CPUA Cluster	-0.005146094	-0.02857143	0.02411764	0.3314016
CPUA Spaced	-0.02524583	-0.02857143	0.02276212	0.8838405

Table A4.1.4. *Levene and Shapiro Wilk test for Structural Integrity (SI). ANOVA for type III ("ez" package, factorial design and repeated measures over time) with treatment type and time as a factor (time as case identifier). Ls-means comparison to identify differences between structures within years ("lsmeans" package). ezANOVA(data=dat, between=.(Type),dv=SI, wid=Time, type=3, detailed=TRUE)*

Levene's Test	Df	F-value	Pr(>F)
group	11 108	8.5772	9.986e-11
Shapiro-Wilk	W	p-value	
	0.7657	1.474e-12	

Effect	DFn	DFd	SSn	SSd	F	p-value	ges (generalized Eta-squared)
Intercept	1	10	82419.187	4428.042	186.130	8.665955e-08	0.9490134
Type	1	10	1092.521	4428.042	2.467	1.473127e-01	0.1979003

Interaction (Tree – Bundle)	p-value for lsmeans
1 Day	1.0000000
1 Week	0.5785412
1 Month	0.0002338
3 Months	0.0000133
1 Year	0.0000000
2 Years	0.0000000

Table A4.1.5. *Bayesian model specifications (Model <- brm(CPUE or CPUA ~ Year * Treatment + (1 | Blocking factor) used to account for pseudo-replication within treatments, acknowledging that samples for each sampling time and treatment are derived from the same area. Probabilities estimated through the listed posterior calculations. Model built through R-packages "rstan" and "brms". Bulk_ESS and Tail_ESS minimum of 100 controlled for. Prior selection through get_prior function (class "b" – population level effect, normal prior 0, x).*

Iterations	Chain	Seed	Delta	Accepted rhat
20000	3	123	0.9999	<1.05**

Probability estimate*	mean (sample (predictions[, 2]) > predictions[, 1])

**Probability that sample 2 will be larger than sample 1 based on (in our case CPUE or CPUA from a sample from Treatment A compared to Treatment B)*
***Vehtari et al. 2019*

Table A4.1.6. *CPUE – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.15	1.11	-2.25	2.55	1.00	4492	6015
2018 Post	0.44	0.07	0.30	0.59	1.00	3169	7276
2019	1.00	0.07	0.85	1.15	1.00	3385	6900
2020	1.41	0.07	1.26	1.56	1.00	2748	6171
Control	0.02	1.46	-3.16	3.17	1.00	4116	6015
Spaced	-0.01	1.47	-3.26	3.14	1.00	4668	6535
2018Post Control	-0.40	0.11	-0.61	-0.19	1.00	3492	7814
2019 Control	-0.95	0.11	-1.16	-0.74	1.00	3760	7598
2020 Control	-1.37	0.11	-1.58	-1.16	1.00	3024	7012
2018Post Spaced	0.02	0.10	-0.19	0.22	1.00	3477	7827
2019 Spaced	-0.20	0.11	-0.41	0.01	1.00	3724	7625
2020 Spaced	-0.60	0.11	-0.81	-0.39	1.00	3192	7420

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.24	0.01	0.22	0.26	1.00	10533	14679

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.30	1.18	0.04	4.37	1.00	3921	5078

CPUE – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018-Pre	Spaced	0.17	0.24
2018-Pre	Cluster	0.15	0.24
2018-Pre	Control	0.15	0.24
2018-Post	Spaced	0.63	0.24
2018-Post	Cluster	0.60	0.24
2018-Post	Control	0.19	0.24
2019	Spaced	0.97	0.24
2019	Cluster	1.15	0.24
2019	Control	0.20	0.24
2020	Spaced	0.98	0.24
2020	Cluster	1.55	0.24
2020	Control	0.19	0.24

Table A4.1.7. *CPUA – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	10.07	4.91	-1.16	20.33	1.00	14652	14514
Control	-0.36	7.36	-16.30	16.08	1.00	15692	14709
Spaced	0.49	7.53	-15.63	16.94	1.00	15357	15580
2018 Post	4.09	0.95	2.24	5.96	1.00	13540	22092
2019	9.76	0.94	7.90	11.60	1.00	13549	22515
2020	18.79	0.94	16.93	20.63	1.00	13670	22604
Control 2018 Post	-6.15	1.34	-8.77	-3.53	1.00	15280	24113
Spaced 2018 Post	-2.45	1.34	-5.07	0.17	1.00	15885	24306
Control 2019	-11.92	1.33	-14.50	-9.28	1.00	15576	23777
Spaced 2019	-3.86	1.33	-6.49	-1.24	1.00	15620	23533
Control 2020	-19.94	1.33	-22.54	-17.35	1.00	15702	24697
Spaced 2020	-10.81	1.33	-13.41	-8.21	1.00	16289	23804

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	2.83	0.14	2.57	3.12	1.00	39322	29051

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	4.55	4.11	0.17	15.11	1.00	12568	17404

CPUA – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018-Pre	Cluster	10.42	2.89
2018 Pre	Control	9.99	2.90
2018 Pre	Spaced	10.91	2.90
2018 Post	Cluster	14.48	2.90
2018 Post	Control	7.96	2.92
2018 Post	Spaced	12.56	2.90
2019	Cluster	20.16	2.90
2019	Control	7.85	2.91
2019	Spaced	16.8	2.92
2020	Cluster	29.22	2.91
2020	Control	8.81	2.92
2020	Spaced	18.90	2.93

Table A4.1.8. *Relative Weight Spottail Shiner – Site + Blocking factor for Pseudoreplication; Prior (normal (0, 175), class "b"); Model: 15000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	142.44	19.27	102.01	185.05	1.00	9333	7326
Control 2020	4.64	28.01	-56.37	62.83	1.00	9782	7998
2018 Pre	1.72	27.84	-56.46	60.64	1.00	9990	8173
Spaced	15.10	27.48	-46.67	73.05	1.00	10038	7583

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	16.52	1.41	14.06	19.54	1.00	18132	13262

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	15.23	14.09	0.58	51.56	1.00	5374	8074

Relative weight – Predictions

Year*Site	Posterior Mean	Posterior SD
Clustered 2020	142.81	28.79
Control 2020	147.44	29.93
Spaced 2020	157.91	29.38
Pre 2018	144.48	30.16

Clustered > Spaced 0.45; Clustered > Control 0.32; Clustered > Pre 0.48; Control > Pre 0.53; Spaced > Pre 0.66; Spaced > Control 0.63

Table A4.1.9. *Relative Weight Northern Pike – Site + Blocking factor for Pseudoreplication; Prior (normal (0, 140), class "b"); Model: 15000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	128.36	15.42	95.83	159.31	1.00	8875	8284
Steepbank Mean	-10.18	23.07	-57.54	36.93	1.00	8355	8432
Spaced 2020	0.41	23.47	-46.95	48.29	1.00	8961	7682

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	16.20	2.42	12.29	21.78	1.00	15130	12447

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	12.05	11.38	0.42	39.88	1.00	5477	6904

Relative weight – Predictions		
Year*Site	Posterior Mean	Posterior SD
Steepbank Mean	118.73	24.81
Clustered 2020	128.85	24.74
Spaced 2020	129.40	25.03

Clustered > Spaced 0.49; Clustered > SB 0.63; Spaced > SB 0.55

Table A4.1.10. *Relative Weight White Sucker– Site + Blocking factor for Pseudoreplication; Prior (normal (0, 120), class "b"); Model: 15000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	100.32	4.68	90.36	109.49	1.00	8462	7723
Steepbank Mean	-0.44	7.15	-14.80	14.75	1.00	7354	6860

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	3.72	0.65	2.69	6.25	1.00	11930	10797

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	3.64	3.77	0.13	12.94	1.00	6427	6466

Relative weight – Predictions		
Year*Site	Posterior Mean	Posterior SD
Steepbank Mean	100.01	6.24
Enhanced 2020	100.42	6.39

SB > E 0.47

Table A4.1.11. *CPUE – Enhancement Type and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.65	0.72	-0.89	2.24	1.00	8907	7689
Tree	-0.04	0.10	-0.25	0.6	1.00	12635	16457
2019	0.37	0.11	0.15	0.58	1.00	13783	18537
2020	0.42	0.11	0.21	0.64	1.00	13191	17562
Tree 2019	0.08	0.15	-0.21	0.36	1.00	12942	17359
Tree 2020	0.43	0.15	0.14	0.72	1.00	12058	16129

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.33	0.02	0.29	0.38	1.00	19832	18259

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	0.83	1.03	0.04	3.69	1.00	5751	10440

CPUE – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Post	Tree	0.6063807	0.3480537
2018 Post	Bundle	0.6535156	0.347989
2019	Bundle	1.021598	0.3472977
2019	Tree	1.054458	0.3464193
2020	Tree	1.456325	0.3461763
2020	Bundle	1.076867	0.3496636

Table A4.1.12. *CPUA – Enhancement Type and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	11.98	2.99	7.61	16.00	1.00	8879	9293

Tree	2.87	1.26	0.41	5.35	1.00	11583	16073
2019	5.97	1.25	3.54	8.43	1.00	13101	16794
2020	7.43	1.26	4.92	9.87	1.00	13034	16458
Tree 2019	-2.05	1.77	-5.51	1.44	1.00	12005	17088
Tree 2020	6.16	1.78	2.67	9.66	1.00	11728	16012

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	3.77	0.27	3.29	4.34	1.00	19864	18722

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	2.13	2.59	0.05	9.18	1.00	6774	12015

CPUE – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Post	Bundle	12.10171	3.901354
2018 Post	Tree	14.97483	3.906085
2019	Bundle	18.07398	3.9113
2019	Tree	18.92341	3.897791
2020	Bundle	19.52395	3.886572
2020	Tree	28.56325	3.892142

Chapter 4.1 Supplements

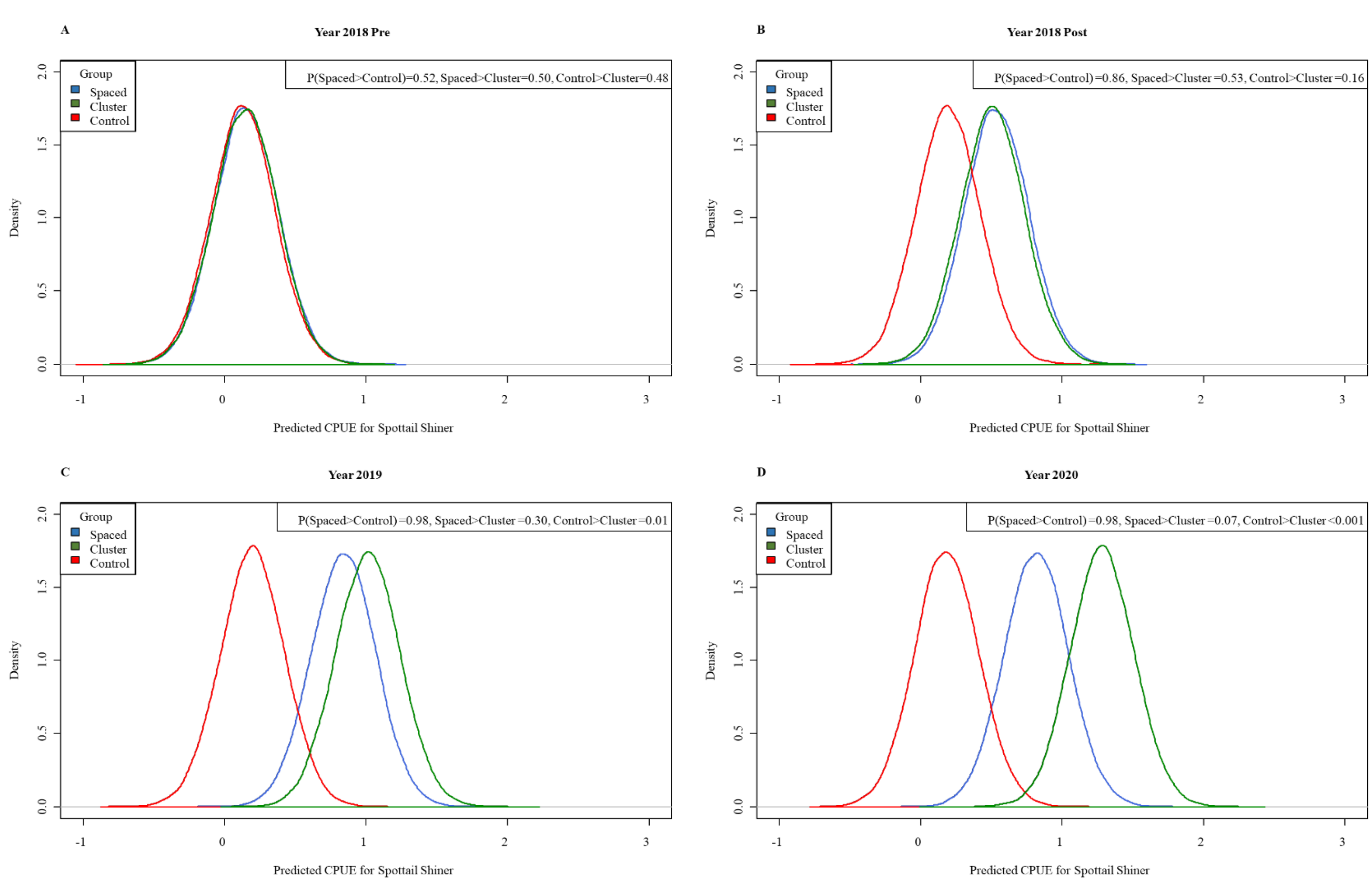


Figure S4.1.1. *Catch per unit effort (CPUE) per 12-hour minnow traps for spottail shiner density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

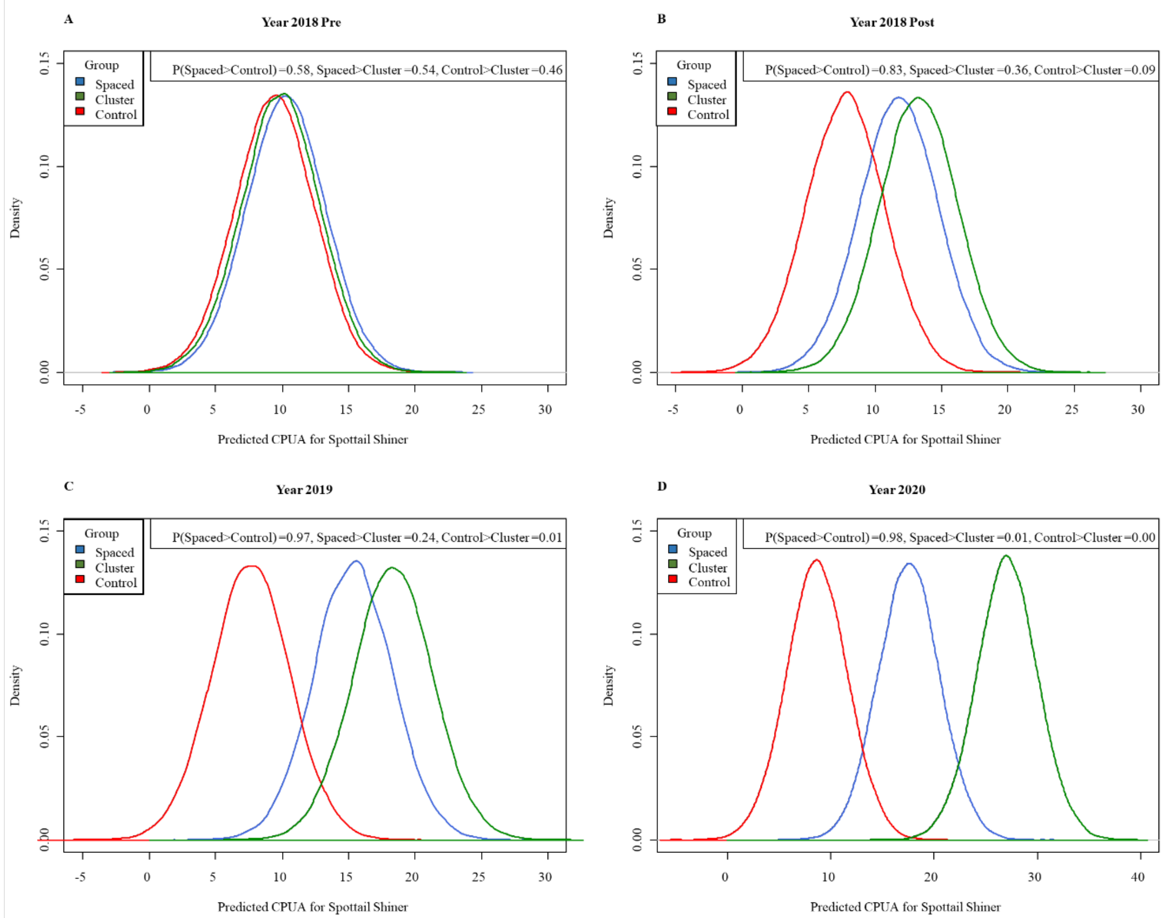


Figure S4.1.2. *Catch per unit area for 50 m seine hauls (100 m2 standardized) for spottail shiner density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

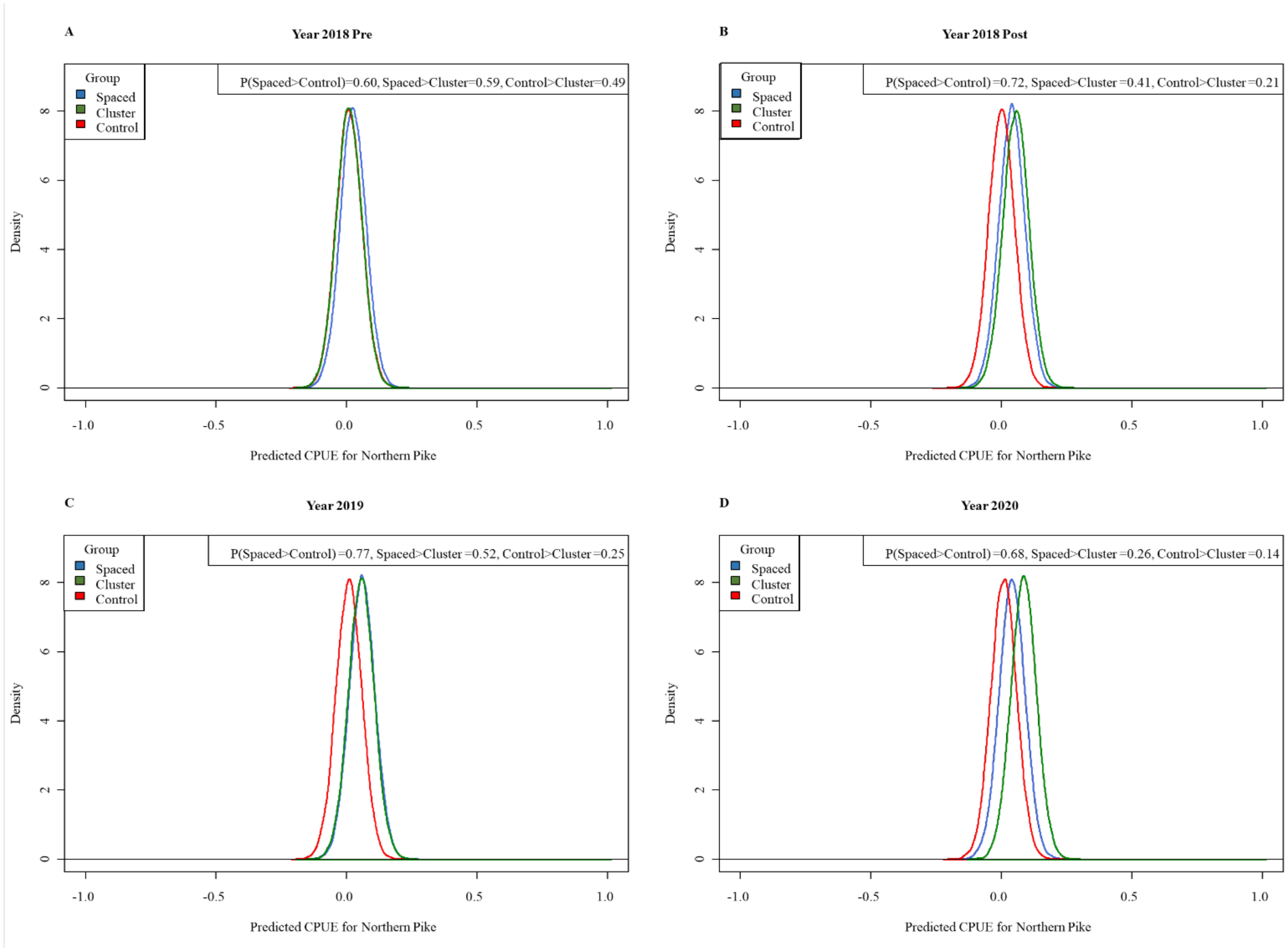


Figure S4.1.3. *Catch per unit effort (CPUE) per 12-hour minnow traps for northern pike density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

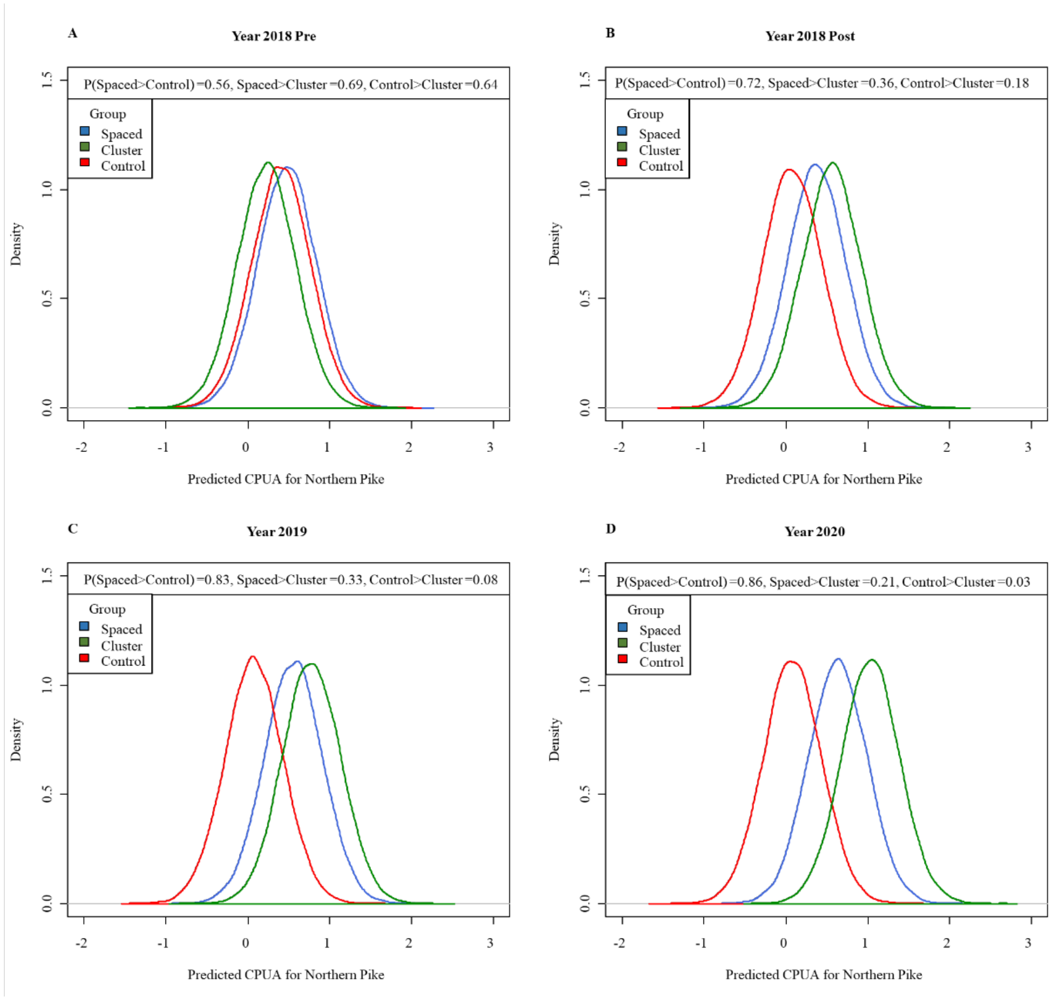


Figure S4.1.4. *Catch per unit area for 50 m seine hauls (100 m2 standardized) for northern pike density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

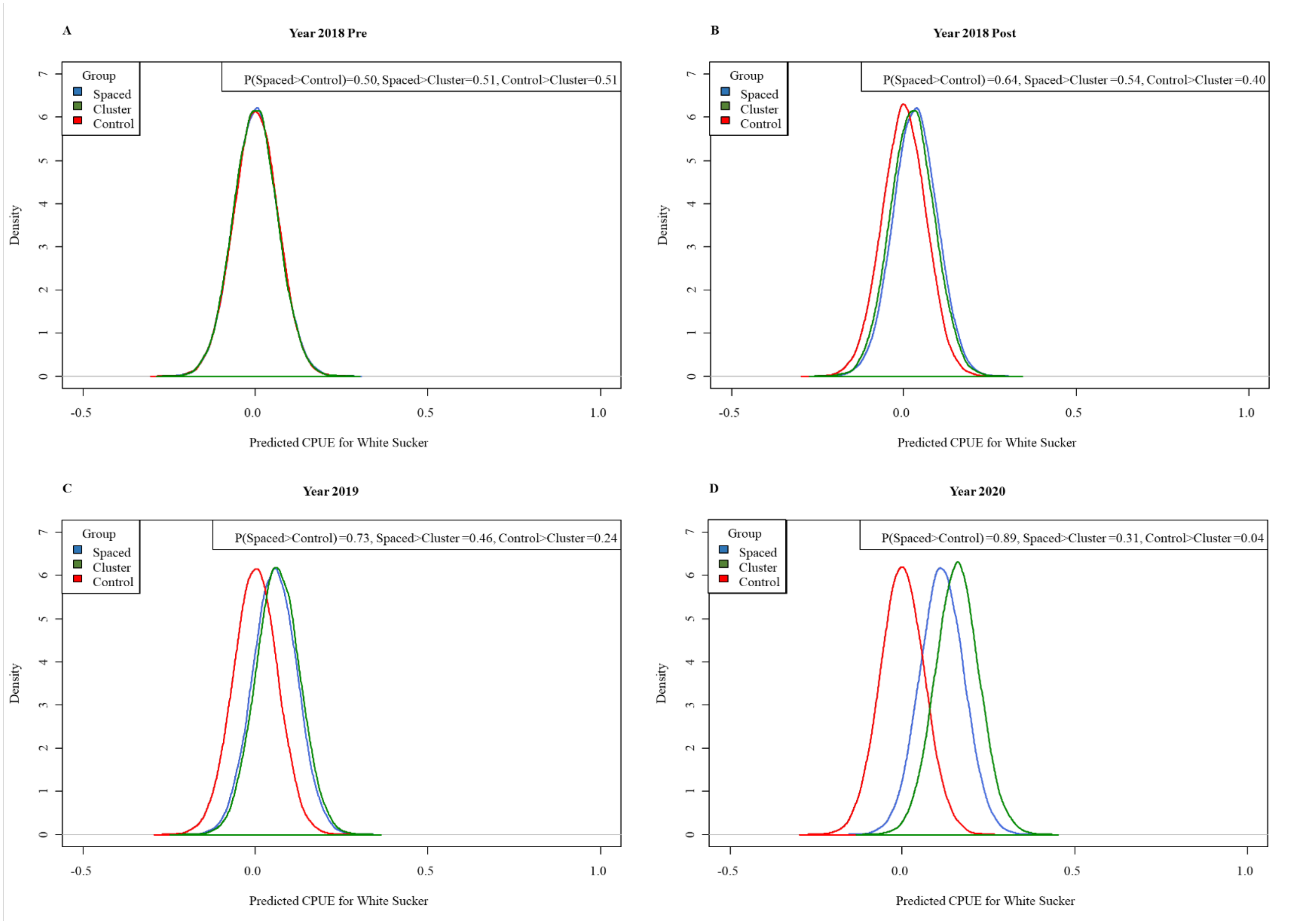


Figure S4.1.5. *Catch per unit effort (CPUE) per 12-hour minnow traps for white sucker density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

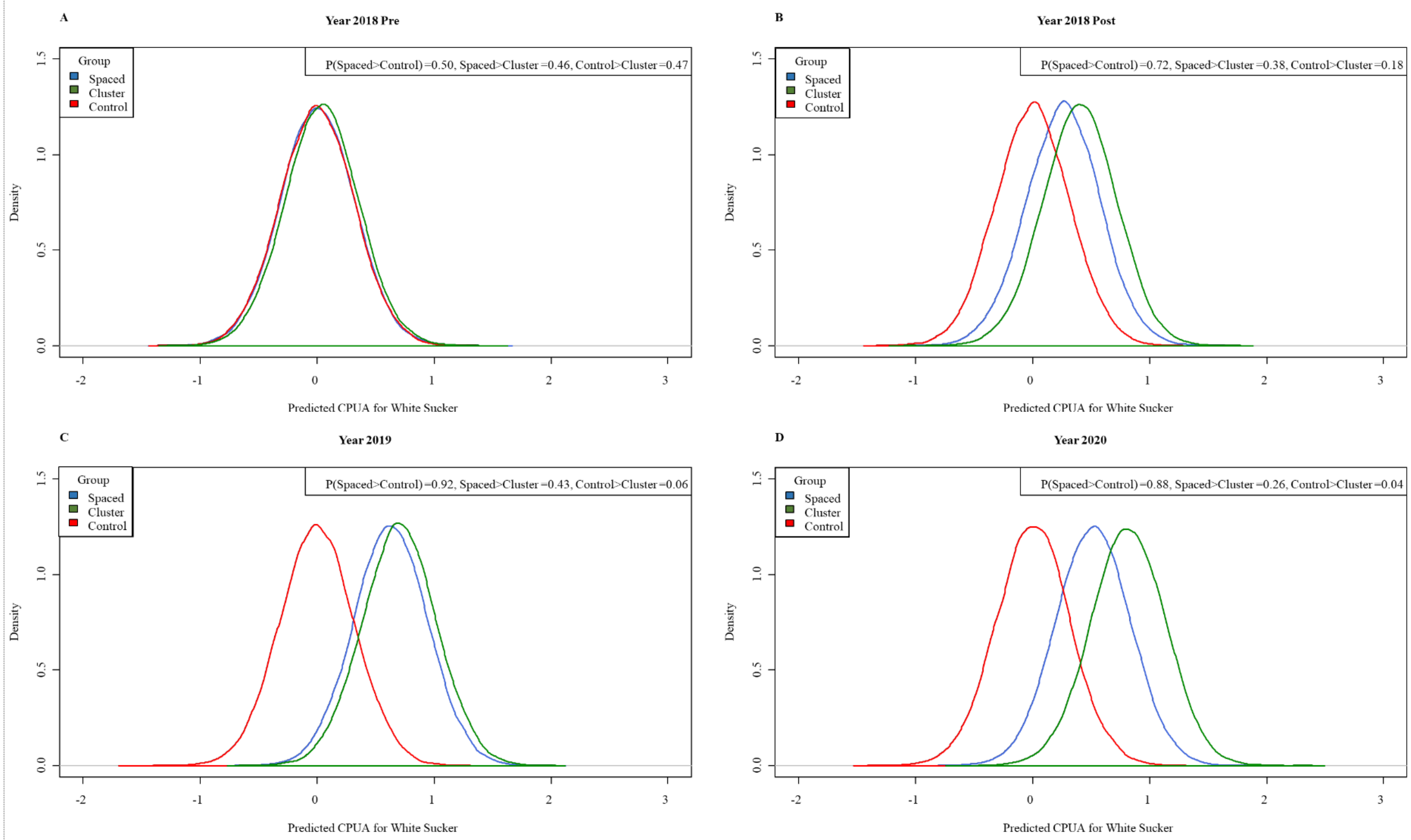


Figure S4.1.6. *Catch per unit area for 50 m seine hauls (100 m2 standardized) for white sucker density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

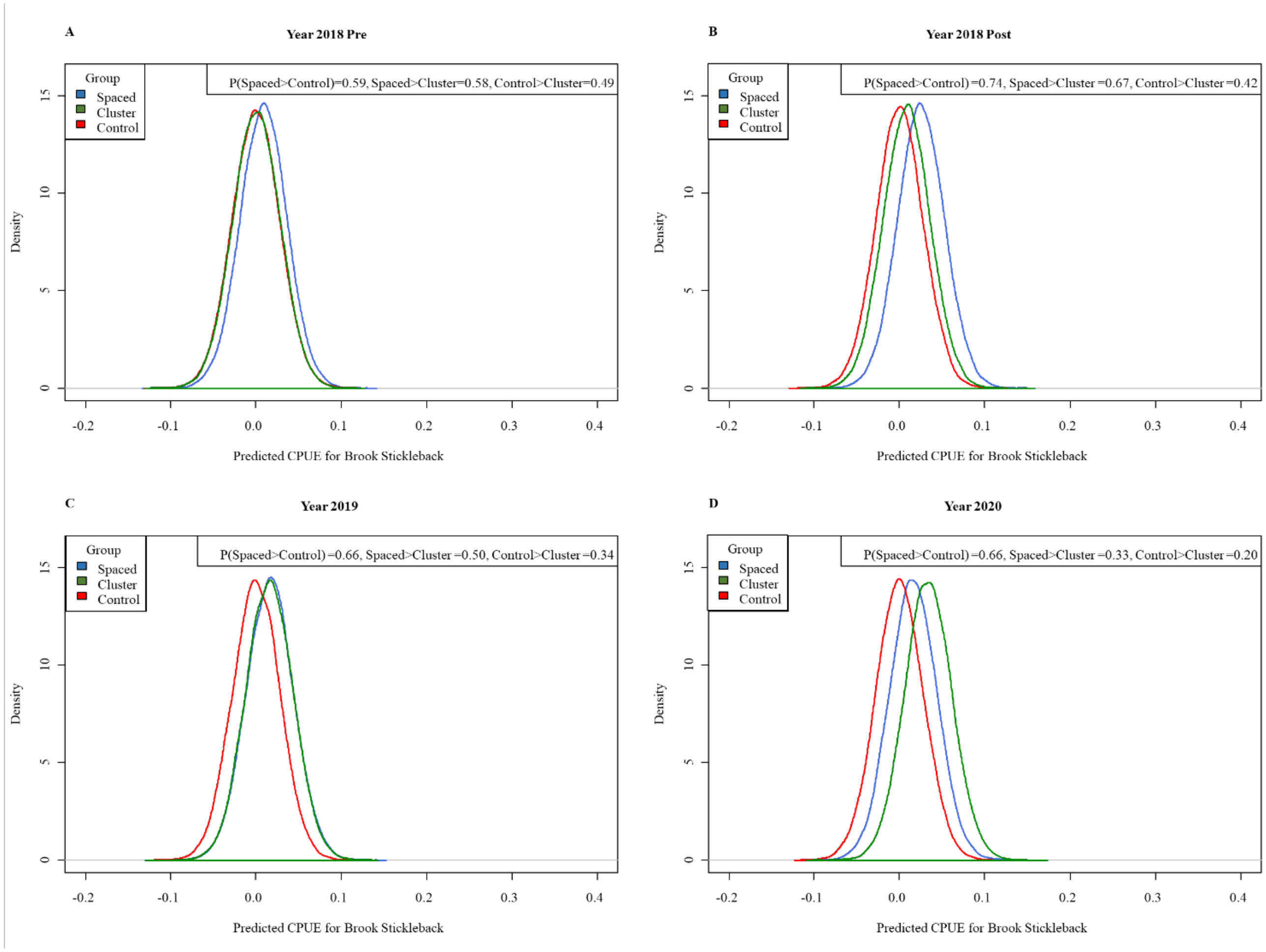


Figure S4.1.7. *Catch per unit effort (CPUE) per 12-hour minnow traps for brook stickleback density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

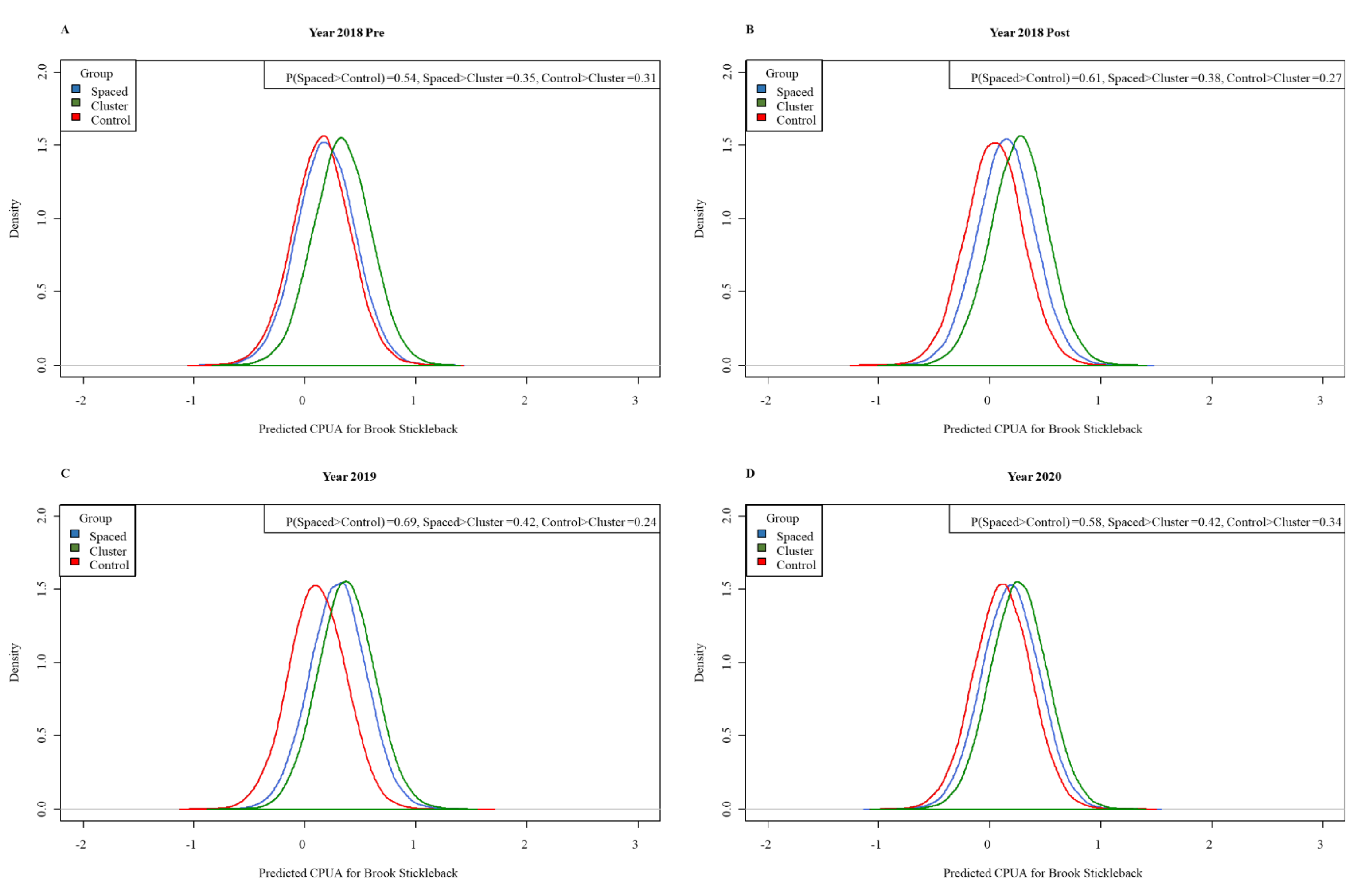


Figure S4.1.8*. Catch per unit area for 50 m seine hauls (100 m2 standardized) for brook stickleback density estimates derived from the posterior (predicted mean) of a Bayesian model with hierarchical structure to account for pseudoreplication as part of the Coarse Woody Habitat (CWH) study in Lakeland County, northern Alberta across years (2018 Pre-Enhancements A; 2018 Post-Enhancements B; 2019 C; 2020 D) and treatments (Clustered; 15 m between enhancements, Control; no enhancements, Spaced; 30 m between enhancements). Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

Table S4.1.1. *CPUE-Spottail Shiner – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 2), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.13	0.92	-1.83	2.12	1.00	18187	17150
Control	-0.01	1.10	-2.39	2.38	1.00	21951	21114
Spaced	0.00	1.09	-2.34	2.35	1.00	23231	22136
2018 Post	0.36	0.10	0.17	0.55	1.00	17184	23495
2019	0.87	0.10	0.68	1.06	1.00	17381	23685
2020	1.14	0.10	0.95	1.33	1.00	17587	23802
Control 2018 Post	-0.30	0.14	-0.57	-0.03	1.00	19448	24796
Spaced 2018 Post	0.03	0.14	-0.24	0.30	1.00	19274	26341
Control 2019	-0.80	0.14	-1.07	-0.53	1.00	19464	25913
Spaced 2019	-0.17	0.14	-0.44	0.09	1.00	19851	26412
Control 2020	-1.09	0.14	-1.35	-0.82	1.00	20121	26306
Spaced 2020	-0.47	0.14	-0.74	-0.21	1.00	20081	26659

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.22	0.01	0.19	0.25	1.00	38628	27202

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.08	1.03	0.04	3.74	1.00	12874	17608

CPUE-Spottail Shiner – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	0.15	0.23
2018 Pre	Cluster	0.15	0.23
2018 Pre	Control	0.13	0.23
2018 Post	Spaced	0.54	0.23
2018 Post	Cluster	0.51	0.23
2018 Post	Control	0.19	0.23
2019	Spaced	0.84	0.23
2019	Cluster	1.01	0.23
2019	Control	0.19	0.23

2020	Spaced	0.81	0.23
2020	Cluster	1.28	0.23
2020	Control	0.18	0.23

Table S4.1.2. *CPUA-Spottail Shiner – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	9.56	4.52	-0.75	18.88	1.00	17545	15519
Control	-0.41	6.94	-15.27	14.99	1.00	18358	16652
Spaced	0.32	6.90	-14.97	15.22	1.00	17824	16126
2018 Post	3.44	1.32	0.84	6.03	1.00	18311	25182
2019	8.47	1.32	5.88	11.08	1.00	18848	23980
2020	17.23	1.31	14.64	19.81	1.00	18569	23905
Control 2018 Post	-5.03	1.86	-8.72	-1.35	1.00	20701	26927
Spaced 2018 Post	-1.91	1.87	-5.57	1.79	1.00	20328	25913
Control 2019	-10.25	1.86	-13.91	-6.59	1.00	21171	25428
Spaced 2019	-3.39	1.85	-7.03	0.25	1.00	20704	26547
Control 2020	-17.99	1.85	-21.60	-14.34	1.00	21003	27159
Spaced 2020	-9.91	1.86	-13.56	-6.28	1.00	20245	26528

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	2.80	0.21	2.44	3.24	1.00	36928	26933

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	4.13	3.74	0.15	13.68	1.00	12870	16756

CPUA-Spottail Shiner – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	10.24	2.97
2018 Pre	Control	9.42	2.96
2018 Pre	Cluster	9.85	2.95
2018 Post	Spaced	11.78	2.97
2018 Post	Control	7.78	2.97
2018 Post	Cluster	13.28	2.96

2019	Spaced	15.32	2.96
2019	Control	7.63	2.96
2019	Cluster	18.32	3.00
2020	Spaced	17.56	2.96
2020	Control	8.65	2.96
2020	Cluster	27.08	2.94

Table S4.1.3. *CPUE-Northern Pike – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 2), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.03	0.92	-1.92	2.00	1.00	519	723
Control	-0.03	1.14	-2.56	2.31	1.01	386	415
Spaced	-0.02	1.17	-2.71	2.48	1.02	466	413
2018 Post	0.05	0.02	0.01	0.09	1.00	294	1012
2019	0.05	0.02	0.00	0.09	1.01	393	1183
2020	0.08	0.02	0.03	0.12	1.00	319	953
Control 2018 Post	-0.05	0.03	-0.11	0.00	1.00	329	1137
Spaced 2018 Post	-0.03	0.03	-0.09	0.03	1.00	433	899
Control 2019	-0.05	0.03	-0.10	0.01	1.00	463	1376
Spaced 2019	-0.01	0.03	-0.07	0.04	1.01	505	1346
Control 2020	-0.08	0.03	-0.13	-0.02	1.01	375	1099
Spaced 2020	-0.06	0.03	-0.12	0.00	1.00	472	1454

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.05	0.00	0.04	0.05	1.01	842	1054

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.13	1.03	0.05	3.81	1.01	767	1985

CPUE-Northern Pike – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	0.02	0.05
2018 Pre	Cluster	0.01	0.05
2018 Pre	Control	0.01	0.05
2018 Post	Spaced	0.04	0.05

2018 Post	Cluster	0.06	0.05
2018 Post	Control	<0.01	0.05
2019	Spaced	0.06	0.05
2019	Cluster	0.06	0.05
2019	Control	0.01	0.05
2020	Spaced	0.04	0.05
2020	Cluster	0.08	0.05
2020	Control	0.01	0.05

Table S4.1.4. *CPUA-Northern Pike – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.26	1.11	-2.15	2.71	1.00	17920	17943
Control	0.12	1.46	-3.15	3.20	1.00	20880	20396
Spaced	0.19	1.46	-3.00	3.33	1.00	20850	20438
2018 Post	0.32	0.16	0.01	0.63	1.00	18118	26441
2019	0.54	0.16	0.23	0.85	1.00	18460	25374
2020	0.80	0.16	0.49	1.12	1.00	19198	24436
Control 2018 Post	-0.65	0.23	-1.09	-0.21	1.00	20331	26686
Spaced 2018 Post	-0.43	0.22	-0.87	0.01	1.00	20912	26459
Control 2019	-0.87	0.22	-1.31	-0.43	1.00	19755	26427
Spaced 2019	-0.47	0.22	-0.91	-0.02	1.00	20949	26697
Control 2020	-1.13	0.23	-1.57	-0.69	1.00	21209	26845
Spaced 2020	-0.65	0.22	-1.09	-0.21	1.00	21840	26892

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.34	0.02	0.29	0.39	1.00	35140	27171

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.27	1.15	0.04	4.20	1.00	13496	17824

CPUA-Northern Pike – Predictions

Year	Treatment	Pred. Posterior Mean	Pred. Posterior SD
2018 Pre	Spaced	0.47	0.36
2018 Pre	Control	0.40	0.36

2018 Pre	Cluster	0.23	0.36
2018 Post	Spaced	0.37	0.36
2018 Post	Control	0.08	0.36
2018 Post	Cluster	0.56	0.36
2019	Spaced	0.56	0.36
2019	Control	0.08	0.36
2019	Cluster	0.78	0.36
2020	Spaced	0.63	0.36
2020	Control	0.08	0.36
2020	Cluster	1.03	0.36

Table S4.1.5. *CPUE-White Sucker – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 2), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.03	0.86	-1.89	2.07	1.01	682	793
Control	0.00	1.12	-2.49	2.35	1.01	736	1000
Spaced	0.00	1.07	-2.41	2.29	1.00	641	849
2018 Post	0.02	0.03	-0.03	0.08	1.00	774	1659
2019	0.06	0.03	0.01	0.12	1.00	689	1708
2020	0.16	0.03	0.10	0.21	1.00	707	1257
Control 2018 Post	-0.02	0.04	-0.10	0.05	1.00	725	1680
Spaced 2018 Post	0.01	0.04	-0.07	0.08	1.01	833	2026
Control 2019	-0.06	0.04	-0.14	0.01	1.00	729	1809
Spaced 2019	-0.01	0.04	-0.08	0.07	1.00	826	1865
Control 2020	-0.16	0.04	-0.24	-0.08	1.00	715	1719
Spaced 2020	-0.04	0.04	-0.12	0.03	1.01	816	1723

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.06	0.00	0.05	0.07	1.00	1575	3751

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.05	0.98	0.05	3.63	1.00	1146	2608

CPUE- White Sucker – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	<0.001	0.06
2018 Pre	Cluster	<0.001	0.06
2018 Pre	Control	<0.001	0.06
2018 Post	Spaced	0.03	0.06
2018 Post	Cluster	0.02	0.06
2018 Post	Control	<0.001	0.06
2019	Spaced	0.06	0.06
2019	Cluster	0.06	0.06
2019	Control	<0.001	0.06
2020	Spaced	0.11	0.06
2020	Cluster	0.16	0.06
2020	Control	<0.001	0.06

Table S4.1.6. *CPUA- White Sucker – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 4; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	-0.02	1.16	-2.61	2.37	1.00	7431	7862
Control	-0.02	1.48	-3.27	3.18	1.00	8669	12157
Spaced	-0.04	1.47	-3.22	3.18	1.00	10039	12140
2018 Post	0.37	0.14	0.09	0.64	1.00	7583	13785
2019	0.66	0.14	0.38	0.94	1.00	7023	14034
2020	0.77	0.14	0.50	1.05	1.00	6848	13054
Control 2018 Post	-0.37	0.20	-0.76	0.03	1.00	8528	15022
Spaced 2018 Post	-0.11	0.20	-0.50	0.29	1.00	8469	15562
Control 2019	-0.66	0.20	-1.05	-0.27	1.00	8224	14230
Spaced 2019	-0.03	0.20	-0.43	0.37	1.00	8437	14140
Control 2020	-0.77	0.20	-1.16	-0.38	1.00	8062	15869
Spaced 2020	-0.25	0.20	-0.64	0.14	1.00	8206	15161

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.30	0.02	0.26	0.35	1.00	20413	25263

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.30	1.18	0.05	4.34	1.00	8222	12311

CPUA- White Sucker – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	<0.01	0.32
2018 Pre	Control	<0.01	0.32
2018 Pre	Cluster	0.04	0.32
2018 Post	Spaced	0.26	0.32
2018 Post	Control	<0.01	0.32
2018 Post	Cluster	0.40	0.32
2019	Spaced	0.63	0.32
2019	Control	<0.01	0.32
2019	Cluster	0.70	0.32
2020	Spaced	0.52	0.32
2020	Control	<0.01	0.32
2020	Cluster	0.82	0.32

Table S4.1.7. *CPUE-Brook Stickleback – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 3), class "b"); Model: 20000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	-0.01	0.79	-1.78	1.65	1.02	208	341
Control	-0.10	1.00	-2.43	1.94	1.01	241	309
Spaced	0.05	1.11	-2.40	2.52	1.01	245	223
2018 Post	0.01	0.01	-0.01	0.03	1.04	175	469
2019	0.02	0.01	-0.01	0.04	1.033	195	594
2020	0.03	0.01	0.01	0.06	1.04	139	488
Control 2018 Post	-0.01	0.02	-0.04	0.03	1.04	146	363
Spaced 2018 Post	0.01	0.02	-0.02	0.04	1.04	171	312
Control 2019	-0.02	0.02	-0.05	0.02	1.02	248	587
Spaced 2019	-0.01	0.02	-0.04	0.03	1.03	188	535
Control 2020	-0.03	0.02	-0.07	-0.00	1.01	384	544
Spaced 2020	-0.03	0.02	-0.06	0.01	1.01	331	278

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.03	0.00	0.02	0.03	1.01	457	1108

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS

Sd(Intercept)	1.04	1.01	0.03	3.65	1.03	220	238

CPUE- Brook Stickleback – Predictions			
Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	<0.001	0.03
2018 Pre	Cluster	<0.001	0.03
2018 Pre	Control	<0.001	0.03
2018 Post	Spaced	0.03	0.03
2018 Post	Cluster	0.01	0.03
2018 Post	Control	<0.001	0.03
2019	Spaced	0.02	0.03
2019	Cluster	0.02	0.03
2019	Control	<0.001	0.03
2020	Spaced	0.02	0.03
2020	Cluster	0.03	0.03
2020	Control	<0.001	0.03

Table S4.1.8. *CPUA- Brook Stickleback – Treatment and Year + Blocking factor for Pseudoreplication; Prior (normal (0, 30), class "b"); Model: 20000 iterations; Chains: 3; Delta = 0.9999.*

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Intercept	0.28	1.14	-2.23	2.64	1.00	7883	9980
Control	-0.13	1.46	-3.29	3.06	1.00	8965	12351
Spaced	-0.09	1.48	-3.28	3.17	1.00	8766	11452
2018 Post	-0.07	0.12	-0.30	0.15	1.00	6658	14259
2019	0.04	0.12	-0.19	0.27	1.00	6744	14063
2020	-0.07	0.12	-0.30	0.16	1.00	6647	12820
Control 2018 Post	-0.04	0.16	-0.36	0.29	1.00	7860	16027
Spaced 2018 Post	0.04	0.16	-0.29	0.36	1.00	7725	15065
Control 2019	-0.08	0.16	-0.40	0.25	1.00	7698	15969
Spaced 2019	0.07	0.16	-0.26	0.39	1.00	7620	15777
Control 2020	0.04	0.16	-0.29	0.36	1.00	7622	14451
Spaced 2020	0.07	0.16	-0.25	0.39	1.00	7377	13904

Draws were sampled using sampling (NUTS). For each parameter, Bulk_ESS and Tail_ESS are effective sample size measures, and Rhat is the potential scale reduction factor on split chains (at convergence, Rhat = 1).

Family specific Parameters:

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sigma	0.25	0.02	0.21	0.28	1.00	18355	21864

Group-level Effects (Blocking factor for Pseudoreplication):

	Estimate	Est. Error	l-95% CI	u-95% CI	Rhat	Bulk_ESS	Tail_ESS
Sd(Intercept)	1.29	1.18	0.05	4.25	1.00	7858	11986

CPUA- Brook Stickleback – Predictions

Year	Treatment	Posterior Mean	Posterior SD
2018 Pre	Spaced	0.19	0.26
2018 Pre	Control	0.15	0.26
2018 Pre	Cluster	0.33	0.26
2018 Post	Spaced	0.15	0.26
2018 Post	Control	0.04	0.26
2018 Post	Cluster	0.26	0.26
2019	Spaced	0.30	0.26
2019	Control	0.11	0.26
2019	Cluster	0.37	0.26
2020	Spaced	0.18	0.26
2020	Control	0.11	0.26
2020	Cluster	0.26	0.26

Chapter 4.2 Appendices

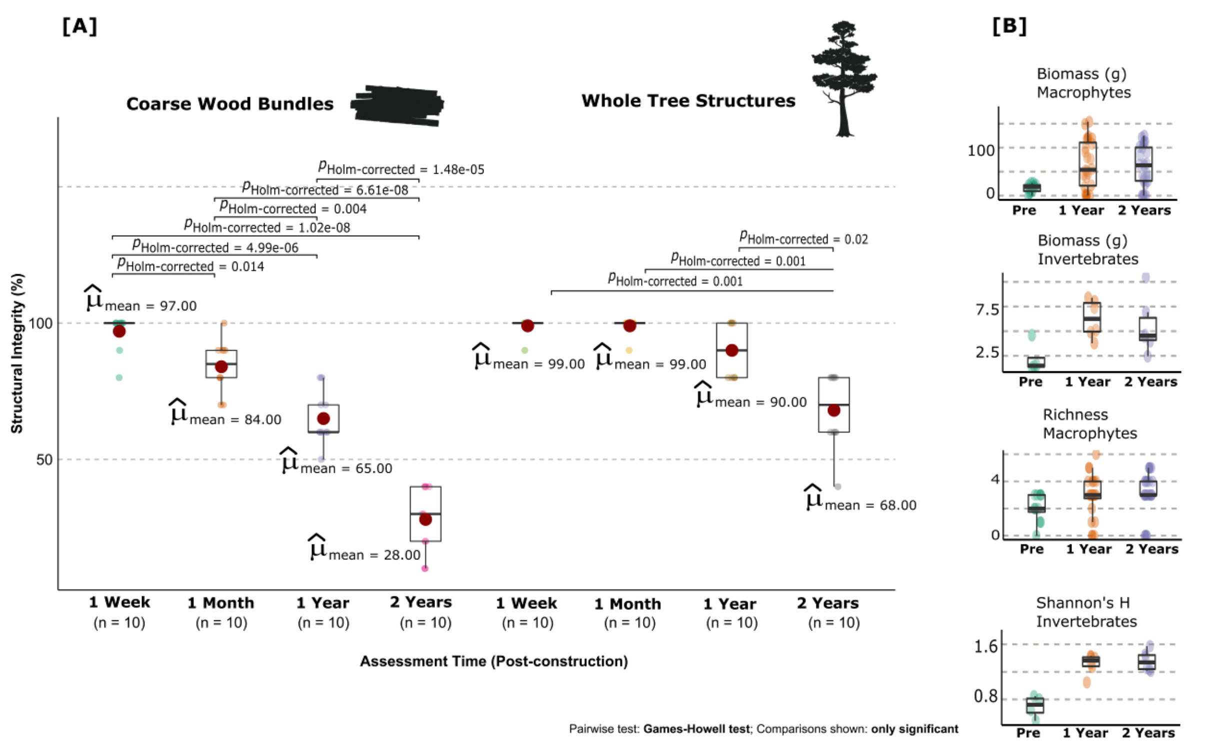


Figure A4.2.1. *Structural Integrity (STIN) assessment for coarse wood bundles and whole tree structures (A) as part of the Coarse Woody Habitat (CWH) study in Lakeland County, Northern Alberta, 1 week, 1 month, 1 year and 2 years post construction related to mean biomass, richness and diversity changes in aquatic macroinvertebrates and invertebrates (B). Significant differences in SI values are indicated by Holm corrected p-values.*

Table A4.2.1. *Structural assessment of the littoral zone of the study lake, Lake Steepbank. Coarse woody habitat (CWH) was counted in 100 m sections up to a depth of 2 m; n = 26. Large woody debris were divided into small and large categories: small = diameter 10-15 cm & 1.5 – 3 m length, large = > 15 cm diameter & > 3 m. Overall structural complexity and CWH richness was presented through the number of CWH in both size classes per surveyed area in m^2. Treatment area to be enhanced was chosen from all surveyed areas (treatment T = experimental treatment area).*

ID	Section Length m	Section Width m	LWD Count	Section Area m^2	Habitat complexity CWH/100 m^2	Small CWH	Large CWH	Treatment
1	99.8	3.2	7	319.36	2.191884	6	1	-
2	95.3	2.8	5	266.84	1.873782	5	0	-
3	94.2	2.7	8	254.34	3.145396	6	2	-
4	99	3	10	297	3.367003	9	1	-
5	100	2	5	200	2.5	4	1	-
6	98.4	2.5	8	246	3.252033	8	0	-
7	97.4	2.7	4	262.98	1.521028	4	0	-
8	93.4	4	6	373.6	1.605996	6	0	-
9	97	4.5	3	436.5	0.687285	2	1	-
10	91.7	5	2	458.5	0.436205	2	0	-
11	93.3	5.3	8	494.49	1.617828	6	2	-
12	94.6	6	3	567.6	0.528541	3	0	-
13	96.3	9	6	866.7	0.692281	5	1	-
14	89.3	9.4	8	839.42	0.953039	7	1	-
15	100	12	4	1200	0.333333	4	0	-
16	100	11.7	5	1170	0.42735	3	2	-
17	99.1	12.1	13	1199.11	1.084137	10	3	-
18	94.7	10.4	9	984.88	0.913817	8	1	-
19	95	8.9	12	845.5	1.419279	10	2	-

20	88.5	10.3	8	911.55	0.877626	7	1	-
21	100	13	4	1300	0.307692	4	0	T
22	99.5	14.4	6	1432.8	0.41876	6	0	T
23	98.7	13.8	2	1362.06	0.146836	2	0	T
24	96	14.7	3	1411.2	0.212585	3	1	T
25	93.9	15	2	1408.5	0.141995	2	0	T
26	94	14	3	1316	0.227964	3	0	T

Table A4.2.2. *Structural integrity (SI) assessment scores for coarse wood bundles and whole tree structures as part of the coarse woody habitat (CWH) study on Lake Steepbank, Lakeland County, Alberta. Percent (%) score gets subtracted from the original state of 100%, monitoring over the span of 2 years.*

Category	Characteristic	Structure Aspect	Structural Integrity %
Bundles			
1	Log detached (up to 60% for all 6)	Integrity	-10
2	Sandbag detached (up to 40% for all 4)	Anchor	-10
3	Structural move minor	Placement	-15
4	Structural move major	Placement	-30
5	Rope tear (up to 60% for 3 sides)	Integrity	-20
6	Sandbag leaking (up to 20% for all 4)	Anchor	-5
Trees			
7	Tree broken (up to 50% for both)	Integrity	-25
8	Tree shifted (up to 20% for both)	Placement	-10
9	Anchor line ripped (up to 40% for both)	Anchor	-20
10	Anchor point unearthed	Anchor	-50
11	Tree moved significantly (up 50% for both)	Placement	-25
12	Tree joint point undone	Integrity	-10

Chapter 4.2 Supplements

Table S4.2.1. *Aquatic macroinvertebrates and macrophytes identified in the enhanced and control areas during the three sampling years. Invertebrates are identified to family level and macrophytes to species level.*

Macroinvertebrates	**Aquatic Macrophytes**
Gammaridae	*Chara vulgaris*
Aeshnidae	*Ruppia cirrhosa*

Baetidae	*Elodea canadensis*
Leptoceridae	*Hippus vulgaris*
Helicopsychidae	*Ceratophyllum demersum*
Gordiidae	*Persicaria amphibia L.*
Lymnaeidae	
Sphaeriidae	
Physidae	
Planorbidae	
Lestidae	
Lepidostomatidae	

Table S4.2.2. *Cover data for macrophytes compared across treatments through Analysis of variance (Welch ANOVA adjusted for violation of assumption of homogeneity of variances, normality controlled for through Shapiro Wilk test and heteroskedasticity through Levene test) and pairwise comparison for significant factors (alpha < 0.05) with Holm adjusted p-values.*

Welch-ANOVA

Welch	n	statistic	DFn	DFd	p-adjusted
Cover	84	2.81	6	33.5	0.025

Levene test

Levene	Df	F-value	Pr(>F)
group	3	2.9125	0.03942
	80		

Pairwise comparison

	Clustered	Control	Pre
Control	0.057	-	-
Pre	0.139	1.000	-
Spaced	1.000	0.025	0.046

***Table S4.2.3**. Shannon index for Invertebrates compared through Analysis of variance (ANOVA normality controlled for through Shapiro Wilk test and heteroskedasticity through Levene test) across treatments (Pre-enhancement, Control, Clustered, Spaced) and years (2018, 2019, 2020). Pairwise comparison for significant factors (Alpha < 0.05) through post hoc tests (Games Howell, Holm adjusted p-values) and mean Shannon's H values and standard deviation (SD).*

ANOVA

	Df	Sum Sq	Mean Sq	F-value	Pr(>F)
Year	2	0.5314	0.2657	13.209	0.000492*
Treatment	2	2.1214	1.0607	52.734	1.64e-07*
Year:Treatment	2	0.0154	0.0077	0.384	0.687678
Residuals	15	0.3017	0.0201		

Games Howell

Treatments	diff	lwr	upr	p-adjusted
Control-Cluster	-0.74566667	-0.9660928	-0.5252406	0.0000001
Pre-Cluster	-0.65925000	-0.9056939	-0.4128061	0.0000030
Spaced-Cluster	-0.03616667	-0.2565928	0.1842594	0.9660131
Pre-Control	0.08641667	-0.1600272	0.3328605	0.7562904
Spaced-Control	0.70950000	0.4890739	0.9299261	0.0000002
Spaced-Pre	0.62308333	0.3766395	0.8695272	0.0000065

Years	diff	lwr	upr	p-adjusted
19-18	-0.37358333	-0.1733373	0.9205039	0.0014581
20-18	-0.42369444	-0.1232262	0.9706151	0.0004629
20-19	-0.05011111	-0.3789287	0.4791509	0.7385079

Mean Shannon's H for invertebrates and Standard deviation

Year	Treatment	Mean Diversity	SD
2018	Pre	0.41	0.21
2019	Control	0.31	0.05
2019	Cluster	1.27	0.13

2019	Spaced	1.27	0.17
2020	Control	0.4	0.05
2020	Cluster	1.23	0.03
2020	Spaced	1.3	0.14

Table S4.2.4. *Richness for Macrophytes compared through Analysis of variance (ANOVA, normality controlled for through Shapiro Wilk test and heteroskedasticity through Levene test) across treatments (Pre-enhancement, Control, Clustered, Spaced) and years (2018, 2019, 2020). Pairwise comparison for significant factors (alpha < 0.05) through post hoc tests (Games Howell, Holm adjusted p-values) and mean Shannon's H values and standard deviation (SD).*

ANOVA

	Df	Sum Sq	Mean Sq	F-value	Pr(>F)
Year	1	3.1	3.096	1.655	0.20196
Treatment	1	17.5	17.503	9.359	0.00302*
Year:Treatment	1	0.35	0.347	0.186	0.66772
Residuals	80	149.62	1.870		

Games-Howell

Treatment	diff	lwr	upr	p-adjusted
Enhanced:Unenhanced	-1	-1.593213	-0.4067868	0.0012099

Mean Macrophyte Richness and Standard deviation

Treatment	Mean Rich	SD
Pre	2.08	0.10
Control	1.96	1.02
Cluster	2.75	1.35
Spaced	3.25	1.81

Table S4.2.5*. Biomass differences for aquatic macroinvertebrates assessed through permutation test for homogeneity of multivariate dispersions (Family distance) over treatments (Pre-enhancement, Control, Clustered, Spaced) and years (2018, 2019, 2020) based on beta diversity and Bray-Curtis dissimilarity. Permutations = 999. Pairwise adonis comparison for significant factors (alpha < 0.05). Proportionate biomass (%) of individual invertebrate families listed across treatments.*

Permutational Multivariate Analysis of Variance

	Df	Sum of Sqs	Mean Sqs	F.Model	R^2	Pr(>F)
Year	1	0.1633	0.16335	2.1785	0.04944	0.127872
Treatment	3	1.9052	0.63508	8.4699	0.57669	0.000999*
Treatment:Year	2	0.1105	0.5524	0.7367	0.03344	0.570430
Residuals	15	1.1247	0.07498		0.34043	
Total	21	3.3038			1.00000	

Pairwise adonis for invertebrates

Treatment	Pr(>F)
Pre-Spaced	0.011
Pre-Cluster	0.009
Pre-Control	0.568
Spaced-Cluster	0.363
Spaced-Control	0.003
Cluster-Control	0.002

Biomass proportions invertebrates

Family	Spaced BiomassP	Cluster BiomassP	Control BiomassP	Pre BiomassP
Gam	16.23	11.07	79.63	55.22
Aesh	17.94	9.65	0	0
Baet	0.31	0.11	0.27	0.14
Lepto	3.88	2.53	0	0
Helio	2.56	2.49	1.57	4.02
Gord	3.44	2.48	7.99	8.39
Lymn	42.46	61.33	0	22.63
Sphaer	8.63	6.83	5.65	5.54
Phys	2.39	1.53	0.76	0.91

Planor	1.47	1.16	3.68	2.29
Lestid	0.17	0.26	0.14	0
Lepido	0.53	0.54	0.29	0.86

Table S4.2.6. *Biomass differences for aquatic macrophytes assessed through permutation test for homogeneity of multivariate dispersions (Family distance) over treatments (Pre-enhancement, Control, Clustered, Spaced) and years (2018, 2019, 2020) based on beta diversity and Bray-Curtis dissimilarity. Permutations = 999. Pairwise adonis comparison for significant factors (Alpha < 0.05). Proportionate biomass (%) of individual invertebrate families listed across treatments.*

Permutational Multivariate Analysis of Variance

	Df	Sum of Sqs	Mean Sqs	F.Model	R^2	Pr(>F)
Treatment	3	4.5418	1.51393	9.7360	0.29986	0.000999
Year	1	0.0788	0.07885	0.5071	0.00521	0.729271
Treatment:Year	1	0.1074	0.05370	0.3453	0.00709	0.939061
Residuals	2	10.1484	0.15550		0.68785	
Total	73	15.1465			1.00000	

Pairwise adonis for macrophytes

Treatment	Pr(>F)
Pre-Spaced	0.001
Pre-Cluster	0.001
Pre-Control	0.093
Spaced-Cluster	0.096
Spaced-Control	0.001
Cluster-Control	0.001

Biomass proportions macrophytes

Species	Spaced BiomassP	Cluster BiomassP	Control BiomassP	Pre BiomassP
Chara	17.8	25.99	49.03	61.19

Ruppia	22.18	20.56	44.6	27.16
Persi	45.58	43.49	0	0
Cerat	6.41	5.86	0	0
Elodea	5.51	3.61	3.69	6.1
Hippus	2.52	0.48	2.67	5.55

Table S3.27*. Permutation test for dispersion based on results from a Principal Coordinates Analysis (PCoA, = Multidimensional scaling, MDS) for invertebrates and macrophytes (frequency data across treatments). Average distance to centroid for macrophytes and invertebrates listed across treatments (Pre-enhancement, Control, Clustered, Spaced).*

Permutation test invertebrates

	Df	Sum of Sqs	Mean Sqs	F N.Perm	Pr(>F)
Treatment	3	0.007828	0.002609	4.0107	0.021
Residuals	18	0.011711	0.000650		

Permutation test macrophytes

	Df	Sum of Sqs	Mean Sqs	F-value	Pr(>F)
Treatment	3	0.6028	0.20094	0.9825	0.339
Residuals	70	1.43169	0.020453		

Average distance to centroid - Invertebrates

Cluster	Control	Pre	Spaced
0.1450	0.1011	0.1187	0.1321

Average distance to centroid - Macrophytes

Cluster	Control	Pre	Spaced
0.3310	0.2781	0.3397	0.3469

Table S4.2.8*. SDR-Simplex analysis for invertebrates and macrophytes comparing SDR components (Similarity, Richness Difference, Replacement) and their relative importance between enhanced and unenhanced areas* *(pairwise comparison of each component, alpha < 0.05, relative importance stated in percent across components and significant components marked).*

Pairwise comparison of simplex components for Invertebrates

Repl (R)	Df	Sum Sq	Mean Sq	F-value	Pr(<F)
En:Unen	1	363.7	363.7	293.2	<2e-16
Residuals	109	135.2	1.2		

Dif (D)	Df	Sum Sq	Mean Sq	F-value	Pr(<F)
En:Unen	1	8.68	8.684	7.131	0.00874
Residuals	109	132.74	1.218		

Sim (S)	Df	Sum Sq	Mean Sq	F-value	Pr(<F)
En:Unen	1	15.32	15.31	11.43	0.001
Residuals	109	146.11	1.34		

Relative importance of simplex components for Invertebrates and significant components

Treatment	R	D	S	Sign
Unenhanced	56.9	22.4	20.7	R, D, S
Enhanced	64.3	17.5	18.2	

Pairwise comparison of simplex components for Macrophytes

Repl (R)	Df	Sum Sq	Mean Sq	F-value	Pr(<F)
En:Unen	1	43.1	43.1	33.27	<2e-16
Residuals	1627	2108.1	1.3		

Dif (D)	Df	Sum Sq	Mean Sq	F-value	Pr(<F)
En:Unen	1	116.4	116.43	105	<2e-16
Residuals	1627	1804	1.11		

Sim (S)	Df	Sum Sq	Mean Sq	F-value	Pr(<F)
En:Unen	1	132.5	1132.5	95.72	<2e-16
Residuals	1627	2252.9	1.38		

Relative importance of simplex components for Macrophytes and significant components

Treatment	R	D	S	Sign
Unenhanced	47.9	23.8	28.3	R, D, S
Enhanced	40.7	25.9	33.4	

Table S4.2.9. *Kruskal Wallis rank sum test for identifying significant interactions for Structural Integrity (STIN) over time for coarse wood bundles and whole tree structures as part of the Coarse Woody Habitat (CWH) study on Lake Steepbank, Lakeland County, Alberta. STIN score calculated through Table 1 Appendix over a span of 2 years.*

Wood bundles	Integrity of Bundles ~ Year		
	Df	Chi-squared	Pr (>F)
Year	3	33.936	<0.001

Whole trees	Integrity of Whole trees ~ Year		
	Df	Chi-squared	Pr (>F)
Year	3	27.176	<0.001