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Thèse présentée en vue de l'obtention du diplôme de *Philosophiæ Doctor*

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présentée par **Gabrielle TROTTIER**

en vue de l'obtention du diplôme de *Philosophiæ Doctor*

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DEDICATION

To me, just me, my perseverance and my patience.

« À moi, juste moi, ma persévérance pis ma patience. »

"We pay a price for everything we get or take in this world; and although ambitions are well worth having, they are not to be cheaply won, but exact their dues of work and self-denial, anxiety and discouragement."

Lucy Maud Montgomery, Anne of Green Gables

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RÉSUMÉ

Les écosystèmes aquatiques, via l'approvisionnement en eau douce pour la consommation humaine, l'irrigation des terres, le transport et la production d'énergie, sont une ressource essentielle à la survie des humains. L'intégrité et la fonction de ces écosystèmes dépendent de leur état général et de la biodiversité qu'on y retrouve. Malgré leur dépendance aux écosystèmes aquatiques, les humains impactent ces écosystèmes lorsqu'ils accumulent l'eau douce dans des réservoirs et l'utilisent pour la production d'énergie via l'hydroélectricité. En effet, ces usages de l'eau douce conduisent à la dégradation et la destruction de l'habitat, et à la modification du régime des flux hydrologiques, entre autres menaces. Depuis plusieurs années, la communauté scientifique a étudié les impacts de l'hydroélectricité, des barrages et des réservoirs sur la biodiversité aquatique. Toutes ces études, dans leur ensemble, ont permis de souligner une très grande fourchette d'impacts (tant en termes de direction [positive ou négative] que de magnitude) dépendante de multiples facteurs tels le groupe taxonomique étudié, le biome et le type de production hydroélectrique, par exemple.

L'Analyse du Cycle de Vie (ACV), une méthodologie basée sur la science naturelle, est utilisée pour évaluer les impacts environnementaux potentiels d'un produit, d'un procédé ou d'un service, sur l'entièreté de son cycle de vie, c'est-à-dire de l'extraction des ressources à la fin de vie. Ainsi, pour générer une évaluation des impacts représentative de la réalité, les impact potentiels associés à chacune des phases du cycle de vie d'un produit (procédé ou service), incluant les impacts associés à la source d'énergie utilisée pour générer ces produits et services, doivent être pris en compte. Malgré le nombre croissant de preuves concernant les impacts de l'hydroélectricité, des barrages et des réservoirs sur la biodiversité aquatique, c'est seulement récemment que l'ACV a commencé à prendre en compte ces impacts dans ses bilans. Précédemment, ces impacts sur la biodiversité aquatique n'étaient pas évalués et donc, considérés comme nuls, ce qui est loin de la réalité. La prise en compte de ces impacts pourrait influencer la place qu'occupe l'hydroélectricité en termes d'impact environnementaux potentiels, lorsque comparée à d'autres sources d'énergie.

Trois problématiques ont mené à la genèse de ce projet de recherche doctoral; 1) dans une synthèse de la littérature, l'hydroélectricité, les barrages et les réservoirs ont un impact sur la biodiversité aquatique, mais il n'y a pas de consensus au sujet de la direction et la magnitude de ces impacts, 2) les Facteurs de Caractérisation (FC) sur l'impact de l'hydroélectricité et de l'occupation des

réservoirs sont exclusifs aux poissons et/ou basés sur des courbes théoriques plutôt que des données empiriques, et 3) les FCs utilisent la fraction d'espèces (résolution taxonomique à l'espèce) potentiellement disparues ou « Potentially Disappeared Fraction (PDF) of species », mais il n'y a toujours pas d'étude qui évalue l'influence de la résolution taxonomique sur les FCs. Ce projet est donc détaillé en trois objectifs :

1. Qualifier et quantifier les impacts de l'hydroélectricité, des barrages et des réservoirs sur la biodiversité aquatique, en utilisant la richesse des macroinvertébrés, via une robuste revue de la littérature et suivie d'une méta-analyse.
2. Développer des FCs régionalisés basés sur des données empiriques pour évaluer l'impact de la transformation du lit d'une rivière en réservoir et l'occupation subséquente du lit de la rivière par le réservoir, ci-après nommé impact de l'occupation des réservoirs, sur la richesse des macroinvertébrés, en ACV.
3. Évaluer l'influence de la résolution taxonomique sur les FCs précédemment développés.

Pour le premier objectif, un cadre de travail méta-analytique a été utilisé pour identifier la direction et la magnitude globale des impacts de l'hydroélectricité sur la richesse des macroinvertébrés. Les résultats ont montré une grande variété de réponses des macroinvertébrés à l'hydroélectricité, plus spécifiquement des réponses significativement négatives et positives, ainsi que des réponses non-significativement différentes de zéro. La méta-analyse n'a pas démontré un effet global significatif de l'hydroélectricité sur la richesse des macroinvertébrés due à la grande variabilité de réponses observées. La détection d'un impact significatif a probablement été entravée par la grande quantité de bruit environnemental et méthodologique. C'est donc pourquoi l'influence de certaines sources d'hétérogénéité a été évaluée, par exemple le biome, le type d'impact, le design d'étude, ainsi que la saison et l'engin d'échantillonnage. Toutefois, aucune de ces sources d'hétérogénéité n'a démontré d'effet significatif, ce qui force à croire que d'autres variables environnementales, qui pourraient influencer la richesse des macroinvertébrés, n'ont pas été ciblées, par exemple des processus environnementaux de plus petite envergure, comme la granulométrie, la température, la perturbation par les vagues et la présence/absence de plantes aquatiques dans l'habitat.

Pour le deuxième objectif, une approche de substitution espace-temps a été utilisée pour développer des FCs (en PDF·m^2·an/m^2·an) qui évaluent les impacts de l'occupation des réservoirs sur la richesse des macroinvertébrés à trois échelles spatiales; le réservoir, les écorégions et le pays (États-Unis), ainsi qu'un modèle empirique pouvant expliquer la variabilité observée dans les CFs des réservoirs, à l'aide de seulement quelques variables environnementales associées aux dits réservoirs. Au sein de cet objectif, il a été possible d'observer une perte statistiquement significative de 28% de la richesse des macroinvertébrés au sein des États-Unis suite à l'occupation du lit d'une rivière par un réservoir, un gradient longitudinal des CFs spécifiques aux écorégions, où les CFs sont plus élevés dans l'ouest que dans l'est, et enfin que l'état trophique, l'aire du réservoir et son élévation peuvent expliquer la variation observée dans les CFs spécifiques aux réservoirs, où un CF est plus élevé dans un réservoir large, oligotrophe et situé en haute élévation. Les résultats de cette étude ont démontré un support important pour la régionalisation des FCs, ont montré que les FCs étaient uniformes et constants à travers les échelles spatiales (*c.-à-d.*, l'échelle du réservoir, de l'écorégion et du pays) et enfin, ont fourni un modèle empirique simple basé sur l'élévation, l'état trophique et l'aire du réservoir qui pourrait être utilisé par les développeurs et les praticiens ACV pour estimer un PDF quand des données de richesse de macroinvertébrés ne sont pas disponible.

Pour le troisième objectif, qui est considéré comme un complément au second objectif, les FCs (en PDF·m^2·an/m^2·an) ont été calculés en utilisant des données de richesse de macroinvertébrés résolues en utilisant deux niveaux de résolutions taxonomiques (*c.-à-d.*, basées sur le genre ou la famille) et incluant un critère d'inclusivité (*c.-à-d.*, exclusif ou inclusif). Les résultats suggèrent que l'utilisation d'une résolution taxonomique au genre ou à la famille transmettent des messages similaires. Dans les deux cas, la régionalisation des FCs est statistiquement significative, la directionnalité et la magnitude des FCs spécifique aux écorégions sont très semblables, et les modèles empiriques sont aussi similaires en termes de sélection de modèle et capacité de prédiction. Basé sur ces résultats, il est possible de recommander l'utilisation d'une résolution taxonomique au genre-inclusive puisque celle-ci explique une quantité de variation additionnelle dans les FCs et est considérée comme plus représentative de la richesse des écosystèmes échantillonnés, en comparaison à une résolution taxonomique à la famille et/ou exclusive. Cet article est de nature plutôt fondamentale et méthodologique et a d'importantes implications pour les praticiens ACV puisqu'il pourrait permettre à certains groupes d'organismes, rarement

identifiés à l'espèces, d'être éventuellement inclus dans de nouveaux FCs, ce qui augmenterait la représentativité et la robustesse des FCs pour l'Aire de Protection (AdP) de la qualité des écosystèmes en ACV.

Cette thèse est très versatile et présente plusieurs attributs. Certains chapitres sont plutôt de nature fondamentale, alors que d'autres ont des résultats surtout pratiques. Le choix des macroinvertébrés comme organisme de recherche est considéré comme original, spécifiquement dans la communauté ACV. D'autant plus, cette thèse démontre des caractéristiques de nouveauté, plus spécialement dans le dernier article. À notre connaissance, l'évaluation de l'influence de la résolution taxonomique sur les FCs n'a jamais été fait auparavant. La multidisciplinarité de cette thèse lui confère un avantage majeur et ses implications sont d'autant plus pertinentes, profondes et complexes. Elle intègre des données biologiques dans un outil d'évaluation des impacts qui est dirigé pour la prise de décision dans un contexte de développement durable de produits, procédés et services. La contribution majeure de cette thèse est les FCs opérationnels qui permettent d'estimer l'impact de l'occupation par un réservoir sur la richesse des macroinvertébrés à trois échelles spatiales, c'est-à-dire les États-Unis, neuf écorégions et de nombreux réservoirs.

ABSTRACT

Freshwater ecosystems are an essential resource to the survival of humans, namely through drinking and irrigation water, transportation and energy generation. The integrity and the function of these ecosystems strongly depend on their general state and biodiversity. Despite their dependance to these ecosystems, humans store freshwater into reservoirs and use it to produce energy via hydropower. This freshwater use leads to habitat degradation and destruction, and flow regime modification, amongst other threats. The scientific community has been studying the impacts of hydropower, dams and reservoirs on freshwater biodiversity for years now and a synthesis of these outcomes highlighted a wide range of impacts (both in their direction [positive or negative impact] and magnitude) dependent of multiple factors, such as taxonomic group studied, biomes and hydropower types, for instance.

Life Cycle Assessment (LCA) is a methodology based on natural science that is used to evaluate the potential environmental impacts of a product, process or service, from resource extraction to end-of-life. Thus, to get an assessment that is representative of reality, we need to be able to account for the potential impacts of each phase in a product (or process, or service) life cycle. This includes the impacts associated with the energy source used to create these goods or services. Despite the mounting evidences regarding the impacts of hydropower, dams and reservoirs on freshwater biodiversity, LCA has only just started to account for these impacts into their assessment. Before that, impacts to freshwater biodiversity were not assessed and thus considered null, which is far from the reality. Accounting for theses impacts could influence the overall ranking of hydropower in terms of potential environmental impacts, when compared to other energy sources.

Three issues led to the genesis of this doctoral research project; 1) in the review of the literature, hydropower, dams and reservoirs impact freshwater biodiversity but there is no consensus with regards to the direction (*i.e.*, positive or negative) and the magnitude (*i.e.*, delta) of these impacts, 2) current Characterization Factors (CF) for the impacts of hydropower and reservoir occupation are exclusive to fish and/or based on theoretical curves instead of empirical data, and 3) CFs use Potentially Disappeared Fraction (PDF) of species (species level of taxonomic resolution), but there is still no study assessing the influence of taxonomic resolution (genus or family) on CFs. Thus, this project consists in three specific objectives:

1. Qualifying and quantifying the impacts of hydropower, dams and reservoirs on freshwater biodiversity, using macroinvertebrate richness, through a thorough review of the literature and followed by a meta-analysis.
2. Developing regionalized CFs, using empirical data, to assess the impact of a riverbed transformed and occupied by a reservoir, hereafter referred to as the impact of reservoir occupation, on macroinvertebrate biodiversity (*i.e.*, richness), within LCA.
3. Evaluating the influence of taxonomic resolution on previously developed CFs.

For the first objective, we used a meta-analytic framework to uncover the overall direction and magnitude of the impacts of hydropower on freshwater macroinvertebrate richness. Our results showed a wide range of responses of macroinvertebrates to hydropower, dams and reservoirs, namely significantly negative, significantly positive and non-significant responses. Because of that large variability of responses, the meta-analysis did not show a significant overall impact of hydropower, dams and reservoirs on macroinvertebrate richness. Lots of environmental and methodological noise might have hindered the detection of a significant impact, which is why we also investigated the influence of some sources of heterogeneity and accounted for variables such as biomes, impact type, study design, as well as sampling seasons and gears. None of these variables showed a significant effect, which lead us to think that we may not have targeted the appropriate variables that could influence the richness of macroinvertebrates and that these organisms may be impacted by much finer-scale processes, such as granulometry, temperature, wave disturbance and macrophytes.

For the second objective, we used a space-for-time substitution approach and developed empirical CFs (in PDF·m^2·yr/m^2·yr), to assess the impacts of reservoir occupation on macroinvertebrate richness at three spatial scales; reservoirs, ecoregions and the United States, as well as built an empirical model to explain the variability observed in reservoir PDF with only a few reservoir-related environmental variables. In this objective, we observed a statistically significant loss of macroinvertebrate richness of 28% following reservoir occupation in the United States, that CFs specific to ecoregions followed a longitudinal gradient of impact, from lowest in the east to highest west, and that elevation, trophic state and reservoir surface area could be used to explain the variation observed in reservoir-specific CFs, where high elevation, oligotrophic and large

reservoirs had higher CF.[GT1] Our results provided strong support for regionalized CFs, showed that CFs were uniform and consistent across scales (*i.e.*, reservoir, ecoregion and country scales) and offered a simple empirical model based on reservoir elevation, trophic state and surface area to be used by LCA modelers and practitioners to estimate CF where macroinvertebrate richness data is not available.

For the third objective, which is considered as a complement to the second objective, we calculated CFs (in PDF·m^2·yr/m^2·yr) using richness data resolved at two levels of taxonomic resolution (*i.e.*, genus- and family-based) and including a criterion of inclusiveness (*i.e.*, exclusive and inclusive). Results in this objective suggest that using either genus- or family-based taxonomic resolutions conveys analogous messages.[GT2] In both cases, regionalization of CFs is significant, directionality and magnitude of ecoregion CFs are very similar and empirical models are also similar with regards to model selection and predictive abilities. Based on these outcomes, we recommend favoring genus-inclusive taxonomic resolution because they explained additional variation in CF and were more representative of the richness observed in the sampled ecosystems, as opposed to family-based and/or exclusive categories of taxonomic resolutions. The nature of this article is more fundamental and methodological. It has important implications for Life Cycle Impact Assessment (LCIA) practitioners because it could potentially allow the inclusion of certain groups of organisms, which are rarely identified up to the species level of taxonomic resolution, to be included in new CFs, thus enhancing the representativity and robustness of CFs for the ecosystems quality Area of Protection (AoP) in LCA.

This thesis is very versatile and has many attributes. Some chapters are more fundamental, others have practical outputs. The choice of macroinvertebrate as research taxa is considered original, especially in the LCA community. Moreover, this thesis also showcases a novelty characteristic, especially the last article. To our knowledge, evaluating the influence of taxonomic resolution on CFs had never been done in the past. A major advantage of this thesis is its multi-disciplinarity, which makes its outputs especially relevant, deep and complex. It integrated biological data into an impact assessment tool aimed for decision making in the context of environmental sustainability of products, processes and services. The main contribution of this thesis is operational CFs assessing the impact of reservoir occupation on macroinvertebrate richness at three spatial scales, namely the United States, nine ecoregions and numerous reservoirs.

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LIST OF SYMBOLS AND ABBREVIATIONS

ACV	Analyse du Cycle de Vie
AdP	Aire de Protection
AIC	Akaike Information Criterion
ANOVA	Analysis of Variance
AoP	Area of Protection
A&HCI	Arts & Humanities Citation Index
BACI	Before-after Control-impact
BIC	Bayesian Information Criterion
CSBQ	Centre de la Science de la Biodiversité du Québec
CF	Characterization Factor
CH_4	Methane
CI	Confidence Interval
CIRAIG	Centre international de référence sur le cycle de vie des produits, procédés et services
CO_2	Carbon dioxide
CPCI-S	Conference Proceedings Citation Index – Science
CPCI-SSH	Conference Proceedings Citation Index – Social Science & Humanities
CPL	Coastal Plains
CV	Cross-validation
df	Degrees of freedom
DOC	Dissolved Organic Carbon
DGGE	Denaturing Gradient Gel Electrophoresis
EC	Effective Concentration
EDDEC	Institut de l'Environnement et le Développement Durable et l'Économie Circulaire
EF	Effect Factor
EPT	Ephemeroptera-Plecoptera-Trichoptera
ESCI	Emerging Sources Citation Index
FC	Facteur de Caractérisation
FF	Fate Factor

FQRNT	Fonds Québécois de la Recherche sur la Nature et les Technologies
GHG	Greenhouse Gas
GW	Gigawatt
GWP	Global Warming Potential
HDH	Habitat Diversity Hypothesis
ISO	International Standardization Organization
kWh	Kilowatt-hour
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LULUC	Land-use and Land-use Change
MSA	Mean Species Abundance of original species
NAP	Northern Appalachians
NLA	National Lake Assessment
NPL	Northern Plains
NRSA	National Rivers and Streams Assessment
NSERCC	National Science and Engineering Research Council of Canada
NTNU	Norwegian University of Science and Technology
OP	Observed versus Predicted
PDF	Potentially Disappeared Fraction
PRISMA	Preferred Reporting Items for Systematic and Meta-analyses
RCT	Randomized Controlled Trials
RKFCV	Repeated k-fold Cross-validation
RMSE	Root Mean Squared Error
ROR	Run-of-river
RVE	Robust Variance Estimation
SAP	Southern Appalachians
SAR	Species-area Relationship
SCI-EXPANDED	Science Citation Index Expanded
SD	Standard Deviation
SDR	Species-discharge Relationship

SE	Standard Error
SETAC	Society of Environmental Toxicology and Chemistry
SF	Severity Factor
SPL	Southern Plains
SSCI	Social Sciences Citation Index
TNL	Total Nitrogen Level
TPL	Total Phosphorus Level
TS	Trophic State
USA	United States
USEPA	United States Environmental Protection Agency
UMW	Upper Midwest
WBS	Work Breakdown Structure
WMT	Western Mountains
WoS	Web of Science
XER	Xeric
XF	Exposure Factor

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CHAPTER 1 INTRODUCTION

Freshwater ecosystems represent a vital resource for humans (*i.e.*, drinking and irrigation water, transportation and energy production). They support a rich biota, that is highly endemic and sensitive to human impacts, and their function and integrity often depend on species richness and composition (Strayer & Dudgeon, 2010), indices used to estimate biodiversity. The large and growing demands on freshwater resources has led to the imperilment or the extinction of multiple freshwater species (Ricciardi & Rasmussen, 1999; Strayer & Dudgeon, 2010). Moreover, freshwater biodiversity is particularly vulnerable to environmental changes and human impacts, more so than their terrestrial and marine counterparts, given its disproportionate richness to area ratio (Gleick, 2011; Sala et al., 2000).

Two of the biggest threats to freshwater biodiversity are flow alteration and habitat degradation (Dudgeon et al., 2006), both of which are commonly associated with hydropower, dams and reservoirs through multiple pathways (*i.e.*, freshwater habitat degradation, water quality alteration and land use change; Gracey & Verones, 2016). At the end of 2019, approximately 13 000 hydropower stations were producing a total of 1038 gigawatts (GW) in more than 150 countries (IHA, 2020a). Thus, it is important to understand how hydropower, dams and reservoirs impact freshwater biodiversity.

Life Cycle Assessment (LCA) is a methodology used to assess the potential environmental impacts of a product, process or service during its entire life cycle, from resource extraction to end-of-life (ISO, 2006a). It is used, amongst other methodologies, to evaluate and compare energy production sources. However, current Life Cycle Impact Assessment (LCIA) methodologies are still in their infancy with regards to the assessment of the potential environmental impacts of hydropower, dams and reservoirs on freshwater biodiversity. As a consequence, in the actual practice of LCA, hydropower, dams and reservoirs are generally considered as having no associated environmental impacts on freshwater biodiversity. This mainly results from the methodological gap in LCIA, where the impacts of hydropower on freshwater biodiversity are not actually accounted for, and not from the lack of empirical evidences, as seen in other studies (*e.g.*, Santucci et al., 2005; Poff et al., 2007; Liermann et al., 2012). [GT3]

To our knowledge, only a few studies attempted to evaluate changes in fish richness in relation to hydropower, dams and reservoirs within the LCA framework (see Turgeon et al., 2021 and Dorber

et al., 2019). The work of Dorber et al. (2019) relies on theoretical species richness curves, such as Species-discharge Relationship (SDR; Dorber et al., 2019) or Species-area Relationship (SAR), which are based on ecosystems that are in a state of equilibrium and are not representative of the biological reality in an environment impacted by hydropower (Rosenberg et al., 2000; Xenopoulos & Lodge, 2006). The work of Turgeon et al. (2021) is a first attempt to build Characterization Factors (CF) based on empirical data, but so far has only been built using fish species richness. Ideally, robust impact characterization would rely on both empirical data and multiple (more than one) group of organisms.

This project aims to 1) understand the impacts of hydropower, dams and reservoirs on macroinvertebrate biodiversity, 2) generate regionalized CFs that translate the impacts of reservoir occupation into biodiversity impact indicators in the LCA framework, as well as build an empirical model that allows LCA developers to use few reservoir-related environmental variables to estimate reservoir-specific CFs without macroinvertebrate richness data[GT4] and finally, 3) evaluate the influence of taxonomic resolution on CFs and empirical models.

CHAPTER 2 LITERATURE REVIEW

To contextualize this doctoral thesis, we start by presenting a thorough review of the literature. In the following pages, the reader will find a discussion on the importance of biodiversity in freshwater ecosystems, as well as a review of their biggest threats, a summary of the impacts of hydropower, dams and reservoirs on freshwater ecosystems and their biodiversity, a definition of the LCA framework and a detailed summary of how the impacts of hydropower, dams and reservoirs on freshwater ecosystems are modelled within this specific framework. A close read of this chapter should allow readers from various background to grasp and understand the significance of the contribution of this doctoral thesis to the scientific community.

2.1 Biodiversity and freshwater ecosystems threats

The Millennium Ecosystem Assessment (2005) defines biodiversity as "the variability among living organisms from all sources, including terrestrial, marine, and other aquatic ecosystems, and the ecological complexes of which they are part; this includes diversity within the individuals of a species, among species, and across ecosystems".

2.1.1 Roles of biodiversity for ecosystems quality and human health

The value of biodiversity is first and foremost intrinsic and then, instrumental. An intrinsic value refers to the "value given for the sake of existence in itself" (Verones et al., 2017), for example loss of value associated with the non-existence of a species, whereas as an instrumental value refers to the "value that has a clear utility to humans and is defined from an anthropocentric standpoint" (*e.g.*, socio-economic assets and ecosystem services; Verones et al., 2017). The intrinsic value of biodiversity is primordial to ecosystem integrity and was historically used as an argument for biodiversity conservation (Hungate & Cardinale, 2017). However, nowadays, the instrumental point of view is often perceived as a much more powerful motivator for biodiversity conservation (Hungate & Cardinale, 2017). Indeed, biodiversity plays a determinant role, whether it is as a regulator of ecosystem processes, as a final ecosystem service or even as a good that has an economic or cultural value (Mace et al., 2012). As summarized by Cardinale et al. (2012), there is evidence that biodiversity loss reduces the ecological community's efficiency to capture biologically essential resources, to produce biomass, recycle and decompose essential nutrients (Balvanera et al., 2006; Cardinale et al., 2011; Cardinale et al., 2006; Quijas et al., 2010). As diverse

ecosystems contain key species that can have effects on productivity and multiple functional traits that increase resource capture (Cardinale et al., 2012), many ecological processes, such as nutrient and carbon cycling, primary production, and energy flows, are promoted by an enhanced biodiversity. Those processes are important for both ecosystems quality (*e.g.*, long-term viability and integrity) and human health (*e.g.*, clean water, food provision and climate change regulation; Duraiappah et al., 2005). Thus, a change in biodiversity is likely to impact ecosystem processes (Covich et al., 2004), which could imperil the goods and services that they provide and diminish the ecological and cultural value given to these ecosystems (Costanza et al., 1997).

2.1.2 Threats to freshwater ecosystems

Humans are transforming freshwater ecosystems for urbanization, industrialization, and water flow management (*i.e.*, reservoirs and irrigation). Even if these transformations serve human condition around the globe, they come with potential impacts on ecosystems and their biodiversity (Vörösmarty et al., 2010). This has lead freshwater ecosystems to be the most endangered, their biodiversity loss being even more important than their terrestrial or marine counterparts (Sala et al., 2000), because of their disproportionate richness to area ratio. Indeed, almost 6% of all scientifically described species live in 0.8% of the world's surface (Gleick, 2011; Hawksworth et al., 1995; Sala et al., 2000), this makes freshwater ecosystems especially vulnerable to human and environmental impacts. There are five groups of threats to the global freshwater biodiversity: 1) overexploitation, 2) water pollution, 3) exotic species invasion, 4) habitat degradation and/or destruction, and 5) flow regime modification (Dudgeon et al., 2006). Hydropower, dams and reservoirs lead to three of those five global threats, that is flow regime modification (*e.g.*, drawdown and hydropeaking), habitat degradation and/or destruction (*e.g.*, free-flowing to stagnant water, low oxygen and water turbidity), and the facilitated invasion of exotic species (Johnson et al., 2008).

2.2 Hydropower as an energy source

Hydropower is produced using the kinetic potential of water. It transforms water potential into mechanical power through turbines, which is then transformed into electricity via alternators (EIA, 2020b). There are four different types of hydropower production: 1) reservoirs-storage, with a potential distinction for alpine reservoirs, 2) run-of-river (ROR), 3) pump-storage, and 4) offshore power (*i.e.*, tidal currents and power of waves; IHA, 2020b).

2.2.1 Global and North American portraits

Since the 1930s, more and more dams have been constructed, a trend that peaked in the 1970s, when up to three large dams where commissioned everyday somewhere on the globe (WCD, 2000). This is due to the fact that hydropower, dams and reservoirs have played an important role in energy production but also, have rendered food production, flood control, and domestic use of water available to numerous communities worldwide (WCD, 2000). In 2019, more than 150 countries produced a total of capacity of 1038 GW across 13 000 hydropower stations, that is 16% of the world's electricity (IHA, 2020a). This hydropower potential is however not equally distributed around the world (Darmawi et al., 2013). Close to 50% of the world's hydropower is produced within four countries: China (27.2%), Brazil (8.3%), the United States (7.9%) and Canada (6.2%; IHA, 2020a). Some countries (and provinces), like Canada (and Québec), Switzerland, Iceland, Brazil and Norway, currently have most of their electricity produced via hydropower, 60% (and 94%) (Natural Resource Canada, 2020; Whitmore & Pineau, 2021), 60% (Confédération suisse, 2019), 70% (Richter, 2020), 75% (EIA, 2020a), and 90% (Norwegian Ministry of Petroleum Energy, 2021), respectively.

In the United States, the energy production mix, as of 2019, was mainly relying on non-renewable sources of energy, namely 38% natural gas, 23% coal, 20% nuclear and 17% renewable (7.3% wind, 6.6% hydropower, 1.8% solar, 1.4% biomass and 0.4% geothermal; EIA, 2020a). In comparison, Canada's energy production mix in 2018 was a lot more renewable; 60% of the energy produced is from hydropower, 15% nuclear, 11% fossil fuel, 7% coal and another 7% of non-hydro renewable energy (Natural Resource Canada, 2020). Over 40% of Canada's hydraulic resources are located in Québec, that is over a million lakes and 130 000 rivers, which together make up for 12% of the province entire territory (Hydro-Québec, 2021b). The province of Québec is one of the most freshwater abundant region of in the world (MDDELCC, 2021) and produces more hydropower than the other Canadian provinces (MERN, 2016). Unsurprisingly, one of the biggest hydropower complex worldwide is nested in the northern part of the province, the "La Grande" complex (MERN, 2016). Out of the 63 hydropower stations in Québec, 28 are reservoir-based (44%) and are responsible for most of the electricity produced in the province (Hydro-Québec, 2021a). In 2019, Québec's energy mix was 94.0% hydropower, 5.3% wind-powered, 0.7% biogas and biomass and 0.3% fossil fuels (Whitmore & Pineau, 2021).

2.2.2 Renewable energy and advantages

Hydropower is the most important source of renewable electricity across the world (*i.e.*, >1300 GW, 16% of the global electricity production; IHA, 2020a), and is considered the most price-competitive and reliable (Brown et al., 2011; Scherer & Pfister, 2016). Moreover, it is also flaunting the highest payback ratio among all renewable energy sources, thus making it a very efficient (*i.e.*, 95% of available energy is converted into electricity, compared to 60% for the best fossil fuel energy source; CHA, 2016) and a financially interesting option (Raadal et al., 2012). Hydropower clearly has some advantages over other renewable energy sources (*e.g.*, wind, solar, tide, and geothermal energy), especially in terms of flexibility (*i.e.*, reservoirs provide built-in energy for a quick response to higher power demands; Armaroli & Balzani, 2011; Kaygusuz, 2016) and greenhouse gases (GHG) emissions. In a Canadian boreal-temperate context, hydropower has the same GHG emissions as wind power, four times less than solar power, and 100 times less than coal-fired power (Hydro-Québec, 2021b). These GHG emissions could, however, be a lot more important in productive, nutrient-rich reservoirs[GT5] (*i.e.*, temperate, sub-tropical and tropical ecosystems; Deemer et al., 2016), where there is more dissolved organic carbon and less oxygen, conditions that contributes to higher fractions of algae-derived organic matter likely to enhance methane (CH_4) production. CH_4 emissions, which are responsible for 79% of reservoir's radiative forcing over a 100 years time horizon (Deemer et al., 2016), and carbon dioxide (CO_2) have important implications regarding climate change. Thus, in a Canadian-boreal context (or more generally in in less productive, nutrient-poor reservoirs), hydropower appears to be a better alternative to fossil fuel power.

2.3 Hydropower impacts freshwater ecosystems

Freshwater biodiversity is affected by hydropower, dams and reservoirs at all stages of hydropower's life cycle (Gracey & Verones, 2016). Indeed, hydropower infrastructures (*i.e.*, dams, reservoirs, power stations, and access roads) lead to several physical alterations such as land use change and river fragmentation, which in turn affect freshwater biodiversity (Gracey & Verones, 2016; Renöfalt et al., 2010). Moreover, to produce hydropower, natural flow regimes may have to be modified, thus altering the quality and the quantity of water available (Gracey & Verones, 2016).

2.3.1 Physical alterations

Building a dam creates a physical obstacle and fragments the original hydrological network, both longitudinally and laterally. Longitudinal fragmentation can challenge or impede organisms migration and/or dispersal (and/or lead to direct mortality, when organisms are caught in turbines; Nilsson et al., 2010). Longitudinal fragmentation can also alter the sediment and nutrient dynamics. Substantial amount of sediment/organic matter and nutrients are retained against the dam, which disrupts downstream sediment dynamics and nutrient supplies (Kummu & Varis, 2007; Ward & Stanford, 1995). This reduced sediment discharge is also potentially responsible for the sediment deprivation downstream of the dam and all the way to the coastal margin, leading to increased erosion (Liquete et al., 2004). Lateral fragmentation follows reservoir filling and impairs the natural flow of sediments, nutrients and gametes from the riverine environment to the riparian and/or floodplain zone (Renöfalt et al., 2010).

2.3.2 Flow alterations

Flow is one of the most important variables of riverine physical habitats (Poff et al., 1997). It influences the quality of riverine habitats, and thus, is a critical determinant of species assemblage. Modifying hydrological flows can affect freshwater biodiversity through various mechanisms such as mismatch timing in life history strategies, altered longitudinal and lateral connectivity that lowers riverine species viability and, facilitated introduction of exotic species (Bunn & Arthington, 2002). These modified hydrological flows can also degrade water quality, namely dissolved oxygen, pH, organic carbon, turbidity and temperature (Friedl & Wüest, 2002; Santucci et al., 2005). These changes in water quality have been known to influence freshwater biodiversity (Liermann et al., 2012; Poff et al., 2007; Santucci et al., 2005).

2.4 Life Cycle Assessment (LCA)

LCA is a well-established and accepted methodology that builds on natural science to evaluate the potential environmental impacts of a product, process, or service throughout its entire life cycle, from resource extraction to end-of-life (Jolliet et al., 2010). It is often used to identify environmentally preferrable alternatives (*e.g.*, power coming from coal, oil, nuclear or hydro), to pinpoint where are the most important potential environmental impacts and trade-offs in a product, process or service's life cycle, and for eco-design purposes (Hellweg & Milà i Canals, 2014). It has the advantage over other methods, like meta-analyses, environmental impact assessments and

risk analyses, to act on multiple levels of spatial scales (*i.e.*, local, regional and global) and life cycle scales (*e.g.*, for hydropower, impacts from construction, operation and decommissioning, when possible; Jolliet et al., 2010).

2.4.1 LCA phases

LCA is defined by the International Standardization Organization (ISO) standards 14040 and 14044. Within the ISO guidelines (ISO, 2006b), LCA is divided into four phases: 1) goal and scope, 2) Life Cycle Inventory (LCI), 3) Life Cycle Impact Assessment (LCIA) and 4) interpretation. The first phase initially defines the goal of the study: intended application, reasons for the study, target audience, and future use of the results. Then, it defines the scope of the study: product systems studied, system function, functional unit, system frontiers, and many other parameters regarding hypotheses, data requirements and limitations. The second phase is when the inputs (*e.g.*, raw materials) and outputs (*e.g.*, pollutant emissions to soil, air and water), also called the inventory flows of a product, process or service are qualified, quantified, and compiled, throughout its entire life cycle (Jolliet et al., 2010). Third, the inventory flows are translated into potential environmental impacts through Characterization Factors (CF), this is the LCIA phase. CFs translate inventory flows into impact indicators relative to impact categories (Curran et al., 2011; Pennington et al., 2004). These impacts categories are then assigned to Areas of Protection (AoP) that is, human health, ecosystem quality and, natural resource and ecosystem services (Verones et al., 2017). Fourth, the interpretation phase allows the interpretation of the results, the identification of improvement points regarding a specific product, process or service, the execution of uncertainty and sensitivity analyses, as well as a critical analysis of the previously formulated hypotheses and limits of the results' interpretation within the LCA framework (Jolliet et al., 2010).

2.4.2 Characterization Factors (CF)

CFs result from characterization models, which are a mathematical representation of a cause-effect chain relating specific environmental impact[GT6] to a specific impact indicator. Characterization models are based on natural science modelling, grouped into consistent frameworks and made available to LCA practitioners through LCIA methodologies (*e.g.*, Impact World+). Characterization models are usually calculated as follows:

$$CF_{ij} = \sum FF_{ij} \cdot XF_j \cdot EF_j \cdot SF$$

where, the Fate Factor (FF) models the relationship between an environmental intervention *(i.e.*, emission into a compartment or a resource consumption) and a change in a state variable in the receptor environment *j*, the Exposure Factor (XF) models a change in population or ecosystem exposure to this state variable due to a change in *j*, the Effect Factor (EF) calculates the consequences of a change in exposure on human population or ecosystems, ultimately in damage units by applying a Severity Factor (SF; Bulle et al., 2019). A CF is generally based off this equation and the author, as well as the reader, is usually able to explicitly deconstruct the CF into the FF, XF and EF (Núñez et al., 2016). A good example of a CF is the Global Warming Potential (GWP500) for climate change impact characterization. It is a measure of the relative radiative forcing effect of a given substance compared to CO_2, integrated over a 500 years time horizon (IPCC, 2001). The global warming potential of CO_2 (CF = 1 kg $CO_{2eq}/kg_{emitted}$) is lower than that of CH_4 (CF = 28 kg $CO_{2eq}/kg_{emitted}$).

2.4.3 Biodiversity metrics

In this thesis, we qualify and quantify the potential environmental impacts of hydropower, dams and reservoirs on the ecosystem quality AoP (damages to the intrinsic value of natural ecosystems; Curran et al., 2016; Goedkoop et al., 2009; Teixeira et al., 2016) in terms of biodiversity.

In general, in the scientific community, many biodiversity metrics can be used to quantify environmental impacts,[GT7] namely species richness (α-diversity), Fisher's α, Shannon's entropy (*H*), Mean Species Abundance of original species (MSA), Sorensen's S_s, and β-diversity, which are all metrics that represent different components of the concept of biodiversity. The choice of a biodiversity metric was analyzed by de Baan et al. (2013) and is summarized in the following lines.

Species richness is a measure of the number of species in the habitat. It is simple, widely applied, and has extensive data availability as it only requires the presence/absence of species, not the abundance (it is also not specific to the species level of taxonomic resolution, the number of genera or families can also be used to compute richness, in which cases we refer to genus richness or family richness). However, this metric gives equal weight to all species, whether they are endemic or invasive and regardless of their abundance in the community. It also does not provide information on between-habitat diversity (*i.e.*, β-diversity or species turnover), and is affected by under sampling (*i.e.*, the number of species increasing with sampling effort). Fisher's α is a biodiversity metric that corrects for under sampling. It fits the observed values of species richness

and the total number of species to a theoretical relationship (*i.e.*, "true" species richness as a function of the "true" number of species). Shannon's entropy merges information about richness and abundance (*i.e.*, highest when all species in the habitat are equally abundant). The next two metrics compare species composition before and after an environmental impact (*i.e.*, reference versus impacted). The mean species abundance of original species, MSA, informs about the change of abundance of each originally present species (before impact), whereas the Sorensen's S_s assesses how many originally occurring species are present or absent in the impacted habitat. Contrary to the MSA, the Sorensen's S_s does not document a change in species abundance, only the presence or absence of originally occurring species. β-diversity represents the variation in species composition (*i.e.*, number of species and their respective abundance) between habitats within a spatial gradient of interest (Legendre & Legendre, 2012). It conveys a much more detailed analysis of the changes in biodiversity (*e.g.*, species turnover), but requires information on the abundance, like Shannon's entropy and MSA biodiversity metrics, which is not always available.

In their analysis, de Baan et al. (2013) came to the conclusion that even considering some drawbacks (*i.e.*, dependence on sampling effort, no information on abundance and species turnover), species richness should be targeted as a biodiversity metric in LCIA since it is relatively easy to compute, communicate and more readily available than other metrics (*i.e.*, metrics that require abundance data; de Baan et al., 2013). For these reasons, it is currently recommended to use Potentially Disappeared Fraction (PDF) of species, which observes changes in richness between reference and impacted ecosystems, as a robust common impact indicator in LCIA (Verones et al., 2017).

2.4.4 Freshwater use and its impact assessment in LCA

Despite the vital nature of freshwater for both humans and ecosystems, freshwater use hasn't received a lot of attention in LCA until recently (Koehler, 2008). Tackling the entire span of environmental impacts, including the impacts from freshwater use, is essential to ensure the holistic quality and robustness of the LCA methodology (Bayart et al., 2010). There are many different types of water uses, which have been classified using the Owens (2001) terminology, a terminology that is also especially well summarized in Berger and Finkbeiner (2010). First, freshwater resources can be classified into three main types, which differ mainly in terms regeneration potential: deposits, funds and flows. The character and the renewability of these sources of water, however, are still not clearly demarcated within the LCA framework (Owens, 2001). Fossil groundwater

aquifers are a type of deposit freshwater resources. Their lifetimes are typically beyond human life span, they are sealed-in and thus, considered a non-renewable source of freshwater (*i.e.*, fossil). A freshwater fund is a lake or a groundwater aquifer (non-fossil), while a stream or a river represents a flow (Koehler, 2008; Milà i Canals et al., 2009). Both freshwater funds and flows are considered renewable. For the purpose of this thesis, only funds and flows freshwater resources will be addressed.

Second, freshwater use can either be in-stream or off-stream, as well as degradative or consumptive, which are generic terms that apply to both flows and funds of freshwater resources. In-stream means that the water is used *in situ*, like hydropower production, while off-stream means that the water is used *ex situ*, the way drinking, or irrigation water is used. If the water isn't returned to the waterbody in the same physico-chemical condition, it is qualified as a degradative use (*e.g.*, contaminated water discharge), whereas if the water is not returned to the original water body, or even returned to another watershed, it is qualified as consumptive. This leads to at least, but not exclusively, four possible types of freshwater use:

- In-stream and degradative (*e.g.*, retains the water but increases the temperature)
- Off-stream and degradative (*e.g.*, extracts pristine water and returns wastewater)
- In-stream and consumptive (*e.g.*, additional evaporation from reservoirs)
- Off-stream and consumptive (*e.g.*, additional evaporation of extracted water away from a source, like seen in irrigation systems)

Within the Owens (2001) qualification, hydropower is qualified as an in-stream, non-consumptive, and non-degradative water use, and is considered to only affect water flows, not the quality nor the quantity of water. This classification is debatable considering the fact there is evidence of water consumption through reservoir evaporation (Mekonnen & Hoekstra, 2012) and that in some cases, water quality degradation occurs, such as changes in temperature (*e.g.*, cold-bottom water releases; Olden & Naiman, 2010) and oxygen content (*e.g.*, hypoxic or anoxic releases; Bednarek & Hart, 2005; Hoback & Barnhart, 1996). Moreover, water can also be used off-stream in some cases of hydropower (*i.e.*, pump storage; Bayart et al., 2010).

In 2008, a freshwater use impact category was proposed for the first time by Koehler (2008). Since then, it has been further developed by Bayart et al. (2010), Boulay et al. (2011), Kounina et al. (2013) and Boulay et al. (2018). It is only since the last decade that the impact of freshwater use received notable attention in LCA, and yet, its assessment is still missing for wide variety of

applications (Pfister et al., 2011). As of 2018, most of the freshwater use CFs were developed with a focus on water consumption or availability (or scarcity), whereas the impacts of changes in hydrological processes and flow dynamic were mostly overlooked (Karimpour et al., 2021; Núñez et al., 2016). Indeed, the majority of the work that has been conducted so far concerns freshwater consumption. Pfister et al. (2009), studied the impact of agricultural freshwater consumption on human health, ecosystem quality, and natural resources and ecosystem services AoPs through a worldwide cotton production case study. In (2011), Hanafiah et al. tackled the impacts of GHG emissions and freshwater consumption on fish species diversity. Van Zelm et al. (2011) assessed the impacts of ground water extraction for drinking, cooling and irrigation purposes on terrestrial plant species diversity. In (2012), Verones et al. studied the impacts of freshwater use on wetland plants and then, animal diversity in (2013a) and (2013b), whereas Tendall et al. (2014) quantified the impact of freshwater consumption (irrigation) on macroinvertebrates biodiversity. Some work has also been done on freshwater degradation. Verones et al. (2010) characterized the impacts of thermal pollution on freshwater species diversity, and Amores et al. (2013) generated CFs assessing the effects of salinity increases on aquatic algae, crustaceans, fish and plants[GT8].

2.4.5 Hydropower impact assessment

In the last years, hydropower has mostly been addressed in LCA with regards to climate change and GHG emissions (*e.g.*, Gagnon & van de Vate, 1997; Raadal et al., 2011; Varun et al., 2009). Studies like that of Pacca and Horvath (2002), where energy alternatives are ranked based on their LCA scores, often report lower potential environmental impacts for hydropower than for fossil fuels, nuclear and other renewable energies (*i.e.*, wind and solar). One study that stands out in our geographical context is the LCA of Quebec's kilowatt-hour (kWh; CIRAIG, 2014). In this study, the CIRAIG assessed and compared different sources of power production and different energy mixes. What resulted from this study, is that hydropower, because of its low resource usage at the production phase, had the best results based on the studied impact categories (*i.e.*, acidification, eutrophication, climate change, ozone formation, photochemical oxidation [smog], and human toxicity). These results were also similar to what was found in in the literature (Hertwich, 2013; Sathaye et al., 2011). However, in these studies, the impacts related to hydropower freshwater use (*i.e.*, in-stream or off-stream consumptive and/or degradative) and to reservoir occupation on freshwater biodiversity have not been considered, which is as good as accepting that these impacts are null.

2.4.5.1 Studies assessing the impacts of hydropower on freshwater biodiversity

To our knowledge, two studies have recently assessed the impacts of hydropower, dams and reservoir on freshwater biodiversity via a freshwater use standpoint; Dorber et al. (2019) and Turgeon et al. (2021), with the exception of an unpublished study from Humbert & Maendly in (2008). In (2019), Dorber et al. developed spatially-explicit CFs for the impact of freshwater use (through hydropower) on fish biodiversity, using region-specific SDRs in Norway. It is quite venturous to use SDRs to relate change in species richness to change in water discharge due to water consumption as these relationships (*i.e.*, SDRs and SARs) rely on theoretical species richness curves that are based on ecosystems that are at the equilibrium state (Rosenberg et al., 2000; Xenopoulos & Lodge, 2006). Thus, they are not categorically representative of the reality of a hydropower impacted ecosystem. To palliate to this drawback, Turgeon et al. (2021) used change in fish species richness, from empirical data, to assess the impact of hydropower on freshwater biodiversity.

2.4.5.2 Reservoir occupation

Prior studies (Dorber et al., 2019; Humbert & Maendly, 2008) evaluated the impacts of hydropower in terms of freshwater use (*e.g.*, water turbined; in m^3). Turgeon et al. (2021) attempted to move away from this characterization approach since it was criticized by Nunez et al. (2016). Indeed, directly linking the amount of freshwater use (or turbined) to the potential environmental impacts on freshwater biodiversity means that the impacts of hydropower, dams and reservoirs are solely proportional to the amount of water turbined. The impacts of hydropower are not only related to freshwater use. Indeed, when assessing hydropower impact from the standpoint of freshwater use, the impact of dam construction, the creation of a reservoir, the alteration of the downstream river flow, mitigation and stream regulation practices are all omitted. However, this does not necessarily mean that some of these impacts are not characterized in other impact categories (*e.g.*, the impact of concrete used in dam construction on the climate changes).

Although it is quite intuitive that the impacts of hydropower be assessed in terms of freshwater use, it is probably more representative and integrative of the whole suite of impacts to assess them in terms of Land-use and Land-use Change (LULUC). In this case, the impacts from hydropower are neither assessed in terms of consumption or degradation but rather in terms of transformation and occupation. In a nutshell, when land is transformed for a different use (*e.g.*, natural prairie to arable land) its species composition is changed and most likely reduced (Milà i Canals et al., 2007).

Following transformation, land can be occupied (*i.e.*, land occupation) and the species composition is prevented from returning to its original state, thus extending the impacts of land use in time (Lindeijer et al., 2002; Milà i Canals et al., 2007). Turgeon et al. (2021) used this framework to assess the impact of hydropower, dams and reservoirs. Instead of assessing the impacts through freshwater use (*e.g.*, the volume of water turbined), Turgeon et al. (2021) adapted the framework used for land transformation and occupation, which was presented by de Baan et al. (2013) and Chaudhary et al. (2015), and initially proposed by Milà i Canals et al. (2007) and Koellner et al. (2013), to assess the impacts of the transformation of a river, and the subsequent occupation of the riverbed by a reservoir for hydropower, on fish species richness. They created the first empirical-based CFs that translated the impacts of hydropower, through reservoir occupation, on fish, within the LCA framework. Their results also showed that the impacts of reservoir occupation on fish richness was not statistically significant in boreal reservoirs, marginally significant in temperate reservoirs and statistically significant in tropical reservoirs, which further highlighted the need to develop spatially differentiated (regionalized) CFs.

2.4.5.3 Taxonomic coverage

As previously mentioned, impacts to the ecosystem quality AoP are often measured using biodiversity impact indicators (*i.e.*, CFs), which can be calculated at the genetic, species, community and ecosystems levels (Curran et al., 2011; Núñez et al., 2016). Because organisms are not all impacted in similar ways, impact indicators that include more than one group of organisms (the higher the better) are considered more faithful as they can portrait a more realistic biodiversity loss (Núñez et al., 2016). For example, a CF covering fish, macroinvertebrates and aquatic plants, would be considered more robust and representative of the biodiversity observed in an impacted ecosystem than a CF based solely on fish. As reference, a CF for the eco-toxicity impact category requires at least three groups of organisms to ensure a good representation of the variability of physiological and biological responses to environmental impacts (Henderson et al., 2011). However, biodiversity datasets including more than one group of organisms are often unavailable. This explains why most CFs are typically very limited in their biological scope (Núñez et al., 2016). As of now, the two studies assessing the impact of hydropower in LCA (Dorber et al., 2019; Turgeon et al., 2021) only cover fish. The validity of their model may only be good for this specific group of organisms and the outcomes cannot be generalized to the entire freshwater ecosystem.

This presses the need for the generation of multi-group (of organisms), more robust and representative CFs.

2.5 Macroinvertebrates as research organisms

Many different organisms (*e.g.*, fish, macroinvertebrates, zooplankton, phytoplankton, and macrophytes) can be used to assess environmental and human impacts, each having specific advantages and disadvantages. In the literature, macroinvertebrates are often used to assess environmental impacts on freshwater ecosystems, especially those of hydropower, dams and reservoirs (*e.g.*, Aroviita & Hämäläinen, 2008; Baumgärtner et al., 2008; Furey et al., 2006; McEwen & Butler, 2010; Scheifhacken et al., 2007; White et al., 2011).

2.5.1 Robust bioindicators

In this research, we use macroinvertebrates as research organisms because they are considered robust bioindicators due to their relatively long life cycle, their low to no motility, their abundance and their ubiquity (Rosenberg & Resh, 1993). They are also known to play a key role regarding energy transfers (Scheifhacken et al., 2007). Along with being food resources for aquatic and terrestrial consumers (*i.e.*, fishes, amphibians, and birds), macroinvertebrates hold critical roles within the ecosystems. They contribute to accelerating detrital decomposition (van de Bund et al., 1994; Wallace & Webster, 1996) and releasing nutrients into the water during their feeding activities, excretion, and burrowing behaviors in sediments (Covich et al., 1999), which provide essential ecosystem processes and services. Released dissolved nutrients are then absorbed by bacteria, fungi, algae, and aquatic plants, which quickens microbial and plant growth (Covich et al., 1999). Many macroinvertebrates are predators that control the size of their prey, their abundances and their distribution (Crowl & Covich, 1990). Moreover, they act as an important link between terrestrial and aquatic ecosystems, and contribute to nutrient transfers between these ecosystems (Covich et al., 1999). Although they required a substantial amount of time and effort regarding identification, counting, and general laboratory work, macroinvertebrates are relatively easy and inexpensive to sample. Because macroinvertebrate communities are essential components of freshwater ecosystems, and have high taxonomic richness and diverse ecological functions, they are considered excellent bioindicators (Rosenberg & Resh, 1993).

CHAPTER 3 PROCESS FOR THE RESEARCH PROJECT AS A WHOLE, GENERAL ORGANIZATION OF THE DOCUMENT AND COHERENCE BETWEEN THE RESEARCH ARTICLES AND THE SPECIFIC OBJECTIVES

3.1 Problem

Following a thorough review of the literature, few issues were identified and gave rise to this doctoral research. As highlighted in the previous chapter, freshwater ecosystems are vital for humans and support a rich, highly endemic biota that is sensitive to human impacts (Strayer & Dudgeon, 2010). Biodiversity, which is essential to the integrity and function of freshwater ecosystems, is critically threatened by flow alteration and habitat degradation (Dudgeon et al., 2006), two threats that are commonly associated with hydropower generation (and reservoir occupation; Gracey & Verones, 2016). The scientific literature regarding the impacts of hydropower, dams and reservoirs on freshwater biodiversity is quite substantial. A myriad of results is reported, where the direction of observed impacts (positive or negative) and their magnitudes are dependent on variables such as taxonomic group, geographical location and type of hydropower operation, amongst others. In this thesis, we are interested in understanding the impacts of hydropower, dams and reservoirs on macroinvertebrate biodiversity. In the scientific literature, some studies have shown statistically significant hydropower impacts that have negative repercussions on macroinvertebrate biodiversity. Other studies have shown statistically significant hydropower impacts that have positive repercussions on macroinvertebrate biodiversity. Whereas the remaining studies have shown that hydropower does not have statistically significant impacts on macroinvertebrate biodiversity. Thus, the first issue targeted in this thesis is:

1. Studies investigating the impacts of hydropower, dams and reservoirs on macroinvertebrate biodiversity *(i.e.*, richness) have shown a wide range of outcomes and there is still no clear general consensus about these impacts in the scientific community.

In LCA, impacts of hydropower, dams and reservoirs on freshwater biodiversity have rarely been assessed. To our knowledge, only two published studies, exclusive to fish biodiversity, assessed the impacts of hydropower water use and reservoir occupation (Dorber et al., 2019; Turgeon et al., 2021). This brings about the second issue targeted in this thesis:

2. There are no peer-reviewed CFs for the impacts of hydropower, through reservoir occupation, on taxonomic groups other than fish. This is preventing the potential development of multi-group (of organisms) CFs in LCA, a major obstacle in portraying realistic impacts to the biodiversity of freshwater ecosystems.

Most CFs are computed in terms of PDF of species (taxonomic resolution). However, in large geographical endeavors, it is sometimes difficult to collect biological data resolved to this specific level of taxonomic resolution, especially with regards to macroinvertebrates. Macroinvertebrate data is most often only available at the genus or family level of taxonomic resolutions. Thus, the last issue raised in this thesis is more methodological:

3. CFs in PDF units are calculated with richness data resolved at the species level of taxonomic resolution, but there are currently no studies evaluating the impact of taxonomic resolution (genus and family) on CFs.

The combination of these issues led to the genesis of this doctoral research and the identification of a main research objectives, as well as three specific objectives.

3.2 Objectives

The main objective of this research project is to assess the potential environmental impacts of reservoir occupation (*i.e.*, a river that was transformed and further occupied by a reservoir) on macroinvertebrate richness within the LCA framework, complementing fish-based CFs, and addressing spatial variability (*i.e.*, regionalization) within impact characterization. Three specific objectives have been identified:

1. To evaluate the global impacts of hydropower, dams and reservoirs on macroinvertebrate richness using a literature review and a meta-analysis.
2. To develop regionalized CFs and a model based on empirical data to quantify the impacts of the occupation of a riverbed by a reservoir (*i.e.*, reservoir occupation; regardless of its purpose) on macroinvertebrate richness in the United States.
3. To assess the influence of taxonomic resolution on empirically developed CFs and models.

3.3 General methodology

This section gives a general overview of the methodology applied to conduct a literature review and meta-analysis, build CFs and empirical models, and investigate the influence of taxonomic resolution on CFs. The research project is structured into three steps. To facilitate the visualization of the tasks associated to the different steps, a Work Breakdown Structure (WBS) diagram is presented in Figure 3.1 and detailed in the sections below. The three objectives and the corresponding results are developed in chapters four, five and six, respectively. Results, general outcomes, novelty/originality, limitations, conclusions and potential future research are discussed in chapter seven and eight.

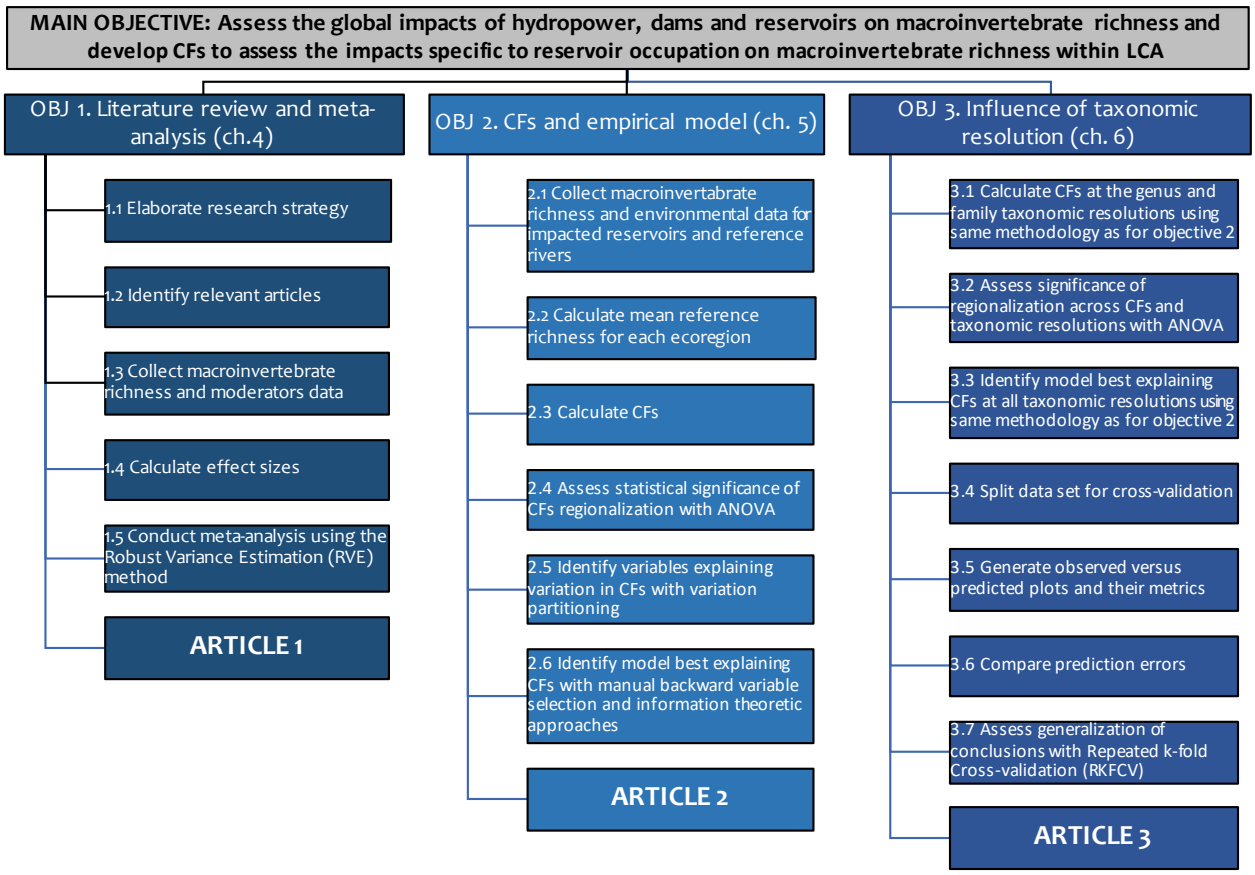

Figure 3.1 Work Breakdown Structure (WBS) diagram.

3.3.1 Objective 1: Literature review and meta-analysis on the global impacts of hydropower, dams and reservoirs on macroinvertebrate richness

The first article, embedded in chapter four, aim to study the consequences of hydropower, dams and reservoirs on macroinvertebrate richness across the world, while simultaneously controlling for the potential influence of moderators (*i.e.*, biomes, impact types, study designs, sampling seasons and gears). We used a meta-analytic framework to tackle this objective, a method that expresses results from multiple studies on a common comparison scale. Findings can then be integrated, generalizations can be made and conflicts may be resolved (Koricheva et al., 2013). This article is dissected into four major phases: 1) research strategy definition, 2) data collection, 3) effect sizes calculation, and 4) statistical analyses.

First, we define a research strategy for conducting a review of the literature in the Web of Science (WoS) databases. Second, from the studies collected, we collect data about richness (reference and impacted), experimental design and moderator variables. Third, we calculate effect sizes specific to studies. Fourth, we proceed to the analysis of a publication bias, to make sure that the range of possible outcomes in the studies collected is not biased to begin with, and heterogeneity sources. We then compute a mean effect size using both a random and mixed-effect meta-analytic model, which allows the assessment of multiple sources of heterogeneity through moderators, and a Robust Variance Estimation (RVE) approach, which allows to account for spatial dependency in effect sizes. The combination of these analyses will allow us to draw conclusions about the overall impact of hydropower, dams and reservoirs on macroinvertebrate richness across the globe and thus, hopefully help reach a clear consensus.

3.3.2 Objective 2: Regionalized CFs and empirical model assessing the impacts of reservoir occupation on macroinvertebrate richness at three spatial scales

The second article, embedded in chapter five, aims to develop empirical CFs assessing the impacts of reservoir occupation on macroinvertebrate richness in the United States, to evaluate the influence of regionalization across ecoregions CFs and to build a simple empirical model for LCA practitioners that estimates CFs without readily available macroinvertebrate richness data. Using a space-for-time substitution approach, which substitute spatial data for unavailable temporal data

(reference versus impacted sites instead of before-after assessment; Banet & Trexler, 2013; Pickett, 1989) and datasets from the United States Environmental Protection Agency (USEPA), we will compute CFs specific to reservoir, ecoregion and country spatial scales. This article methodology can be dissected into four major phases: 1) data collection, 2) CFs calculation, 3) data analysis and 4) empirical modelling.

First, we collect macroinvertebrate richness and a suite of reservoir related environmental variables. Second, we choose to assess CFs in terms of PDF and calculate PDF as the difference between river richness and reservoir richness, divided by river richness, for each reservoir, each ecoregion and for the whole United States. Third, we evaluate if there is a statistically significant effect of regionalization, that is whether ecoregion specific CFs significantly differ from each other across the United States. A significant regionalization would advocate towards the use of regionalized CFs rather than a general country specific CF. Fourth, we build an empirical model to allow LCA modellers and practitioners to estimate a CF (given a quantified amount of error) with only a few easily accessible, reservoirs related, key variables, for reservoirs where macroinvertebrate richness data is not available.

3.3.3 Objective 3: Influence of taxonomic resolution on CFs assessing the impacts of reservoir occupation on macroinvertebrate richness

The third article, embedded in chapter six, aims to study the influence of taxonomic resolution on the direction (positive [loss of taxa] or negative [gain of taxa]) and magnitude of CFs, the statistical significance of regionalization, variables explaining variation observed in CF models and their prediction errors. This methodological companion article is considered as a complement to the second article (presented in chapter 5) and they share a substantial fraction of the methodology. We calculate each CF using different levels of taxonomic resolutions, building on the same first four phases approach developed in the second article. The last phase, 5) validation and generalization, is specific to this article.

We use the same datasets (USEPA) to collect macroinvertebrate richness, using genus and family taxonomic resolutions, as well as environmental variables. Second, we calculate PDFs using the same steps as in the second article but repeat the steps for the different taxonomic resolutions. We use the same methodology to assess the statistical significance of regionalization across CFs, within and across taxonomic resolutions, and to build empirical models specific to each taxonomic

resolution. Because we repeat the model building procedure at each taxonomic resolution, we can identify potential differences due to the influence of taxonomic resolution and assess whether a specific taxonomic resolution is to be privileged. Building on the model with highest support selected for each of the taxonomic resolution, we carry out a cross-validation analysis. This analysis consists in splitting the initial dataset into two unequal parts, the biggest part of the dataset is used for model building purposes *(i.e.*, equation), whereas the smallest part of the dataset is used for validation purpose (*i.e.*, observed versus predicted plots). Following this analysis, we obtain prediction error metrics to compare between taxonomic resolutions and potentially identify a taxonomic resolution model that leads to the smallest prediction error. Finally, we use a Repeated k-fold Cross-validation (RKCFV) method to assess if the outcomes observed with the chosen split of data would be similar to other splits and thus, allow us to make generalizations that go beyond the actual split of data.

CHAPTER 4 ARTICLE 1: THE IMPACTS OF HYDROPOWER ON FRESHWATER MACROINVERTEBRATE RICHNESS: A GLOBAL META-ANALYSIS

4.1 Introduction to Article 1

The article presented in this chapter is a global meta-analysis of the impacts of hydropower, dams and reservoirs on macroinvertebrate richness. It aims to provide conclusions about the direction and the magnitude of these impacts where there is still no clear consensus in the scientific literature. It also evaluates how macro-scale moderators (*i.e.*, biome, impact type, study design, sampling season and gear) influence how hydropower impacts macroinvertebrate richness. This work is at the foundation of this doctoral thesis, it looks at hydropower, dams and reservoirs impacts with a global point of view and investigates the influence of macro-scale moderators. The authors of this article are Gabrielle Trottier, Katrine Turgeon, Daniel Boisclair, Cécile Bulle and Manuele Margni. It was submitted to PLoS One on February 3rd, 2021 and, after careful consideration, was unfortunately rejected because the meta-analysis was not sufficiently up to date (systematic literature search used in meta-analysis included articles published up until December 2019). However, we were invited to resubmit an updated version of the article. We plan on updating the article in the near future and resubmitting to PLoS One.

4.2 Manuscript

4.2.1 Abstract

Hydroelectric dams and their reservoirs have been suggested to affect freshwater biodiversity. However, studies investigating the consequences of hydroelectric dams and reservoirs on macroinvertebrate richness have reached opposite conclusions. We carried out a meta-analysis devised to elucidate the effects of hydropower dams and their reservoirs on macroinvertebrate richness while accounting for the potential role played by moderators such as biomes, impact types, study designs, sampling seasons and gears. We used a random and mixed effect model, combined with robust variance estimation, to conduct the meta-analysis on 72 pairs of observations (*i.e.*, impacted versus reference) extracted from 17 studies (more than one observation per study). We

observed a large range of effect sizes, from very negative to very positive impacts of hydropower. However, according to this meta-analysis, hydropower dams and their reservoirs did not have an overall clear, directional and statistically significant effect on macroinvertebrate richness. We tried to account for the large variability in effect sizes using moderators, but none of the moderators included in the meta-analysis had statistically significant effect. This suggests that some other moderators, which were unavailable for the 17 studies included in this meta-analysis, might be important (*e.g.*, temperature, granulometry, wave disturbance and macrophytes) and that macroinvertebrate richness may be driven by local, smaller scale processes.[GT9] As new studies become available, it would be interesting to keep enriching this meta-analysis, as well as collecting local habitat variables, to see if we could finally draw statistically significant conclusions about the impacts of hydropower on macroinvertebrate richness.

4.2.2 Introduction

Freshwater ecosystems are vital resources for humans and support a biota that is rich, sensitive and characterized by a high level of endemicity (Strayer & Dudgeon, 2010). Ecosystems functions and integrity often depend on biodiversity, which can be described by three indices: species richness (*i.e.*, number of species), community assemblage (*i.e.*, proportions of different species or taxonomic groups in the community) and functional diversity (*i.e.*, variability in organisms' traits that can influence ecosystem functioning; Strayer & Dudgeon, 2010). For millennia, humans have used freshwater ecosystems, through water extraction for drinking and irrigation purposes, water regulation for hydropower production, flood control and recreation (Strayer & Dudgeon, 2010) but these usages often come with a cost on freshwater ecosystems biodiversity (Dudgeon et al., 2006; Vörösmarty et al., 2010).

Hydroelectric dams and the creation of reservoirs, at all stages of their life cycle (*i.e.*, from the construction, to operation and decommission of a dam), can affect freshwater biodiversity (Gracey & Verones, 2016). Dams create a physical barrier, which can impair the natural flow of water, sediments and nutrients (Kummu & Varis, 2007; Ward & Stanford, 1995) and limit the movement of organisms (Nilsson et al., 2010). The alteration of the natural hydrological regime can affect freshwater biodiversity through various biological mechanisms (*i.e.*, mortality through desiccation, mismatch timing in life history strategies, lotic to lentic community changes, reduction/extirpation of endemic and specialist species; Bunn & Arthington, 2002; Poff et al., 1997) and through

degraded water quality (anoxic or hypoxic releases [dissolved oxygen], hypolimnetic or epilimnetic releases [temperature], pH, organic carbon, turbidity; Friedl & Wüest, 2002; Gracey & Verones, 2016; Liermann et al., 2012; Poff et al., 2007; Santucci et al., 2005).

Studies investigating the impact of hydropower on macroinvertebrate richness drew contrasting conclusions. Some studies reported that macroinvertebrate richness is negatively impacted by hydropower, through general flow regulation (Behrend et al., 2012; Jackson et al., 2007; Kullasoot et al., 2017; Takao et al., 2008) and water level fluctuation (or drawdown; Aroviita & Hämäläinen, 2008; Englund & Malmqvist, 1996; Kraft, 1988; Malmqvist & Englund, 1996; Valdovinos et al., 2007; White et al., 2011). Others observed higher macroinvertebrate richness downstream of a dam (Floss et al., 2013; Głowacki et al., 2011) or in regulated rivers (as opposed to natural ones; Smokorowski et al., 2011). Finally, Marchetti et al. (2011) found little difference between impacted flows (*i.e.*, dam-induced permanent low flow) and "natural-like" flows (*i.e.*, high flows in winter and spring, low flows in summer and falls). A meta-analysis could allow to elucidate the patterns and interaction that may exist between hydropower, macroinvertebrate richness, and the context in which studies have been conducted.

Many challenges can be encountered when conducting a meta-analysis, and many ecological facets of studied ecosystems can influence the magnitude and significance of human impacts (De Palma et al., 2018), along with study-specific methodological characteristics (*e.g.*, different study design). The influence and the variability brought about by these facets and characteristics can be accounted for through variables, also called moderators in meta-analysis (Viechtbauer, 2010). For instance, the location of each study site can influence the observed effects (De Palma et al., 2018; Gibson et al., 2011; Willig et al., 2003). A latitudinal biodiversity gradient is a good example of the influence of spatial location. Species richness is known to be highest in the tropics and lowest at the poles (Rosenzweig, 1995; Willig et al., 2003). Hydropower can lead to different types of impacts. A study can analyze the impacts of hydropower upstream of a dam, that is in the reservoir, or downstream of a dam. Impacts also varies depending on the type of water management in place, storage reservoir with winter water level drawdown, hydropeaking or typical run-of-river hydrological regime. These variations in studies, along with the location under study, can introduce variability and heterogeneity in the results, which can be accounted for through moderators. The experimental design can also influence how human-induced impact magnitudes are reflected in a study (De Palma et al., 2018). As demonstrated in Christie et al. (2019), different sampling designs

may affect the conclusion of a study. Using simulations, they demonstrated that Before-After (BA), Control-Impact (CI; analogous to space-for-time substitution) and After designs are far less accurate than Randomized Controlled Trials (RCT) and Before-After Control-Impact (BACI) designs. RCT and BACI are much harder to implement in ecology because true randomization can be difficult with larger scale designs and getting data before impacts or human intervention is sometimes impossible. Thus, we must account for the effect of the experimental design on a study outcome, especially in a meta-analysis, where the effects of multiple different studies are combined. Sampling season can also influence the results across studies, as macroinvertebrate communities differ in terms of abundance and diversity throughout the year (*i.e.*, maximum diversity in late summer and autumn versus underrepresented diversity in spring and early/mid-summer; Hill et al., 2016). At a more local scale, the habitat stratum that is sampled is also most likely to influence the results (De Palma et al., 2018), especially when studying macroinvertebrates. These organisms possess characteristics that make them highly adapted to their environment (McCafferty, 1983) and because lakes, reservoirs and river beds are so heterogeneous, macroinvertebrates are often patchily distributed, requiring extensive sampling (Wetzel, 2001). Thus, the type of sampling gear used to sample will likely affect the type of organisms inventoried in each study.

The objective of this manuscript is to conduct a meta-analysis about the impacts of hydropower, dams and their reservoirs on macroinvertebrates richness while accounting for a series of moderators defining the context of the studies included. The moderators included in this manuscript are the following: 1) biomes (*i.e.*, boreal, temperate, and tropical), a proxy for location/latitudinal gradient, 2) types of impact, which is reflected by the position of a sample in relation to the dam (*i.e.*, upstream or downstream of the dam). Downstream stations are impacted by a reduced flow and hydropeaking dynamics, whereas upstream stations are impacted by drawdown and water level fluctuations due to reservoir management, 3) types of study design such as cross-sectional (*i.e.*, reference natural lake versus impacted reservoir) and longitudinal spatial gradient (*i.e.*, upstream of a dam [reservoir] versus downstream of a dam [river]), which are two different variants of CI study design, 4) sampling seasons (*i.e.*, spring, summer, fall, winter) and 5) sampling gears, a proxy for habitat stratum (*i.e.*, grabs and nets).

4.2.3 Methodology

A meta-analytic framework was used to assess the impacts of hydropower on macroinvertebrate richness, for different biomes, impact types, study designs, seasons and sampling gears. A meta-analysis is a method of research synthesis that is based on expressing results from multiple studies that share a given research subject (but different sampling strategies, sample sizes or sampling gears) on a common comparison scale, and aims to integrate findings, establish generalizations or resolve conflicts (Koricheva et al., 2013). Whereas a research synthesis is usually performed qualitatively, a meta-analysis is a powerful, informative and unbiased tool that allows the quantification of generalizations through various statistical methods (Koricheva et al., 2013). It statistically combines the magnitude and the direction of results or outcomes (later referred to as effect sizes) of multiple studies sharing a common research objective (Koricheva et al., 2013). For example, when comparing richness between an impacted and a natural habitat, the magnitude of the difference in richness (*i.e.*, delta) and its direction (*i.e.*, either gain or loss of taxa) constitute the effect size. Meta-analysis has many advantages over other methods (*i.e.*, narrative review, vote counting). Namely, it takes into account sample size and statistical power from each study (with potentially different sampling strategies), assesses the magnitude and statistical significance of the mean effect size, and allows the analysis of multiple sources of variation among studies (*i.e.*, moderators; Koricheva et al., 2013). For those reasons, we chose this method over the others.

Achieving our objective requires to first establish a research strategy, second to do a data collection comprising information regarding macroinvertebrate richness in hydropower impacted habitat versus reference ones, biome, types of impact, study designs, sampling seasons and gears. Third, it requires to compute effect sizes for each study and finally, combine them to assess if the mean effect size is significantly different from zero (Koricheva et al., 2013) and if the presence of other moderators can influence these results.

4.2.3.1 Research strategy

In this study, we used the PRISMA (Preferred Reporting Items for Systematic and Meta-Analyses) methodology, flow diagram (Fig A.1) and checklist (Table A.1) proposed by Moher et al. (2009), to report systematic literature reviews and meta-analyses. A systematic literature review was conducted using the Web of Science Core Collection database, which includes all journals indexed in Science Citation Index Expanded (SCI-EXPANDED), Social Sciences Citation Index (SSCI),

Arts & Humanities Citation Index (A&HCI), Conference Proceedings Citation Index – Science (CPCI-S), Conference Proceedings Citation Index – Social Science & Humanities (CPCI-SSH) and Emerging Sources Citation Index (ESCI; Clarivate Analytics, 2019). The research strategy was constrained between 1989 and 2019 and contained a combination of the four following field of research: 1) hydropower (hydropower OR hydroelectric* OR dam OR dams OR reservoir* OR impound* OR run-of-river OR "run of river" OR drawdown* OR hydropeak* OR dam* OR "water level fluctuation" OR "water-level fluctuation" OR "water level variation" OR "water-level variation" OR "water level regulation" OR "water-level regulation" OR "water level manipulation" OR "water-level manipulation" OR "water management"), 2) biodiversity ("biodiversity" OR "richness"), 3) freshwater ecosystems ("freshwater" OR "aquatic") and 4) aquatic insects ("*invertebrate*" OR "*benth*" OR "insect*" OR "arthropod*"). Research strategy also excluded all studies pertaining to beaver and agricultural dams (NOT "beaver*" NOT "agricult*"). The research strategy resulted in 408 research articles, as per December 16th 2019.

The results were sorted by Web of Science relevance, which ranks records based on the number of search terms found in each record (Clarivate Analytics, 2018), and extracted as a list to further evaluate the relevance of every studies based on a list of criteria. The following criteria were applied to assess the inclusion of any study in the meta-analysis: 1) the study had to refer specifically to hydropower related impacts (*i.e.*, reservoir, run-of-the-river or hydropeaking, multi-purpose reservoirs were also checked for hydropower impacts), 2) the scope of the study had to address freshwater ecosystems and macroinvertebrates and 3) the study had to be empirical (*i.e.*, excluding literature reviews, modelling exercises) and provide an explicit richness, error and sample size value for both a reference and impacted site (*i.e.*, cross-sectional studies [reference versus impacted]) or gradient of impact (longitudinal spatial gradient studies [upstream of the dam/reservoir versus one or multiple sites downstream of the dam]). Out of the 408 research studies that resulted from the research strategy, only 17 met the above criteria (see geographical disposition of studies in Fig 4.1).

Figure 4.1 World map showing the geographical disposition of the studies used in this meta-analysis; [1] Aroviita and Hämäläinen (2008), [2] Valdovinos et al. (2007), [3] Marchetti et al. (2011), [4] Molozzi et al. (2013), [5] Takao et al. (2008), [6] Kullasoot et al. (2017), [7] White et al. (2011), [8] Smokorowski et al. (2011), [9] Englund and Malmqvist (1996), [10] Jackson et al. (2007), [11] Kraft (1988), [12] Mellado-Dìaz et al. (2019), [13] Bruno et al. (2019), [14] Milner et al. (2019), [15] Steel et al. (2018), [16] Schneider and Petrin (2017) and [17] Vaikasas et al. (2013).

4.2.3.2 Data collection

Richness metrics and moderators were collected for each of these 17 studies (Tables A.2 and A.3). We extracted richness (*i.e.*, number of taxa), error measure (*e.g.*, standard deviation), sample size (*i.e.*, number of reference observations and impacted observations used to compute richness and its associated error) and a suite of moderators such as biome (*i.e.*, boreal, temperate and tropical; 2001), type of impact (*i.e.*, water level fluctuations upstream, due to reservoir management, or flow regulation downstream due to dam operations), type of study, (*i.e.*, cross-sectional [reference natural lake versus impacted reservoir] or longitudinal spatial gradient [reference; upstream of the dam/reservoir versus impacted; downstream of the dam]), sampling season (*i.e.*, spring [March to May], summer [June to August], fall [September to November] and winter [December to February], according to the hemisphere) and sampling gear (*i.e.*, net, grab or colonization basket). These moderators were chosen based on the availability of said moderators in each of the 17 studies, their potential influence on macroinvertebrate richness and complemented with expert judgement. [GT 10] In studies where meaningful data were presented exclusively in graphical format, values were extracted using Engauge Digitizer 10.4 (Mitchell et al., 2017).

4.2.3.3 Effect size

To compute effect sizes, we calculated the standardized mean differences, also called Cohen's d – which expresses the distance between two means (*i.e.*, impact and reference) in terms of their common standard deviation (Borenstein et al., 2011). For most studies – except Takao et al. (2008) and Schneider and Petrin (2017), we computed at least two effect sizes per study, leading to a total of 72 effect sizes, with a certain level of within-study dependency (*i.e.*, effect sizes in one study are not entirely independent from each other). Cohen's d is calculated as per equation 4.1 (Borenstein et al., 2011):

$$d = \frac{\bar{X}_1 - \bar{X}_2}{\sqrt{\left(\frac{(n_1 - 1)s_1^2 + (n_2 - 1)s_2^2}{n_1 + n_2 - 2}\right)}} \tag{4.1}$$

where $\bar{X}_1$ and $\bar{X}_2$ are the mean richness, s_1^2 and s_2^2 are the standard deviation (SD) and, n_1 and n_2 are the number of observations used to compute the mean and SD for impacted and reference samples, respectively. The common variance (V_d) associated with the effect size (d) is calculated using equation 4.2:

$$V_d = \frac{n_1 + n_2}{n_1 \cdot n_1} + \frac{d^2}{2(n_1 + n_2)} \tag{4.2}$$

In the case of small sample size (< 20 studies), a correction factor is applied to Cohen's d to reduce the positive bias (negligible with bigger sample size) and to provide a better estimate. The corrected effect size is then called a Hedges' g (Hedges, 1981). A small sample correction factor (J) was computed using equation 4.3:

$$J = 1 - \frac{3}{4df - 1} \tag{4.3}$$

where df refers to the degrees of freedom ($n_1 + n_2 - 2$). Thus, the corrected effect size g and variance V_g are calculated following equations 4.4 and 4.5, respectively:

$$g = J \cdot d \tag{4.4}$$

$$V_g = J^2 \cdot V_d \tag{4.5}$$

A positive g means the impacted environment or sample has more richness in comparison to a reference environment or sample. The *metafor* package (Viechtbauer, 2010) was used to compute the effect sizes and sampling variances (*i.e.*, *escalc* function) and the *ggplot2* package (Wickham, 2016) was used to graphically visualize the results of the meta-analysis[GT11]. All statistical analyses were made using R version 3.0.2 (R Core Team, 2017).

4.2.3.4 Data analysis

4.2.3.4.1 Publication bias

Studies with large significant results are more likely to be published than studies with non-significant results, this is called publication bias (Koricheva et al., 2013). A funnel plot was used to evaluate the presence of a publication bias in the meta-analysis (Sterne & Egger, 2001), and a regression test for funnel plot was used to detect potential asymmetry (*i.e.*, if only large significant studies, range of outcomes is not well represented and studies with non-significant results might not even be present; Sterne & Egger, 2005). The *metafor* package (Viechtbauer, 2010) was used for asymmetry analysis (*i.e.*, *funnel* and *regtest* functions).

4.2.3.4.2 Heterogeneity

Specific study characteristics (*e.g.*, biome) and dissimilarities in methodologies among studies (*e.g.*, study design) can introduce variability, or heterogeneity (not only due to within study sampling error), among true effect sizes (Huedo-Medina et al., 2006; Viechtbauer, 2010). It is interesting to examine this heterogeneity and identify the different moderators and their relative contributions to the magnitude and direction of these effect sizes (Koricheva et al., 2013). We evaluated the statistical significance and the magnitude of the heterogeneity of effect sizes among studies (*i.e.*, heterogeneity analysis) using, respectively, the Q statistic and I^2 index (Huedo-Medina et al., 2006). The *metafor* package (Viechtbauer, 2010) was used for heterogeneity analysis (*i.e.*, *rma* function).

4.2.3.4.3 Random and mixed-effect model: dealing with heterogeneity

Two types of meta-analytic models can be used in a meta-analysis, fixed or random. A fixed model considers only the studies included in the meta-analysis and within study sampling variability, not between studies (Koricheva et al., 2013). No inference can be made outside this set of studies (*i.e.*, conditional inferences; Viechtbauer, 2010). A random-effect model considers the set of analyzed studies as a sample of a larger population of studies (Viechtbauer, 2010). Thus, it allows the researcher to make inferences regarding what would be found if an entire new meta-analysis, with a different set of studies, was performed (*i.e.*, unconditional inferences; Viechtbauer, 2010). It also allows two sources of variation, within and among (Koricheva et al., 2013). Such an approach is especially appropriate when dealing with heterogeneity among studies (Huedo-Medina et al., 2006)

and with a mixed-effect model approach, it is also possible to include moderators, which can account for some of that heterogeneity (Viechtbauer, 2010). Here, a random and mixed effect modelling is preferred to a fixed modelling approach since 1) a significant amount of heterogeneity was found in the previous heterogeneity analysis and 2) because it offers the possibility to model and explain some of that heterogeneity using moderators (Borenstein et al., 2011).

4.2.3.4.4 Robust variance estimation: dealing with dependency

If our effect sizes were all independent from each other, we could have simply used a random and mixed effects model. However, because this meta-analysis is dealing with multiple effect sizes per study, where observations are not methodologically and spatially independent from each other, it is inappropriate to use a regular meta-analysis approach (*i.e.*, random and mixed effects model), where the effect sizes are assumed to be independent. One way to account for dependency of effect sizes is to combine the random and mixed effect model with the robust variance estimation (RVE) method. The RVE estimates the overall effect size over studies using a weighted mean of the observed effect sizes (Moeyaert et al., 2017). It doesn't require knowledge about the within-study covariance, it can be applied to any type of dependency and effect sizes, it simultaneously accommodates for multiple sources of dependencies, it does not require the effect sizes to comply to any particular distribution assumptions, it leads to unbiased fixed effects and standard errors estimates and can also give an estimate of among-study variance (Fisher & Tipton, 2017; Moeyaert et al., 2017). Because the most common source of dependence within the effect sizes in this meta-analysis is the correlated nature of the observations (*i.e.*, multiple measures within a study; methodological and spatial correlation) and not the hierarchical nature (*i.e.*, common nesting structure between studies; a sample within a transect, within site and within a lake), a correlated effects weighting method was used (Fisher & Tipton, 2017). Thus, we used a RVE based on a correlated effects model and adjusted for small sample size (< 40 studies; Fisher & Tipton, 2017). Finally, a sensitivity analysis was computed to assess the effect of a varying rho (ρ) value, which is a user-specified value of the within-study effect sizes correlation (*i.e.*, the correlation between two samples taken in the same water body in one specific study – spatial and methodological correlation; Fisher & Tipton, 2017). The *robumeta* package (Fisher & Tipton, 2017) was used to fit the RVE meta-regression model (*i.e.*, *robu* function) and to compute the sensitivity analysis (*i.e.*, *sensitivity* function).

4.2.4 Results

4.2.4.1 Methodological results

The purpose of this first set of results is to validate our methodological approach and choices, they will not be the subject of discussion. No statistical asymmetry was observed in the funnel plot (z = -1.0707; p = 0.2843; Fig A.2), a wide range of results and significance levels were represented by the studies included in this meta-analysis. The total heterogeneity among the effect size was statistically significant (Q_{df} = 557.67; p < 0.0001), which indicated greater total heterogeneity than expected by the sampling error alone. The estimated amount of this heterogeneity among the effect sizes was T^2 = 3.78; 95% confidence interval [CI] = 2.75-6.15. Of that total variability, a large amount (I^2 = 90.13%; CI = 86.91-93.69) was due to true heterogeneity between the studies, rather than just methodological variability. Thus, further examination of the heterogeneity is warranted and will be done through the analysis of multiple moderators.

4.2.4.2 Meta-analysis results

The meta-analysis of 17 studies (72 pairs of observations; Table A.2) suggests that hydropower, dams and reservoirs did not have a statistically significant effect on macroinvertebrate richness. The mean effect size (*i.e.*, Hedge's *g*) estimate of our RVE model was -0.864 (95% CI = -1.87 to 0.144; p = 0.088), without accounting for the different moderators (Fig 4.2). The large confidence interval overlapping zero indicates that the mean effect size is not statistically significant and highlights a wide range of effect sizes across studies. The sensitivity analysis shows that the estimates of the mean effect size and standard errors, as well as the estimate of between-study variance in study-average effect sizes (τ^2), are relatively insensitive to different values of ρ (Table A.4).

Figure 4.2 Forest plot of the meta-analysis, the mean effect size is -0.864 (95% CI = -1.87 to 0.144, shaded grey area), where study type is shape-coded (*i.e.*, circle for longitudinal studies and squares for cross-sectional studies) and biome color coded (*i.e.*, boreal in blue, temperate in yellow and tropical in red). A negative effect size means that there is a negative impact of hydropower in impacted sites as opposed to reference sites, whereas a positive effect size means that there is positive impact of hydropower in impacted sites as opposed to reference sites.

Moderators had very little influence in the effects of hydropower, dams and reservoirs on macroinvertebrate richness. Biome did not significantly explain variability in effects sizes. For this moderator, it was only possible to make statistically valid inferences for the temperate level (estimate = -0.56; 95% CI = -2.60 to 1.49; p = 0.522; Fig 4.3A). The boreal and tropical levels had too little degrees of freedom ($df_s < 4$), which invalidates the Satterthwaithe approximation (*i.e.*, calculation of the effective dfs of a linear combination of independent sample variances; Fisher & Tipton, 2017; Spellman & Whiting, 2013; van Emden, 2008), thus the results of the RVE with this moderator in the equations have to be interpreted with caution. On the contrary, both types of impact and study moderators had enough df_s for robust statistical analysis. Statistically significant

effects were neither found for downstream flow regulation (estimate = -0.63; 95% CI = -1.53 to 0.28; p = 0.158) nor for upstream water level fluctuations/drawdown impact types (estimate = -0.83; 95% CI = -4.16 to 2.49; p = 0.575; Fig 4.3B). When study type was used as a moderator, there was no significant difference in effect sizes for cross-sectional design studies (*i.e.*, natural versus impacted; estimate = 0.49; 95% CI = -1.49 to 2.47; p = 0.581), but there was a marginally significant difference for longitudinal gradient type studies (*i.e.*, spatial gradient; estimate = -1.21; 95% CI = -2.76 to 0.33; p = 0.095; Fig 4.3C). Studies that were considered as gradients were most often associated with negative effect sizes. As for results from the season (Fig 4.3D) and sampling gear (Fig 4.3E) moderators, no strong statistical inferences can be drawn due to insufficient df_s (Satterthwaithe approximation invalidated).

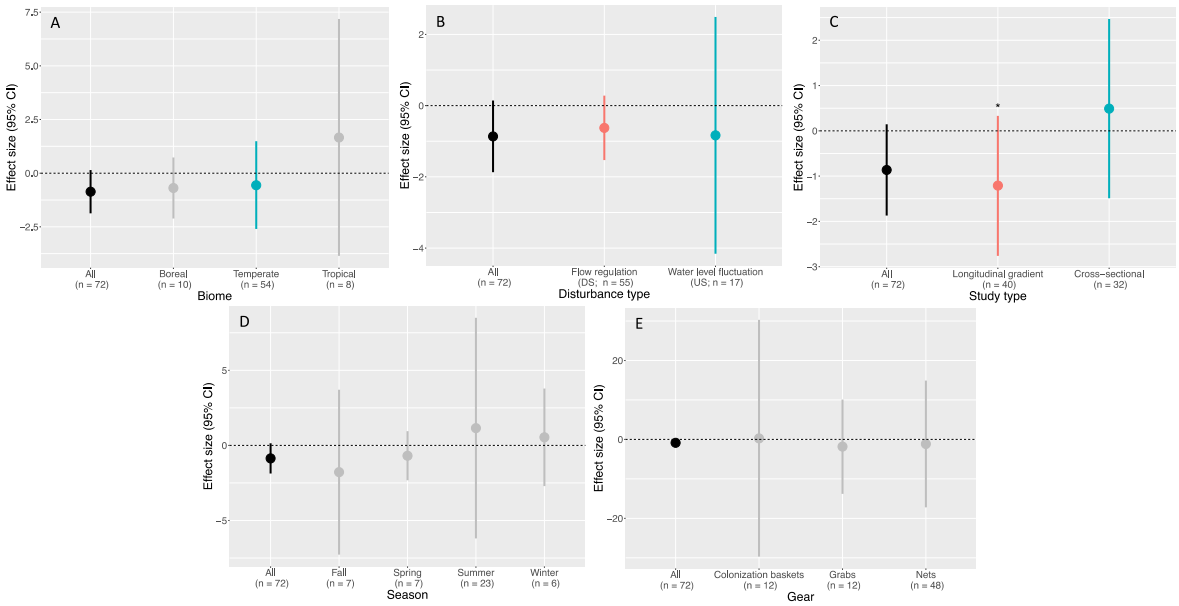

Figure 4.3[GT13] Plots showing the mean effect sizes and their confidence interval for each of the moderators. Value in black is the mean effect size of the meta-analysis and the other colors are related to the different effect sizes when including specific moderators. When in grey, statistical significance of moderator cannot be interpreted with confidence due to statistical power issues (df_s insufficient). When effect size is in color (*i.e.*, blue or red) statistical interpretation can be made with confidence, whether it is significant or not (sufficient df_s). Asterisk signifies statistically marginally significant effect.

4.2.5 Discussion

The meta-analysis conducted on 72 pairs of observations (*i.e.*, impacted versus reference) extracted from 17 studies suggests that hydropower does not statistically impact macroinvertebrate richness in a clear directional way. As highlighted by the literature, there is a divergence in results and it is not obvious as to whether or not hydropower, dams and reservoirs impact macroinvertebrate richness. The lack of statistical significance in the mean effect size seems to result from a large variability in responses, from very negative to very positive impacts, and not from a real absence of impacts on macroinvertebrate richness (Fig 4.2). In fact, more than 86% of the effect sizes were significantly different from zero, from which 60% of the observations showed reduced richness due to hydropower and only 26% of the observations showed reduced richness in reference conditions. These percentages, along with the marginally non-significant mean effect size (*i.e.*, -0.864; 95% CI = -1.87 to 0.144; p = 0.088) could suggest a tendency toward negative impacts of hydropower on macroinvertebrate richness, this is, however, not supported by statistical evidences. We observed a large heterogeneity in macroinvertebrate richness responses to hydropower (Fig 4.2). Part of this variability across observations and studies can usually be explained by environmental and methodological variability, which we can try to control using moderators. In our meta-analysis, we considered biome (*i.e.*, boreal, temperate or tropical), type of impact (*i.e.*, water level fluctuations in the reservoir or flow regulation downstream of the dam), the type of study (*i.e.*, spatial longitudinal gradient [upstream vs downstream] or cross-sectional [natural lake versus impacted reservoir), sampling season (*i.e.*, spring, summer, fall and winter) and sampling gear (*i.e.*, net, grab and colonization basket) as moderators. Despite the documented effects on macroinvertebrate richness of the moderators included, in our analysis, none of the moderators statistically explained heterogeneity in macroinvertebrate richness responses to hydropower, dams and reservoirs.

4.2.5.1 Biome moderator

In general, it is quite common to observe a latitudinal gradient in species richness, where there is a maximum richness in the tropics and a decline towards the poles (Hillebrand, 2004; Rosenzweig, 1995; Willig et al., 2003). In a meta-analysis, Turgeon et al. (2019), showed that the impacts of impoundment on fish richness differed across biomes. Significant declines in richness were observed in the tropics, a lower decline was observed in temperate regions and no impact was

observed in boreal biomes. Thus, we hypothesized that a latitudinal gradient could also influence the mean effect size in macroinvertebrates. Our data did not support a latitudinal trend which is not too surprising because there is no clear pattern about whether or not macroinvertebrate richness follows a latitudinal gradient (Allan & Castillo, 2007; Covich, 1988; Vinson & Hawkins, 2003). Pearson and Boyero (2009) observed a richness peak around the equator for dragonflies (*i.e.*, odonata), but no clear global pattern for caddisflies (*i.e.*, trichoptera), whereas Vinson and Hawkins (2003) observed richness peaks at mid-latitudes in South and North America for mayflies (*i.e.*, ephemeroptera), stoneflies (*i.e.*, plecoptera) and caddisflies (*i.e.*, trichoptera; [EPT] orders). On the contrary, Scott et al. (2011) did not observe this latitudinal gradient for EPT in northern Canada. Geographic range (*i.e.*, narrow range and exclusion of extreme latitudes; Vinson & Hawkins, 2003), sampling effort (Vinson & Hawkins, 1996), macroinvertebrate specific life-history strategies (Pearson & Boyero, 2009; Scott et al., 2011) and data resolution (*i.e.*, weak gradient for local species richness; Heino, 2009) were probably the reason behind this lack of consensus (Shah et al., 2014), and absence of a trend in this meta-analysis. Moreover, the unbalanced sample size across biomes may have impeded any clear conclusions. Boreal samples represented 14% of the data, tropical samples 11% and roughly 75% of the data belong to temperate ecosystems. This prevented us from statistically supporting a trend for both boreal and tropical observations.

4.2.5.2 Impact type moderator

Water level fluctuation in reservoirs is characterized by yearly drawdown (*i.e.*, long-term oscillations in the water level), whereas flow regulation usually is reflected by daily or weekly changes in flow downstream of a dam (*i.e.*, short-term oscillations; Valdovinos et al., 2007), keeping in mind that variation in the reservoir are also reflected downstream. In reservoirs experiencing water level fluctuations in the form of winter drawdown (24% of the reservoirs in this meta-analysis), the littoral zone is exposed to desiccation and freezing for an extended period of time, which has been causing a loss of macroinvertebrate taxa (Aroviita & Hämäläinen, 2008; Palomäki, 1994; Palomäki & Koskenniemi, 1993; Valdovinos et al., 2007) and decreased their overall abundance (Trottier et al., 2019). In the case of daily downstream fluctuations (76% of the reservoirs in this meta-analysis), organisms can burrow in the sediment (*i.e.*, low mobility taxa such as oligochaetes) or follow the water level up and down (*i.e.*, high mobility swimming taxa such as dragonfly nymphs; Valdovinos et al., 2007). Riverine organisms have evolved and developed adaptations to survive flood and drought hydrological dynamics (Humphries &

Baldwin, 2003), but lentic taxa are not adapted to extreme water level fluctuations, such as winter drawdown in reservoirs (> 2m amplitude; White et al., 2011). Because of that, we were expecting a higher impact and effect size of water level fluctuation in reservoirs (*i.e.*, winter drawdown) on macroinvertebrate richness than downstream flow regulation but found no differences nor significant effect of impact type moderator.

4.2.5.3 Study type moderator

Another moderator that could explain heterogeneity in our results was the type of study. An ideal way to analyze the impact of hydropower on richness is to set the reference conditions as richness before impoundment versus impacted conditions as richness after impoundment, within the same ecosystem (*e.g.*, a reference river that was impounded into an impacted reservoir; longitudinal in time, also known as BACI; Christie et al., 2019). However, studies using this type of methodology are rare, even more so for macroinvertebrates, and such studies were mostly absent in the results from the literature review and if present, did not fulfill the required criteria to be included in the meta-analysis (none could be included in this case). Here, we accounted for two types of study methodology; a cross-sectional methodology (*i.e.*, comparing a hydropower impacted ecosystem with a natural/reference ecosystem; $n = 44\%$) and longitudinal gradient in space methodology (*i.e.*, upstream of a dam versus downstream of that same dam, within a single ecosystem; $n = 56\%$), which are both considered as simple CI studies (Christie et al., 2019). These methodologies are not ideal, compared to the longitudinal in time methodology (*i.e.*, BACI or BA studies), as their reference spatial point differs from the impacted spatial point, thus introducing some environmental noise and there is no way to control for environmental stochasticity (Christie et al., 2019). In the cross-sectional type, studies included inter-ecosystem's variation (*i.e.*, impacted ecosystem was spatially independent from reference/non-impacted ecosystem). This inherently added some heterogeneity in the responses and the impacts could be more difficult to detect. In the longitudinal in space studies (reference upstream and impacted downstream of the dam, at a single point in time), the observations were from the same ecosystem. Here, there is less heterogeneity and thus, we could have expected the results to be less variable. However, there was no significant difference between the two study types in this meta-analysis. Even though the patterns were not significant, we observed a negative trend in the spatially longitudinal gradient studies (*i.e.*, higher richness in reference ecosystems, that is upstream of the dam), with a slightly tighter range of variation. The cross-sectional studies had a tendency toward higher richness in reservoirs, with a larger range of

variation. This might highlight a problem with general study design and the choice of the reference ecosystem, and caution is in order when interpreting these trends. Nevertheless, we believe that cross-sectional and longitudinal in space references are the best benchmark available to overcome current limitations regarding the lack of richness data before impoundment in the literature.

The analysis of moderators did not allow a better understanding of the large heterogeneity in the effect sizes and suggests that maybe other moderators, which were not available for the studies included in this meta-analysis, could help tease out some of that heterogeneity. For instance, given that macroinvertebrates are very adapted to their localized environmental conditions (*e.g.*, temperature, granulometry, wave disturbance and macrophytes; McCafferty, 1983), their richness maybe regulated at a much finer scale. Thus, these micro-habitat moderators could be especially relevant to include in a future meta-analysis, although very hard to collect in such a global consolidating endeavour.

4.2.6 Conclusion

Overall, our meta-analysis did not draw any clear, directional, statistically significant conclusions regarding whether or not hydropower, dams and reservoirs impact macroinvertebrate richness. However, the large variability observed across studies (significantly negative to significantly positive results), coupled with the marginality of the statistical significance, suggest that macroinvertebrates could be impacted by hydropower, keeping in mind that this suggestion is not statistically supported. The environmental and methodological heterogeneity in the studies might have hindered the detection of a significant effect but none of our moderators helped untangle that heterogeneity. This advises that other moderators, which have not been included in this study due to unavailability among the studies, may be responsible for some of that heterogeneity. We advocate that local, smaller scale variables pertaining to habitat physicochemical characteristics may bring some clarity about the large heterogeneity in effect sizes. As new studies evaluating the impacts of hydropower on macroinvertebrate richness accumulate, we would advise that information regarding local habitat variables be available so they could be recorded and evaluated as moderators in future meta-analyses. This meta-analysis wasn't able to highlight a clear directional effect of hydropower on macroinvertebrate richness and it would be even more interesting to keep enriching it, as new studies are available, to see if the results could change in a few years and become more assertive. If so, we could finally come to terms with the divergence of

results regarding the impacts of hydropower, dams and reservoirs on macroinvertebrate richness and draw clear, statistically supported conclusions.

4.2.7 Acknowledgments

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CHAPTER 5 ARTICLE 2: EMPIRICAL CHARACTERIZATION FACTORS FOR LIFE CYCLE ASSESSMENT OF THE IMPACTS OF RESERVOIR OCCUPATION ON MACROINVERTEBRATE RICHNESS ACROSS THE UNITED STATES

5.1 Introduction to Article 2

The article presented in this chapter highlights the importance of regionalization, provides LCA modellers and practitioners with multi-scale CFs assessing the impacts of reservoir occupation on macroinvertebrate richness in the United States, as well as a simple empirical model to estimate reservoir specific CFs when macroinvertebrate richness data are not available. This work is a follow-up to article 1. It looks at a specific type of hydropower impacts, namely reservoir occupation, at smaller spatial scales (*i.e.*, country, ecoregion and reservoir, instead of global) and also investigates the influence of micro-scale reservoir related environmental variables. A substantial part of this work was conducted during a half-year research stay at the Norwegian University of Science and Technology (NTNU) with the collaboration of Francesca Verones. The authors of this article are Gabrielle Trottier, Katrine Turgeon, Francesca Verones, Daniel Boisclair, Cécile Bulle and Manuele Margni. It was submitted to Sustainability on February 4th, 2021 and accepted for publication on March 1st, 2021 (https://doi.org/10.3390/su13052701). This work was also presented, in the form of a poster, at the Society of Environmental Toxicology and Chemistry (SETAC) Europe 29th Annual Meeting, which was held on May 26th to 30th 2019 in Helsinki, Finland.

5.2 Manuscript

5.2.1 Abstract

The transformation of a river into a reservoir and the subsequent occupation of the riverbed by a reservoir can impact freshwater ecosystems and their biodiversity. We used the National Lake Assessment (134 reservoirs) and the National Rivers and Streams Assessment (2062 rivers and streams) of the United States Environmental Protection Agency (USEPA), to develop empirical Characterization Factors (CFs; in Potentially Disappeared Fraction [PDF] of species) evaluating

the impacts of reservoir occupation on macroinvertebrate richness (number of taxa) at the reservoir, ecoregion and country spatial scales, using a space-for-time substitution. We used analyses of variance, variation partitioning, and multiple regression analysis to explain the role of ecoregion (or regionalization; accounting for spatial variability) and other potentially influential variables (physical, chemical and human), on PDFs. At the United States scale, 28% of macroinvertebrate taxa disappeared during reservoir occupation and PDFs followed a longitudinal gradient across ecoregions, where PDFs were higher in the west. We also observed that high elevation, oligotrophic and large reservoirs had high PDF. This study provides the first empirical macroinvertebrate-based PDFs for reservoir occupation to be used as CFs by LCA practitioners. The results provide strong support for regionalization and a simple empirical model for LCA modelers.

5.2.2 Introduction

Water abstraction (withdrawal), regulation of water flow by dams (storage reservoirs for drinking water, flood control, and energy production), or water diversion by channels (irrigation and navigation) have benefited human populations worldwide (Chen et al., 2016; Lehner et al., 2011). However, despite clear societal benefits, the use of water is often accompanied by a myriad of environmental impacts (Abell et al., 2008; Gracey & Verones, 2016; Renöfalt et al., 2010; Rosenberg et al., 2000; Vörösmarty et al., 2005).

The environmental impacts brought about by dams are well documented (Agostinho et al., 2008). Geomorphology, water depth and hydrological regime are notably altered. Changes in water depth, temperature and total dissolved solids affect ecosystem productivity (Jackson et al., 1990; Rempel & Colby, 1991; Ryder, 1982; Youngs & Heimbuch, 1982). A change in the hydrological regime (lotic into a lentic ecosystem, upstream of the dam) affects several physical and biological processes, as well as organisms' capacities to thrive and survive in these ecosystems. These changes can ultimately impact ecosystem biodiversity, productivity and the provision of ecosystem services (Bunn & Arthington, 2002; Dudgeon et al., 2006; Poff et al., 1997). To sustain these services and preserve the ecological integrity of our freshwater ecosystems, we must understand the impacts of dams and reservoirs on freshwater biodiversity. In this article, we are interested in macroinvertebrate richness (number of taxa).

Few studies have investigated the impacts of dams and reservoirs on macroinvertebrate richness. Flow regulation and water level fluctuations (also known as drawdown) (Aroviita & Hämäläinen,

2008; Englund & Malmqvist, 1996; Kraft, 1988; Malmqvist & Englund, 1996; Valdovinos et al., 2007; White et al., 2011) can negatively impact macroinvertebrate richness (Behrend et al., 2012; Jackson et al., 2007; Takao et al., 2008). However, Glowacki et al. (2011) and Floss et al. (2013) observed higher richness downstream of a dam, or in regulated rivers, as opposed to natural ones (Smokorowski et al., 2011). Marchetti et al. (2011) found little difference between richness in reduced flows versus higher "natural-like" flows. The literature highlights divergent macroinvertebrate responses to altered flows.

While the impact of dams and reservoirs on macroinvertebrate richness is an interesting subject on its own, in this article, we are interested in integrating these impacts into an engineering tool that helps decision making (Life Cycle Assessment; LCA). This work specifically aims to assess the potential loss of macroinvertebrates due to dams and reservoirs for multiple usages (hydropower, flood control, irrigation, drinking water, transportation or recreation). For example, in the case of hydropower, we would quantify how many macroinvertebrates would potentially be lost following the implementation of a dam and the creation of a reservoir (transformation from a river into a reservoir and the occupation of the river by a reservoir) compared to a natural reference and use this information to relate this loss to the kilowatt-hour produced.

LCA is an interdisciplinary and internationally used approach that evaluates the potential environmental impacts of a product, process or service throughout its entire life cycle from resource extraction to end of life (ISO, 2006a). LCA is often used to support the selection of environmentally preferable alternatives, for eco-design purposes and to identify the largest potential environmental impacts and trade-offs in a product's life cycle (Hellweg & Milà i Canals, 2014). Emissions, resource extraction and change in land use (inventory flows) related to all activities involved in the life cycle of a product, process or service are first inventoried and this is called the Life Cycle Inventory (LCI). Then, these inventory flows are translated into potential environmental impacts through Characterization Factors (CF; Curran et al., 2011). In other words, CFs are used to translate inventory flows into impact indicators. Impact indicators are then attributed to Areas of Protection (AoP), which traditionally include ecosystem quality, human health, and resource and ecosystem services (Verones et al., 2017). Life Cycle Impact Assessment (LCIA) is the characterization and attribution of the impact. For the ecosystem quality AoP, the use of Potentially Disappeared Fraction (PDF) of species, which can also account for time and space, is recommended as a robust impact indicator (Verones et al., 2017).

The impacts of the transformation of a river into a reservoir (and its subsequent occupation by the reservoir, for a given amount of time) on ecosystem quality have received little attention in LCA. To our knowledge, only a few attempts have been made to evaluate changes in fish richness in relation to hydropower within a LCA framework (see Turgeon et al., 2021 and Dorber et al., 2019). Moreover, this type of work has only been conducted on fish (2021) and/or mostly relies on theoretical species richness curves, such as the species-discharge relationship (SDR; Dorber et al., 2019) or the species-area relationship (SAR). Because these curves are based on ecosystems that are in a state of equilibrium, they are not especially representative of the biological reality in a dam/reservoir impacted environment (Rosenberg et al., 2000; Xenopoulos & Lodge, 2006).

In this study, the objective is to assess the potential impacts of reservoir occupation (transforming a river into a reservoir and the subsequent occupation of the riverbed by the reservoir), upstream of the dam, on changes in macroinvertebrate richness using biological empirical data rather than theoretical curves. We used a dataset of 134 reservoirs (impacted sites) and 2062 rivers and streams (reference sites) across the United States and used a space-for-time substitution approach (reference versus impacted sites instead of before-after assessment; Pickett, 1989). We used PDF as a response variable, derived from reference and impacted macroinvertebrate richness, at three spatial scales: the scale of the United States, the scale of nine ecoregions, and the scale of singular reservoirs. We then used variation partitioning to examine which explanatory variables, from a set of 37, best explained the observed variation in reservoir PDF. Finally, we developed an empirical explanatory model to be used by LCA modelers and practitioners, using reservoir-related explanatory variables to explain variation in PDFs. The originality of this study relies (1) on the choice of a new group of aquatic organisms (macroinvertebrates), which should be monitored together with the other types of organisms (fish, aquatic vegetation), within a holistic perspective; and (2) the use of empirical values of richness instead of model predictions from theoretical curves to develop empirical PDFs for the impact of reservoir occupation on macroinvertebrate richness in LCA studies.

5.2.3 Methodology

5.2.3.1 *Life Cycle Impact Assessment (LCIA) framework*

CFs are determined by characterization models based on one of two methods. The first method uses environmental mechanisms of a physical, chemical or biological nature, and links inventory

flows (emissions of pollutant, the extraction/consumption of a resource or a change of land use) to impact indicators. The USEtox model (Rosenbaum et al., 2008), for example, builds mechanistic cause-effect chains to account for the environmental fate, exposure, and effects to potential ecotoxicity impacts from toxic emissions. Alternatively, the second method uses empirical observations of the state of the environment, assuming a causality between the observed impact and the inventory flows. For instance, de Baan et al. (2013) calculated CFs for several types of land use relying on empirical species richness data from both human-modified and undisturbed land in the same region.

Two categories of impact indicators exist for the ecosystem quality AoP. The first quantifies the temporary loss of species in time and space, and is expressed in PDF·m^2·yr (potentially disappeared fraction of species over a given area and duration; Jolliet et al., 2003) or in species·yr (Goedkoop et al., 2009). The second quantifies the permanent loss of species at the continental or global scale and is expressed in PDF (Chaudhary et al., 2015; de Baan et al., 2013). Both categories of indicators are relevant and complementary to one another. The first allows the assessment of temporary degradation of an ecosystem that will ultimately recover, whereas the second allows the assessment of the absolute loss of species. In this study, we chose to use the first category of indicators (temporary loss of species in time and space) to quantify the temporary damage on freshwater ecosystems due to reservoir occupation in space and time, using macroinvertebrates richness.[GT14] The framework used in de Baan (2013) and Chaudhary (2015) for change in land occupation has been adapted to assess the occupation of a water body (with an inventory flow expressed in surface-time units; m^2·yr). In de Baan (2013) and Chaudhary's (2015) approaches, the impact indicator is developed from an empirical model assessing land use impacts on biodiversity, and is expressed in PDF·m^2·yr, with a characterization factor expressed in PDF (implicitly PDF·m^2·yr/m^2·yr of land occupied). In our study, the CF is also expressed in PDF units, or implicitly PDF·m^2·yr[MM15]/m^2·yr of water body occupied. This CF is the observed change in richness, with respect to a reference macroinvertebrate community, and is multiplied by the inventory flow (m^2·yr of water body occupied during a given time) to obtain an impact score expressed in PDF·m^2·yr. We did not measure the damage due to water body transformation (change of water body area, according to certain requirements of a new occupation process, measured in surface unit; Lindeijer et al., 2002; Milà i Canals et al., 2007) but only the damage of water body occupation, although both impacts are complementary, due to the lack of available post-transformation, water body recovery data.

5.2.3.2 Data collection

5.2.3.2.1 Macroinvertebrate richness

To extract data on macroinvertebrate richness in reservoirs (impacted sites, after impoundment), we used the 2012 National Lake Assessment (NLA), a United States Environmental Protection Agency (USEPA) effort that surveys ponds, lakes and reservoirs in the United States, as well as their associated biological, chemical, physical and recreational characteristics (USEPA, 2015a). From this dataset, we retrieved macroinvertebrate richness (RICHNESS; taxonomic resolution at the genus level, except for oligochaetes, mites, and polychaetes, which were identified to the family level, and ceratopogonids at the subfamily level; USEPA, 2017), a unique identifier (UID) for each reservoir, latitude (LAT), longitude (LON), ecoregion (ECO), and a suite of environmental variables from 134 reservoirs across the United States (Figure 5.1; reservoirs shown in black; Table 5.1).

To extract data on macroinvertebrate richness in rivers and streams (reference sites, before impoundment), we used the 2008-2009 National Rivers and Streams Assessment (NRSA), a USEPA initiative to survey United States rivers and streams' biological, chemical, physical, and recreational characteristics (USEPA, 2015b). The same variables were collected (UID, LAT, LON, ECO and RICHNESS) for 2062 rivers and streams across the United States (Figure 5.1; rivers and streams shown in white). Environmental variables found in NLA reservoirs were not available for rivers (no elevation, no surface area and no trophic state, for example). Rivers and streams are referred to as natural reference sites and are not considered as unpolluted or pristine. They represent a wide range of conditions (probability-based design) of rivers that could have been transformed and occupied by reservoirs.

As no macroinvertebrate richness information was available for reservoirs pre-impoundment conditions, we applied a space-for-time substitution approach (Pickett, 1989), that is substituting spatial data for unavailable temporal data, assuming that the temporal relationship can be substituted by the spatial relationship between an explanatory variable and a response variable (Banet & Trexler, 2013).[GT16] We assumed that macroinvertebrate richness in rivers and streams in the surrounding area of a reservoir from the NLA dataset would be comparable to what would have been found in a river prior to its transformation and occupation by a reservoir and thus, could be used to derive PDFs. Both the USEPA-NLA and NRSA surveys used the same sampling procedure. Macroinvertebrates were collected using a semi-quantitative sampling of multiple

habitats (in reservoirs or in rivers and streams) with a 500 µm mesh D-frame dip net (see USEPA (2011) and USEPA (2007) for more information).

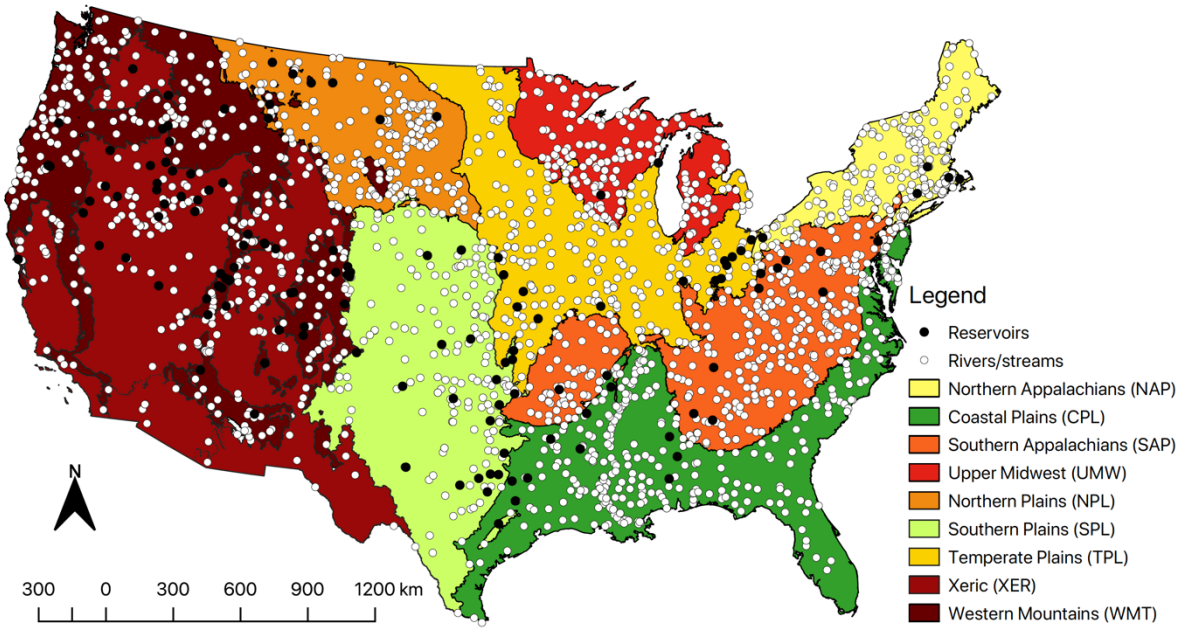

Figure 5.1 Map of the distribution of National Lake Assessment (NLA) reservoirs (n = 134; black circles) and National River and Streams Assessment (NRSA) rivers and streams (n = 2062; white circles) from the United States Environmental Protection Agency (USEPA), as well as the nine color-coded ecoregions.

Table 5.1 Table showing the explanatory variables from four matrices using the United States Environmental Protection Agency – National Lake Assessment (USEPA-NLA) dataset. The table shows the explanatory variables, a short definition of the variables, their respective units and the type of variable (N for numerical and F for categorical). Variables in bold are the most influential variables to explain variation in potentially disappeared fraction (PDF) of species following variation partitioning.

Matrix	Variable	Definition	Units	Type
Spatial	Latitude	Latitude of reservoir	Decimal degrees	N
	Longitude	Longitude of reservoir	Decimal degrees	N
	Ecoregion	National Aquatic Resources Surveys (NARS) 9-level reporting regions, based on aggregated Omernik (1987) level III ecoregions	-	F

Table 5.1 (continued)

Matrix	Variable	Definition	Units	Type
Spatial	Temperature	Annual mean air temperature, specific to ecoregion	°C	N
	Precipitation	Annual mean precipitations, specific to ecoregion	mm	N
	Forested	Percentage of land cover in ecoregion that is forested	%	N
	Cultivated pasture	Percentage of land cover in ecoregion that is cultivated pastures	%	N
	Wetlands	Percentage of land cover in ecoregion that is wetlands	%	N
	Grassland and shrubs	Percentage of land cover in ecoregion that is grasslands and shrubs	%	N
	Developed	Percentage of land cover in ecoregion that is developed	%	N
	Water or barren	Percentage of land cover in ecoregion that is water or barren	%	N
Physical	**Area**	Surface area of reservoir	ha	N
	Elevation	Elevation reservoir coordinates	m	N
	Macrophytes	Index of total cover of aquatic macrophytes of reservoir	-	N
	Shallow water	Shallow water habitat condition indicator	-	N
	Riparian vegetation	Riparian vegetation condition indicator	-	N
Chemical	**Trophic.state**	Trophic state of reservoir (oligotrophic and eutrophic)	-	F
	Secchi	Secchi depth	m	N
	DOC	Dissolved Organic Carbon level	mg/L	N
	PTL	Total Phosphorus Level	µg/L	N
	Color	Water color	PCU	N
	Conductivity	Water conductivity level	µs/cm	N
	NTL	Total Nitrogen Level	mg/L	N
	pH	pH level	pH scale	N
	Methylmercury	Top sediment methylmercury level	ng/L	N
	Chl-α	Chlorophyll-α measurement result of reservoir	µg/L	N

Table 5.1 (continued)

Matrix	Variable	Definition	Units	Type
Human	Buildings	Human influence by buildings around reservoir shoreline	-	N
	Commercial	Human influence by commercial activities around reservoir shoreline	-	N
	Crops	Human influence by crops around reservoir shoreline	-	N
	Docks	Human influence by docks around reservoir shoreline	-	N
	Landfill	Human influence by landfill around reservoir shoreline	-	N
	Lawn	Human influence by lawn around reservoir shoreline	-	N
	Park	Human influence by parks around reservoir shoreline	-	N
	Pasture	Human influence by pastures around reservoir shoreline	-	N
	Powerlines	Human influence by powerlines around reservoir shoreline	-	N
	Roads	Human influence by roads around reservoir shoreline	-	N
	Walls	Human influence by walls around reservoir shoreline	-	N
	Other	Human influence by other around reservoir shoreline	-	N

5.2.3.2.2 Ecoregions

Reservoirs and rivers were distributed across nine terrestrial ecoregions, *a priori* defined by Omernik (1987) and Herlihy et al. (2008). Ecoregions are based on similar environmental characteristics (climate, vegetation, soil type and geology) and macroinvertebrate assemblages (Figure 5.1); Coastal Plains (CPL), Northern Appalachians (NAP), Northern Plains (NPL), Southern Appalachians (SAP), Southern Plains (SPL) Temperate Plains (TPL), Upper Midwest (UMW), Western Mountains (WMT) and Xeric (XER). This aggregation of ecoregion was adopted for both the NLA and NRSA surveys (USEPA, 2016). For each ecoregion, we also extracted land cover variables (USEPA, 2016; spatial matrix in Table 5.1) and variables related to human impacts (Table 5.1; human matrix).

5.2.3.2.3 Native riverine taxa definition

The taxa pool observed in rivers and streams was used as a baseline to compare taxa richness before and after reservoir occupation (reference; native riverine taxa). We used only native riverine taxa and excluded all new taxa that would be encountered in a lake-like habitat (reservoir), since they would most likely not be present in a pre-reservoir occupation, river-like habitat. We considered using pairwise comparisons (impacted site paired with a single reference site) or reference sites found within a fixed radius or within the ecoregion. We decided to go with an ecoregion mean reference because there is neither literature to support the choice of a singular river when multiple rivers were surrounding a reservoir nor to support a fixed radius distance (25 km, 50 km). Comparing to a mean reference in each ecoregion, instead of a single river or stream close to the reservoir, ensures that we are measuring the impacts from a set of reference conditions and not a singular pristine, or impacted, river or stream. Moreover, as specified in section 2.2.2, ecoregions were defined based on similar macroinvertebrate assemblages, which further reinforce the choice of this scale, at the ecological point of view, for our study.

5.2.3.2.4 PDFs calculation

We calculated PDFs as the difference in richness between river (x) and reservoir richness (y), divided by the river richness (x). The PDF is a dimensionless proportion ranging between -1 and 1. At the United States scale (PDF_{usa}), we compared the overall United States mean native riverine richness in rivers and streams (one observation of richness per river or stream averaged over the United States; $\bar{x}_{usa}$; n = 2062) to the overall United States mean richness in reservoirs (one observation of richness per reservoir averaged over the United States; $\bar{y}_{usa}$; n = 134) to obtain a United States specific change in richness, as per equation 5.1;

$$PDF_{usa} = \frac{\bar{x}_{usa} - \bar{y}_{usa}}{\bar{x}_{usa}} \tag{5.1}$$

At the ecoregion scale (PDF_{eco}), we compared ecoregion mean native riverine richness of all rivers and streams (one observation of richness per river or stream averaged over each ecoregion; $\bar{x}_{eco}$) to the ecoregion mean richness in reservoirs (one observation of richness per reservoir averaged over each ecoregion; $\bar{y}_{eco}$) to obtain an ecoregion specific change in richness, as per equation 5.2;

$$PDF_{eco} = \frac{\bar{x}_{eco} - \bar{y}_{eco}}{\bar{x}_{eco}} \tag{5.2}$$

At the reservoir scale (PDF_{res}), we compared ecoregion mean native riverine richness of all rivers and streams (one observation of richness per river or stream averaged over each ecoregion; $\bar{x}_{eco}$)

to the richness of a specific reservoir within the same ecoregion (one specific richness observation per reservoir, no averaging; y_{res}) to obtain a reservoir specific change in richness, as per equation 5.3;

$$PDF_{res} = \frac{\bar{x}_{eco} - y_{res}}{\bar{x}_{eco}} \quad (5.3)$$

5.2.3.3 Data analysis and empirical modelling

5.2.3.3.1 Regionalization and ANOVA

Regionalization is a critical aspect in LCA (Patouillard et al., 2016; Yang, 2016). It accounts for existing spatial variability to improve results' representativeness and reduce spatial uncertainties (Patouillard et al., 2016). CFs must be developed at an appropriate scale to capture the environmental impacts of a product, process or service, and inform decision-makers. As a first step, we ran a one-way randomized-group analysis of variance (ANOVA) to determine whether PDF_{eco} differed across ecoregions (ecoregion scale) and thus test the relevance of this regionalization scale. We assessed the significance of regionalization at the ecoregion scale and identified which ecoregions were significantly different from each other based on the standardized mean difference and its confidence interval (CI). All statistical analyses were made using R version 3.0.2 (R Core Team, 2017). We conducted the ANOVA with the *ind.oneway.second* function in the *rpsychi* R package version 0.8 (Okumura, 2012).

5.2.3.3.2 Variation partitioning to explain the variation observed in our PDF_{res}

As a second step, we were interested in understanding which variables explained the variation observed in PDF_{res} at the reservoir scale in the United States. To do so, we used variation partitioning (Legendre, 2008), a statistical analysis that describes how a set of explanatory matrices explains the shared variation observed in a response variable (PDF). We built four explanatory matrices (spatial, physical, chemical and human matrices) based on the available descriptive variables from the NLA and NRSA datasets and selected a set of variables potentially influencing macroinvertebrate richness based on expert judgment.[GT17] The spatial matrix included variables describing the location of the reservoirs; latitude, longitude, ecoregion, temperature, precipitation and types of land covers (forested, cultivated pastures, wetlands, grasslands and shrubs, developed and water or barren). The physical matrix included variables describing the reservoir; reservoir area, elevation, shallow water and riparian vegetation. The chemical matrix included variables that describe the biochemical state of the reservoir; trophic state, Secchi depth, dissolved organic

carbon (DOC), total phosphorus level (TPL), water color, conductivity, total nitrogen level (TNL), pH, methylmercury and chlorophyll-α (Chl-α). The human matrix included variables describing the human activity, impact or influence around the reservoir shoreline; influence of buildings, commercial activities, crops, docks, landfills, lawns, parks, pastures, powerlines, roads, walls and others. See Table 5.1 for a complete description of the variables included in each matrix. To achieve the most parsimonious analysis, we performed a stepwise selection procedure on each explanatory matrix to identify which variable, or combination of variables, best explained the variation in PDF_{res} (variables in bold; Table 5.1). Variable selection was performed with the function *ordiR2step* and variation partitioning was conducted with the *varpart* function in the *vegan* R package version 2.5-2 (Oksanen et al., 2019).

5.2.3.3.3 Empirical modelling

As a third step, we used a multiple linear regression (*lm* function in the *stat* R package version 3.4.2; R Core Team, 2017) to develop an empirical model to be used by LCA modellers and practitioners. This empirical model can be used to assess PDF_{res} from known values of explanatory variables (related to the reservoir, not the rivers and streams) when we do not have information about empirical change in macroinvertebrate richness in impacted and/or reference sites, within a specific frame of application and range of environmental variables (also called interpolation). We used the variables identified by the variation partitioning analysis as the most influential variables to explain the variation in PDF_{res}. We checked whether assumptions associated with multiple linear regression were violated (the residuals are independent, normal, have a mean of 0 and are homoscedastic; Figure B.1 and B.2), we deleted a few outliers, and performed a model selection procedure. We applied a manual backward selection procedure, used the recommended information theoretic approach based on Akaike information criterion (AIC; Burnham & Anderson, 2002) and the Bayesian information criterion (BIC; Schwarz, 1978) to compare the seven candidate models, and selected the model with the highest support (Table 5.2). For each explanatory variable selected in the final model, we extracted estimates and standard error (SE), where the estimates represent the direction and magnitude of PDF_{res}.

Table 5.2 Summary of statistical candidate models, in order of plausibility (Akaike information criterion; ΔAIC, and Bayesian information criterion; BIC), where PDF stands for potentially disappeared fraction of species, ELE for elevation, AREA for surface area, T.S. for trophic state,

PH for pH level, LAWN for influence of lawns and ROAD for influence of roads. For each candidate model, the estimate for the intercept is labelled b_{int} and all other bs (b_{ELE}, b_{AREA}, $b_{T.S.}$, b_{PH}, b_{LAWN}, b_{ROAD}), estimate for the slope of their respective variable. See Table 5.1 for full description of the variables used. § Marginally significant.

Models	Non-significant variables	ΔAIC	BIC
(A) PDF ~ b_{int} + b_{ELE}*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH + b_{LAWN}*log10(LAWN) + b_{ROAD}*log10(ROAD)	PH§, LAWN and ROAD	2	27
(B) PDF ~ b_{int} + b_{ELE}*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH + b_{LAWN}*log10(LAWN)	PH§ and LAWN	1	24
(C) PDF ~ b_{int} + b_{ELE}*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH	PH§	0	20
(D) PDF ~ b_{int} + b_{ELE}*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS	None	2	20
(E) PDF ~ b_{int} + b_{ELE}*sqrt(ELE) + b_{AREA}*log10(AREA)	None	8	23
(F) PDF ~ b_{int} + b_{ELE}*sqrt(ELE)	None	19	32
(G) PDF ~ b_{int}	-	52	63

5.2.4 Results

5.2.4.1 PDF_{usa} and variation in PDF_{eco} across ecoregions

A total of 973 native riverine macroinvertebrate taxa were inventoried throughout the United States. The mean native riverine richness per ecoregion varied from 26.1 ± 13.1 to 46.0 ± 13.8 (mean ± standard deviation [SD]; Table 5.3 and B.1). The mean reservoir richness per ecoregion varied from 19.7 ± 7.9 to 39.8 ± 9.4 (mean ± SD; Table 5.3). Our empirically derived PDF_{usa} and PDF_{eco} showed a loss in macroinvertebrate richness due to reservoir occupation in the United States and this loss followed a longitudinal gradient associated with the ecoregions (Figure 5.2). At the United States scale, 28% of macroinvertebrate taxa disappeared in response to river impoundment (PDF_{usa} = 0.284 ± 0.168 [mean ± SD]; Table 5.3 and Figure 5.2). At the ecoregion scale, seven out of nine ecoregions (78%) showed a statistically significant loss of macroinvertebrate taxa, with PDF_{eco} varying from 0.135 ± 0.052 to 0.464 ± 0.235. Two ecoregions (CPL and SPL) showed a significant increase in macroinvertebrate taxa (PDF_{CPL} = -0.158 ± 0.100 and PDF_{SPL} = -0.021 ± 0.014; Table 5.3 and Figure 5.2). PDF_{CPL}, PDF_{SPL}, PDF_{XER} and PDF_{WMT} significantly differed from most ecoregions, whereas PDF_{NAP}, PDF_{TPL}, PDF_{NPL}, PDF_{SAP} and PDF_{UMW}, showed much less significant differences (Figure 5.2). Those PDF_{eco} were mostly all characterized by smaller sample sizes (respectively, n = 4, 15, 8, 12 or 2). Results from the ANOVA suggest a longitudinal gradient of impact, where PDF_{eco} are higher in the western part of the country and lower in the eastern part of the country (Figure 5.2). At the reservoir scale, PDF_{res} varied from -0.584 ± 0.342 (observation

± pooled SD) to 0.924 ± 0.464, and 74% of the reservoirs showed a significant loss of macroinvertebrate taxa (Table B.1).

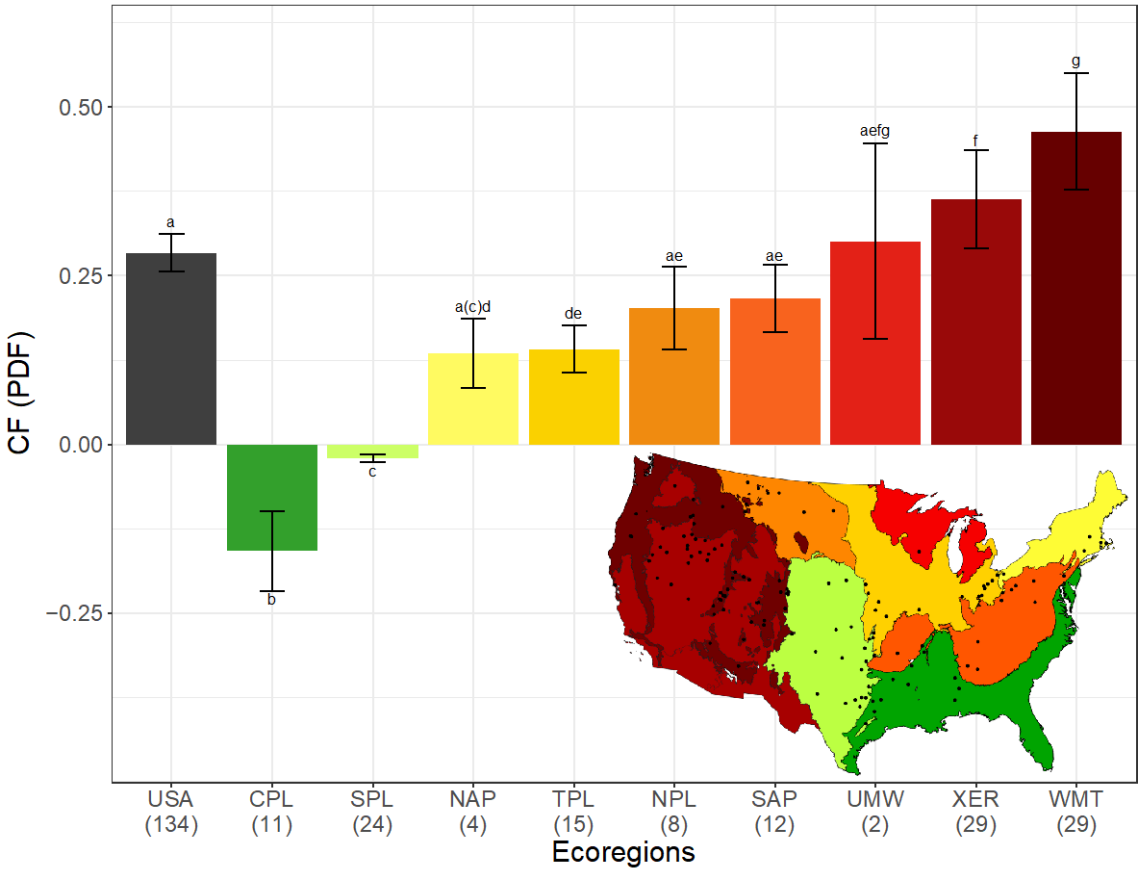

Figure 5.2 Barplot showing a mean characterization factor (CF) in potentially disappeared fraction of species (PDF ± 95% confidence interval; CI) at the United States (USA) level (PDF_{usa} shown in dark grey) and at the ecoregion level (PDF_{eco} color-coded with ecoregions as a gradient of intensity). We used letters to identify which PDF_{eco} differed or not from each other. When two bars share a letter, they are not significantly different from each other and marginally not significantly different from each other when the letter is in parentheses. A positive PDF represents a loss of taxa, whereas a negative PDF represents a gain of taxa. Ecoregions are abbreviated as follows; Coastal Plains (CPL), Northern Appalachians (NAP), Northern Plains (NPL), Southern Appalachians (SAP), Southern Plains (SPL) Temperate Plains (TPL), Upper Midwest (UMW), Western Mountains (WMT) and Xeric (XER). Sample number from which mean reservoir richness and PDFs were calculated is also shown on the x-axis in parentheses. For specific values, refer to Table 5.3.

Table 5.3 Table showing the mean native riverine richness for each ecoregion (± standard deviation; SD), sample number from which mean native riverine richness was computed (*n.riv*), mean impacted reservoir richness for each ecoregion (± SD), potentially disappeared fraction of species (PDF ± SD and ± 95% confidence interval [CI]) values and the sample number (*n.res)* from which mean reservoir richness and PDF was calculated is also shown for the United States and the nine ecoregions. A positive PDF represents a loss of taxa, whereas a negative PDF represents a gain of taxa.

Ecoregion or country	**Mean nat. riv. richness**	**± SD**	***n.riv***	**Mean imp. res. richness**	**± SD**	**PDF·m²·yr /m²·yr**	**± SD**	**± 95% CI**	***n.res***
USA	35.9	15.5	2062	25.7	10.4	0.284	0.168	0.028	134
CPL	28.4	16.6	327	32.9	7.9	-0.158	-0.100	-0.059	11
SPL	26.1	13.1	176	26.7	11.8	-0.021	-0.014	-0.006	24
NAP	46.0	13.8	225	39.8	9.4	0.135	0.052	0.051	4
TPL	32.0	12.9	209	27.5	7.7	0.141	0.069	0.035	15
NPL	29.4	10.1	179	23.5	6.6	0.202	0.090	0.062	8
SAP	45.8	15.1	344	35.9	8.8	0.216	0.089	0.050	12
UMW	39.3	12.7	167	27.5	3.5	0.301	0.105	0.145	2
XER	31.0	11.6	213	19.7	7.9	0.363	0.199	0.073	29
WMT	39.8	12.0	222	21.3	8.6	0.464	0.235	0.086	29

5.2.4.2 *Variables explaining the variation in PDF_{res}*

At the reservoir scale, the four matrices (spatial, physical, chemical, and human) explained approximately 51% of the total variation in PDF_{res} (variation partitioning; Figure 5.3; Table B.2). Approximately 46% of the variation was explained by the combined effects of the spatial (ecoregion) and physical (elevation and surface area) matrices. Spatial matrix (ecoregion) explained 25% of the variation, over which 24% of this variation was shared with the physical matrix (elevation and surface area), 11% was shared with the chemical matrix (pH and trophic state), and 8% was shared with the human matrix (presence of lawn and road adjacent to the reservoir shoreline; Figure 5.3). The physical matrix explained 45% of the variation. Elevation and surface area alone (variation not shared with the other matrices) explained 15% of the variation. The chemical and human matrices explained respectively 18% and 14% of the variation.

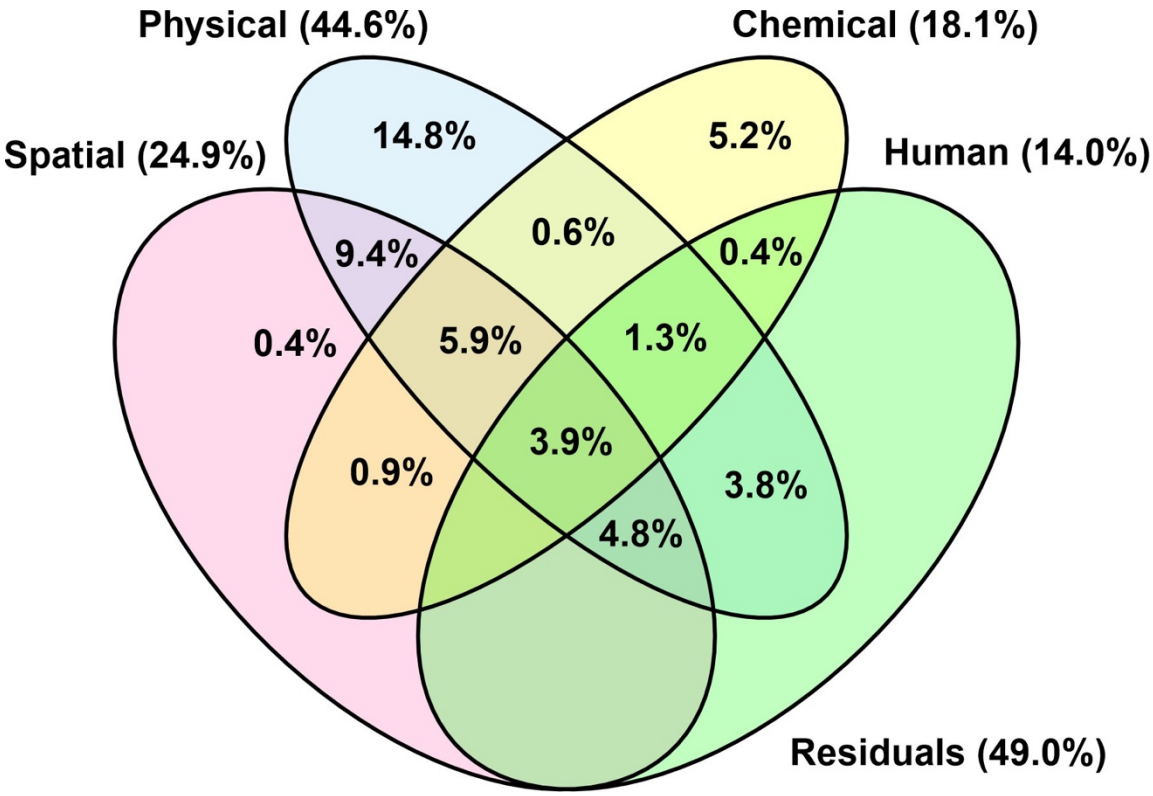

Figure 5.3 Venn diagram showing variation partitioning of a response matrix (potentially disappeared fraction of species; PDF) explained by four matrices, that is spatial matrix (ecoregion; ECO), physical matrix (elevation; ELE, and surface area; AREA), chemical matrix (trophic state; T.S. and, pH) and human matrix (influence of lawns; LAWN, and influence of roads; ROAD). Values < 0 not shown.

5.2.4.3 *PDF_{res} empirical model*

According to the empirical model (Equations 5.4 and 5.5; Figure 5.4), almost 50% of the observed variation in PDF_{res} (partial $R^2_{adj} = 0.49$; $p < 0.001$; $n = 134$) was explained by elevation (35%), trophic state (either oligotrophic or eutrophic; 4%), and reservoir surface area (10%). No more than 50% of the variation explained is acceptable in ecology disciplines, since there is substantial environmental variation that cannot be accounted for, unless specifically sampled for. [GT18] PDF_{res} was positively related to reservoir elevation, where higher elevation was associated to higher PDF_{res} (Figure 5.4). PDF_{res} was negatively related to eutrophication status. Oligotrophic reservoirs (<10μg/l total phosphorus) had higher PDF_{res} than eutrophic reservoirs (>10μg/l total phosphorus). As for reservoir surface area, there was a positive relationship between reservoir surface area and PDF_{res}, where bigger reservoirs had a higher PDF_{res} than smaller ones (Figure 5.4). To summarize,

large oligotrophic reservoirs located at higher elevation were most likely to have higher macroinvertebrate PDF_{res}.

$$PDF_{res[OLIGOTROPHIC]} = -0.129(\pm 0.109) + 0.013(\pm 0.002) \cdot sqrt(ELE) + 0.170(\pm 0.043) \cdot \log_{10}(AREA) \quad (5.4)$$

Values in parentheses are SE of the estimate.

$$PDF_{res[EUTROPHIC]} = -0.454(\pm 0.102) + 0.013(\pm 0.002) \cdot sqrt(ELE) + 0.170(\pm 0.043) \cdot \log_{10}(AREA)$$

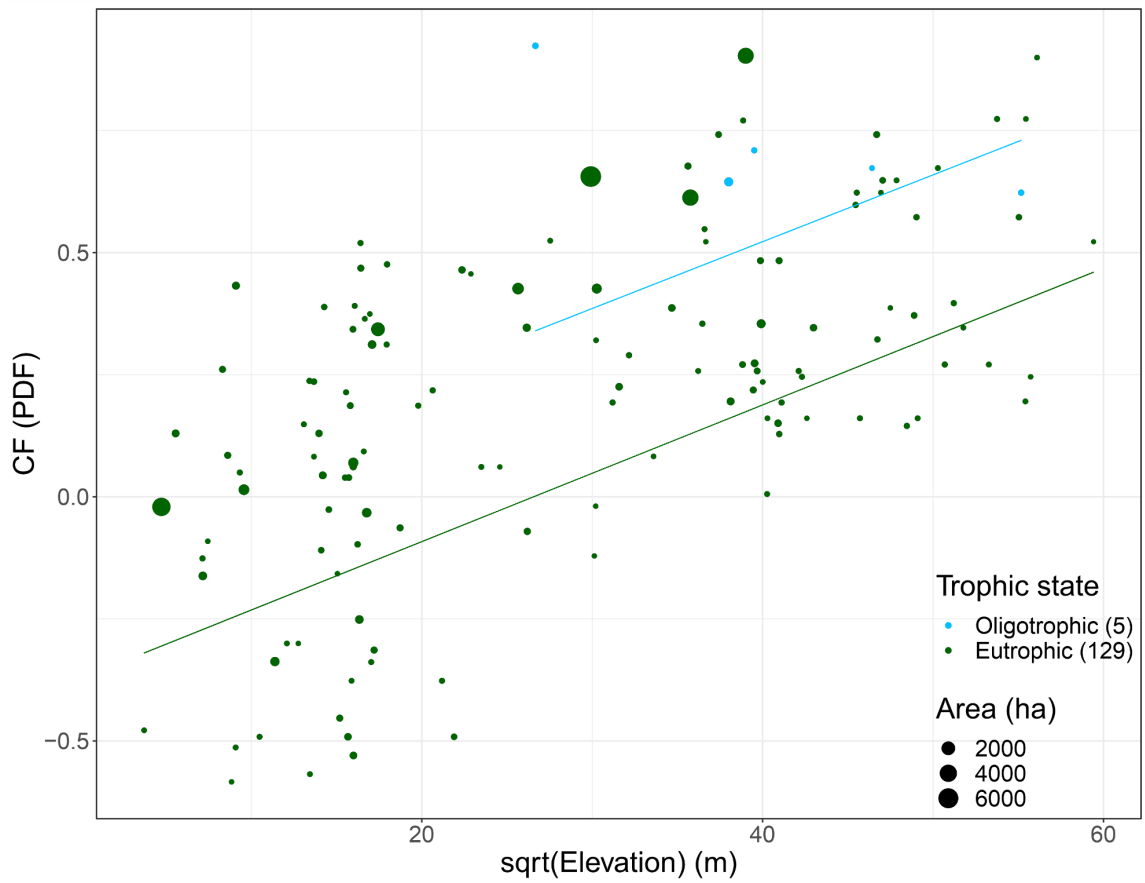

Figure 5.4 Graphical representation of our empirical model showing the relationship between characterization factors (CF) in potentially disappeared fraction (PDF) of species, reservoir elevation in meters and square root-transformed (m; ELE) and trophic state (oligotrophic [<10µg/l total phosphorus] or eutrophic [>10µg/l total phosphorus]; T.S.). Trophic state is color coded (sample number shown in parentheses) and point size is representative of reservoir surface area in hectares (ha; AREA).

5.2.5 Discussion

From the examination of 134 reservoirs of varied usages (flood control [n = 6], hydropower [n = 2], recreational [n = 23], soil erosion prevention [n = 5], transport [n = 2], water supply [n = 49] and unknown [n = 47]) and 2062 rivers and streams across the continental United States, our results showed a general loss of approximately 28% (PDF_{usa}) of macroinvertebrate taxa following reservoir occupation at the scale of the United States. PDF_{eco} also varied across ecoregions. Almost 25% of the total variation observed in PDF_{res} was explained by the nine ecoregions, pressing the need for regionalized CFs. We provided evidence that the empirical PDFs for macroinvertebrates were consistent and uniform across the three spatial scales (macroinvertebrate taxa loss at the scale of the country; PDF_{usa}, the majority of ecoregions; PDF_{eco}, and most reservoirs; PDF_{res}). Overall, the empirical PDFs derived in this study can be used as CFs in the LCA framework to evaluate the potential impact of reservoir occupation on the ecosystem quality AoP for a specific reservoir (PDF_{res}), within a given ecoregion (PDF_{eco}) or over the United States (PDF_{usa}). Potential impact scores expressed in $PDF \cdot m^2 \cdot yr$ can be calculated multiplying PDF by the area-time occupied by the reservoir for a given product or service. We also provided a simple empirical model based on three explanatory variables (elevation, trophic state and reservoir surface area) that explained 49% of the variation in macroinvertebrate PDF_{res}. Reservoirs at higher elevation, with lower levels of eutrophication and bigger surface area had higher PDF_{res}. This empirical model could be used by LCA practitioners to interpolate CFs based on few explanatory variables. However, we did not test the transferability of our model to other countries, or to reservoirs outside of the ranges of application of this model (elevation between 13 and 3531 meters [m] and area between 2 and 6560 hectares [ha] for eutrophic reservoirs, and elevation between 711 and 3044 m and area between 12 and 408 ha for oligotrophic reservoirs; specific regression lines in Figure 5.4).

5.2.5.1 United States taxa loss and regionalization

At the scale of the United States, 28% of macroinvertebrate taxa disappeared following reservoir occupation. This result suggests that reservoir occupation does affect the rate of change in macroinvertebrate richness, and this is consistent with the literature estimates of the impacts of hydropower on macroinvertebrate richness across the world (Aroviita & Hämäläinen, 2008; Behrend et al., 2012; Englund & Malmqvist, 1996; Jackson et al., 2007; Kraft, 1988; Kullasoot et al., 2017; Malmqvist & Englund, 1996; Takao et al., 2008; Valdovinos et al., 2007; White et al., 2011). Presently, there is still no macroinvertebrate CF (PDF) available to assess potential impacts

of reservoir occupation on ecosystems biodiversity associated to a product or service in LCA. Therefore, the environmental impact of activities, such as hydropower supply (implying the occupation), on biodiversity is considered null. Our research provides the first empirically derived multi-scale macroinvertebrate-based PDF values to the LCA community and fills in an important gap in this field of research. Our PDFs, in complement to fish-based PDFs (see Turgeon et al., 2021 and Dorber et al., 2019), could also allow for a more holistic approach, the generation of a multi-phyla CF, which would be more robust and representative of the ecosystem impacts. The PDF_{usa} covers a large geographical range across the United States, with substantial ecoregion variability. For this reason, we suggest using PDF_{eco} (at the ecoregion level). This study also showed that there was a significant difference between PDF_{eco} and the presence of a longitudinal gradient of impact with higher PDF_{eco} in the west. According to these results, reservoir occupation, regardless of its purpose, would have higher impacts in the western ecoregions of the United States. The WMT ecoregion is characterized by its mountains and valleys landscapes and a sub-arid to arid climate, where it gets rather humid and cold at higher elevation (USEPA, 2016). The XER ecoregion has lots of ephemeral rivers, relatively limited surface water supply and its climate varies widely from a xeric warm and dry environment to temperate conditions (USEPA, 2016). These types of conditions usually favor specialist taxa, which are highly adapted to their environment, and are known to be particularly sensitive to human impacts (García-Vega & Newbold, 2020; Harrison & Noss, 2017; Mykrä & Heino, 2017). Our results support these observations because PDF_{eco} are higher in those ecoregions, meaning that reservoir occupation has higher impacts on ecosystem quality and biodiversity. The observed spatial differentiation and longitudinal gradient of impact justify the need for regionalized CFs, which would improve the accuracy and robustness of LCA.

5.2.5.2 Elevation, trophic state and reservoir surface area

In our empirical model, a combination of elevation, trophic state and reservoir surface area explained most the variation in PDF_{res}. As reservoirs increase in elevation, their PDF_{res} also increase. High elevation ecosystems support smaller, isolated, prone-to-extinction populations, as well as a higher proportion of more vulnerable taxa, which makes these alpine ecosystems more sensitive to biodiversity loss following human impacts (Viterbi et al., 2013). Oligotrophic reservoirs, because of their low productivity (Wetzel, 2001), host relatively lower richness compared to mesotrophic/eutrophic reservoirs (Dodson et al., 2000). Thus, they are more sensitive

to taxa loss (loss of one taxon over a few taxa is relatively more important than over multiple taxa). This is reflected in our results, oligotrophic reservoirs have higher PDF_{res} than eutrophic ones. PDF_{res} were also shown to be higher in reservoirs with a larger surface area. This result is not clearly supported by the literature. Lake-size is one of multiple key factors affecting reservoir biodiversity (Heino & Tolonen, 2017; Jackson et al., 2001; Tonn & Magnuson, 1982), bigger reservoirs are more productive and more heterogeneous in terms of potential habitats and thus support more richness (biodiversity; SAR; Connor & McCoy, 1979; Heino, 2000; MacArthur & Wilson, 2001). One could then imagine that high biodiversity ecosystems would be less vulnerable to taxa loss proportionally speaking, which is not the case here. It is not clear to us as to why our larger reservoirs showed higher PDF_{res} because they did not share similar water usage, neither were they specifically located at high elevation, nor clustered in a specific ecoregion (Figure 5.5). This pattern could be biased by the unbalanced sample size.

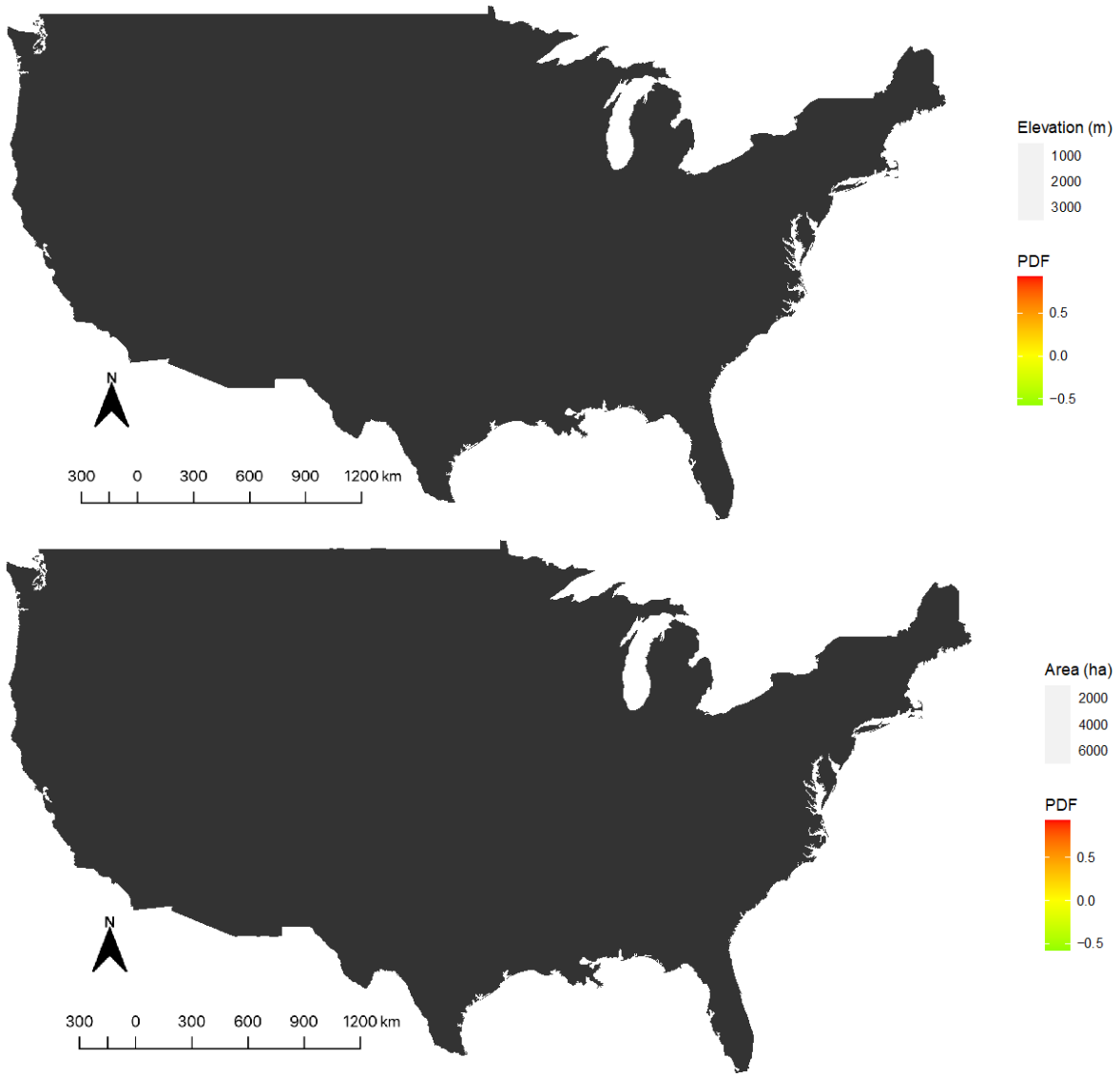

Figure 5.5 Heatmaps of potentially disappeared fraction (PDF) of species, elevation in meters (m; ELE) and surface area in hectares (ha; AREA) of reservoir is proportional to the point size.

Regarding the remaining 49% of unexplained variation in the variation partitioning, it would have been useful to have data related to flow regime dynamics in each reservoir as they are known to influence macroinvertebrates.[GT19] It would have also been useful to have habitat-specific characteristics related to each sample, such as granulometry and macrophyte coverage, two variables known to strongly influence the abundance and biodiversity of macroinvertebrate communities.[GT20] From an ecological point of view, our observations were mostly supported by the literature. The empirical model was built with a specific purpose in mind: to provide LCA practitioners with a simple model, based on a few explanatory variables. Collecting

macroinvertebrate richness data is time consuming and expensive, as well as demanding in terms of expertise for identification. The empirical model allows to interpolate robust PDF_{res} (± quantified error) for a specific reservoir using readily available information such as elevation, reservoir surface area, and trophic state.

5.2.5.3 *Limitations*

Five limitations can influence the strength of our results. First, we defined richness as the number of native riverine taxa. We did not account for the potential gain of lentic-specific taxa following reservoir occupation, therefore our PDFs are considered conservative.[GT21] When a river is transformed and occupied by a reservoir, some native riverine taxa are lost, and some lentic taxa can be gained. Thus, one should be careful when interpreting these gain in taxa (lentic, exotic or non-native invasive taxa) as they might not necessarily represent an ecosystem improvement (Verones et al., 2010). Based on the Habitat Diversity Hypothesis (HDH; Williams, 1964), where diversity of taxa is directly related to the diversity of habitats, lotic environments should be more diversified than lentic environments (reservoirs). Because of their narrowness and longitude, rivers run through a greater range of geological formations, as well as geographical regions, per unit of surface area and vary more in terms of substrate, water temperature and flow dynamics than lentic environment of comparable depth and size (Eadie et al., 1986; Horwitz, 1978; Moyle & Li, 1979). Thus, the higher environmental variability and productivity, as well as the presence of microhabitat heterogeneity in rivers likely support more taxa per surface area[GT22](Eadie et al., 1986; Gorman & Karr, 1978; Matthews, 1982). We could then assume that even after a lotic environment is transformed into and a lentic one, there would still be less taxa in the lentic environment. Moreover, gain of lentic taxa after reservoir occupation is often considered a misleading argument because the littoral zone in reservoirs is less complex, differs in physico-chemical conditions (Walker et al., 1992) and is generally negatively affected by varying water levels. These characteristics can affect the productivity of littoral areas, which are crucial to reservoir productivity, and can, in turn, affect its biodiversity. This further reinforces the potential overestimation of our PDFs. A second limitation of this study is that our CF is not independent from other impact categories, namely eutrophication. Because trophic state was defined as a significant variable to explain PDF, we had to incorporate this information in our model. In the LCA framework, eutrophication is already taken into account and thus, using it in our model could cause some bias in the overall compilation of impacts (double counting).[GT23] A third limitation from this study is the use of space-for-time

substitution approach. We do not have a before-after control-impact study design (BACI). Data on river and stream richness before reservoir occupation are not available so our results, and suggested PDFs, must be interpreted with caution. A fourth limitation of this study is the use of taxa richness (number of taxa) only to evaluate the impacts of reservoir occupation on biodiversity. It would be optimal to also assess changes in community composition (number of taxa and their respective abundance). [GT24]However, given that the current LCA framework (for example, IMPACT World+; Bulle et al., 2019) uses PDF (based on changes in taxa richness) and does not yet include impacts on community composition, it is not yet possible to include the impacts on community composition in the LCA framework. Doing so would also face important challenges regarding data availability to compute such a metric. Finally, the fifth limitation is that the performance of the empirical model has not been evaluated outside the USEPA-NLA dataset. Such evaluation through case studies and independent datasets should be performed to test the robustness of its predictive power.

5.2.6 Conclusion

Using a space-for-time substitution approach, we showed that the transformation and occupation of a riverbed by a reservoir resulted in a loss of 28% of macroinvertebrate taxa in the United States. This loss of richness also varied across ecoregions, pressing the need for regionalized PDFs. Patterns were consistent across scales (the United States, nine ecoregions and 134 reservoirs), where we observed a general loss of macroinvertebrate richness. These PDFs fill in an important gap in LCA, enabling the assessment of reservoir occupation impacts (involved in several common activities in the LCA of a product or service, such as hydropower, irrigation, drinking water, transportation or recreation) onto ecosystem quality. We also derived an empirical model to explain and interpolate PDFs as a function of three explanatory variables: reservoir elevation, trophic state and surface area. Our study generated PDFs using robust empirical richness data, rather than theoretical curves (SARs or SDRs), which is a novel approach in this specific branch of LCA. Our PDF also considered a new type of organism, macroinvertebrates, that can be used to complement the information already generated for fish, thus improving the robustness and representation of biodiversity impacts characterization in the LCA framework. Despite some highlighted limitations, the empirical CFs developed through this study constitute a strong contribution to assess the impacts of reservoir occupation on the ecosystem quality AoP. Natural follow-ups to this study would be to integrate macroinvertebrate-based CFs with fish-based CFs from Turgeon et al. (2021)

to improve the characterization of impacts on ecosystem quality and to evaluate the accuracy of the empirical model to other geographical contexts.

5.2.7 Acknowledgments

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CHAPTER 6 ARTICLE 3: TAXONOMIC RESOLUTION DOES NOT INFLUENCE THE DIRECTION AND MAGNITUDE OF CHARACTERIZATION FACTORS (CF) IN LIFE CYCLE ASSESSMENT (LCA)[GT25]

6.1 Introduction to Article 3

The article presented in this chapter evaluates the influence of taxonomic resolution on CFs assessing the impacts of reservoir occupation on macroinvertebrate richness, more specifically the influence of taxonomic resolution on the CFs' regionalization, directionality and magnitude, and CFs model selection and predictive ability, across reservoirs and ecoregions in the United States. This article is a natural follow-up and companion study to article 2 (chapter 5). It presents itself as more fundamental and methodological paper that could particularly resonates with LCIA modellers, allowing groups of organisms that are rarely identified up to the species level of taxonomic resolution to be included in multi-groups (of organisms) CFs. The authors of this article are Gabrielle Trottier, Katrine Turgeon, Daniel Boisclair, Cécile Bulle and Manuele Margni. It was submitted to the Science of the Total Environment journal on April 26th, 2021.

6.2 Manuscript

6.2.1 Abstract

In Life Cycle Assessment (LCA), Characterization Factors (CF) for the ecosystem quality Area of Protection (AoP) are predominantly calculated in terms of Potentially Disappeared Fraction (PDF) of species, which is based on the number of species (*i.e.*, richness) in reference and impacted ecosystems. For plants, fish and vertebrates, richness data at the species level of taxonomic resolution is readily available. However, when it comes to macroinvertebrates, taxa are rarely identified up to the species level of taxonomic resolution in bioassessments and it becomes difficult to generate multi-group CFs in PDF using a specific taxonomic resolution such as species. The aim of this study is to evaluate the influence of taxonomic resolution on the regionalization, the directionality and the magnitude of CFs assessing the impacts of reservoir occupation on

macroinvertebrate richness, as well as its influence on CFs model selection and predictive ability, across reservoirs and ecoregions in the United States. We identified four categories of taxonomic resolution, namely genus-exclusive, genus-inclusive, family-exclusive and family-inclusive. Our results suggested that taxonomic resolution did not significantly influence the regionalization of CFs, neither did it significantly influence CFs direction and magnitude, as well as empirical model selection and predictive abilities. Therefore, because of the additional amount of variation explained in richness, we advise that LCA practitioners use CFs calculated with a genus-based taxonomic resolution. We also recommend that inclusive taxonomic resolutions be favored as they are more representative of the richness observed in the studied ecosystem. Finally, based on limitations identified in this study, we strongly suggest that future research compare the influence of species-, genus- and family-based taxonomic resolution on CFs, as well as test the validity of this study's outcomes to other groups of organisms and geographical contexts.

6.2.2 Introduction

Biodiversity has been defined as "the variability among living organisms from all sources" by the Millennium Ecosystem Assessment (2005). It includes the variability among individuals of a single species, among species and among ecosystems, whether it be freshwater, marine or terrestrial (Duraiappah et al., 2005). Biodiversity is valued both intrinsically (*i.e.*, value attributed to biodiversity based on its sole existence, in other words useless to humans) and instrumentally (*i.e.*, value attributed to the biodiversity of an ecosystem from an anthropocentric point of view, only benefits to humans are considered; Verones et al., 2017). It is fundamental that biodiversity be preserved for its intrinsic value first, and second for its instrumental values. Fortunately, the intrinsic and instrumental values of biodiversity are closely linked, when preserving the intrinsic value of biodiversity, direct ramifications can be observed for the preservation of the instrumental value of biodiversity. Indeed, intrinsic biodiversity determines the provision of biologically (clean water and air), culturally (recreation and tourism) and economically (lumber, fisheries) important ecosystem services (Díaz et al., 2006; Duraiappah et al., 2005; Mace et al., 2012). These ecosystems services (instrumental values), which are highly dependent on ecosystem biodiversity through biomass production, decomposition and recycling (Cardinale et al., 2012), are critical to the survival of humans. Qualifying and quantifying the impacts of human activities on biodiversity

is therefore essential to develop conservation strategies and preserve said biodiversity, both intrinsically and instrumentally.

There are many ways to assess the impacts of human activities on biodiversity (*e.g.*, environmental impact assessment, risk analyses, etc.; Jolliet et al., 2010), here we are interested in the Life Cycle Assessment (LCA) framework. LCA is a tool used to quantify the potential environmental impacts of products, processes or services, over the entire life cycle, from resources extraction to end-of-life (ISO, 2006a). In LCA, potential impacts are assessed by multiplying an inventory of elementary flows (*i.e.*, quantity of resources used or emissions emitted during the life cycle) with corresponding Characterization Factors (CF; Crenna et al., 2020). A CF is derived from a characterization model and is applied to convert elementary flows into the common unit of an impact category (ISO, 2006b). In its simplest form, a CF is expressed as the multiplication of a fate factor (size, duration and propagation of an environmental intervention, for example reservoir occupation, in an ecosystem) by an effect factor (alteration intensity of the ecosystem; Curran et al., 2011; Núñez et al., 2016). Currently, in LCA, potential impacts are calculated for three Areas of Protection (AoP): natural resources and ecosystems services, human health and ecosystem quality (Bulle et al., 2019; Verones et al., 2017).

The Potentially Disappeared Fraction (PDF) of species is the common unit predominantly used in characterization models assessing potential impacts on the ecosystem quality AoP (Crenna et al., 2020; Curran et al., 2011). To our knowledge, PDFs were always computed using species richness data in LCA methodologies. This level of taxonomic resolution is usually readily available, especially with groups of organisms such as vertebrates and plants. However, when attempting to build CFs that are not specific to one group of organisms (*e.g.*, fish) but rather encompasses multiple groups of organisms (*e.g.*, fish, macroinvertebrates and plants), we encounter some difficulties in the process. Indeed, richness data that is resolved at the species taxonomic resolution can be challenging to access for some groups of organisms (*e.g.*, macroinvertebrates; de Oliveira et al., 2020). This is due to the extended amount of time required to process samples at the species level of taxonomic resolution, the small number of specialized taxonomists and the lack of investment in bioassessments (de Oliveira et al., 2020; Lawton et al., 1998; Magurran & Quieroz, 2010).

One way to overcome the lack of richness data resolved at the species level of taxonomic resolution, and facilitate the generation of multi-group CFs, is to use richness data with coarser taxonomic

resolutions, also referred to as taxonomic sufficiency (or the Higher Taxon Approach [HTA]; de Oliveira et al., 2020). This approach uses biological data at coarser taxonomic resolutions than species (genus, family, order or class; Ellis, 1985) and has already been used by multiple studies (*e.g.*, Alves et al., 2016; Landeiro et al., 2012; Mueller et al., 2013; Villaseñor et al., 2005). In their recent meta-analysis, de Oliviera et al. (2020) evaluated the reliability of the taxonomic sufficiency approach and high effect sizes strongly suggested that HTA was a reliable approach to reveal both richness and compositional patterns for multiple groups of organisms (plants, invertebrates and vertebrates) across numerous types of ecosystems (terrestrial and aquatic). Moreover, their analyses also showed that the effectiveness of the taxonomic sufficiency approach increased with spatial extent of bioassessments (de Oliveira et al., 2020).

Here, we touch basis with the concept of taxonomic sufficiency in the LCA framework. The aim of this study is to estimate the influence of taxonomic resolution on CFs assessing the impact of reservoir occupation (a riverbed that was transformed into a reservoir, subsequently occupied by said reservoir and further prevented from returning to its original state) on macroinvertebrate richness across the United States. While this study is a methodological companion to Trottier et al. (2021), and its objectives were derived from limitations that were identified and discussed in Trottier et al. (2021), this manuscript is standalone. The three main objectives, and related sub-objectives are first, to assess the influence of taxonomic resolution on CFs regionalization. More specifically, we want to examine if taxonomic resolution influences the magnitude and directionality of CFs across ecoregions in the United States. Second, to assess the influence of taxonomic resolution on models' selection. More specifically, we want to examine if explanatory variables and explanatory power differ with different taxonomic resolution. Third, we want to assess the influence of taxonomic resolution on CF models' predictive ability. More specifically, we want to examine the influence of taxonomic resolution on prediction error.

6.2.3 Methodology

The achievement of our objectives requires the computation of CFs using different categories of taxonomic resolution in a series of ecoregions in the United States. CFs are computed in PDF and necessitates macroinvertebrate richness data in ecosystems that serve as reference sites and in ecosystems impacted by reservoir occupation. Modelling of PDFs across ecosystems and

ecoregions entails environmental variables for each ecosystem impacted by the occupation of a reservoir.

6.2.3.1 Data collection

6.2.3.1.1 United States Environmental Protection Agency datasets

We obtained the data needed to attain our objective from the United States Environmental Protection Agency (USEPA). This agency initiated the monitoring of biological, chemical, physical and recreational characteristics of rivers, streams, lakes and reservoirs across the continental United States (USEPA, 2015a, 2015b). Because no macroinvertebrate richness data were available for reservoirs before impoundment in these datasets, we used a space-for-time substitution approach, which substitutes spatial data for unavailable temporal data (Pickett, 1989). Macroinvertebrate richness from free-flowing rivers and streams was taken as a proxy for before impoundment conditions, which we refer to as "reference richness" (see section 6.2.3.1.6 of methodology for details) and macroinvertebrate richness from reservoirs was taken as a proxy for reservoir occupation condition, which we refer to as "impacted richness".

6.2.3.1.2 Taxonomic resolutions

The most precise taxonomic resolution used by the USEPA to identify macroinvertebrates ranges from up to genus (82% of taxa), family (15% of taxa), order (2% of taxa), class (0.8% of taxa) and other coarser levels (0.4% of taxa). Given the predominance of the proportion of genera and families (97%; 82% of genera + 15% of families) in the USEPA dataset, it was decided to estimate PDF using two main levels of taxonomic resolution, namely genus and family. We also added a criterion of inclusiveness for each taxonomic resolution. For example, in this study, a taxonomic resolution is said to be inclusive when taxa that are not resolved up to a targeted taxonomic resolution level (*e.g.*, genus) are included as contributors to richness (*e.g.*, family, order, class, etc.). On the contrary, a taxonomic resolution is said to be exclusive when only taxa resolved up to a targeted taxonomic resolution level (*e.g.*, genus) are included as richness contributors (excluding taxa that are resolved at the family or coarser taxonomic resolutions). The same rationale applies to the family taxonomic resolution. This classification strategy resulted in four possible categories of taxonomic resolution: 1) genus-exclusive (only genus), 2) genus-inclusive (genus, family, order, class, etc.), 3) family-exclusive (only family) and 4) family-inclusive (family, order, class, etc.).

Even though they are not technically "taxonomic resolution" (not always specific to one resolution), we refer to the four categories as taxonomic resolutions from now on. For this study, richness resolved at the species level of taxonomic resolution was not available, see discussion for the implication of this limitation.

6.2.3.1.3 Macroinvertebrate richness

We used the USEPA 2008-2009 National Rivers and Streams Assessment (NRSA) as a reference for pre-impoundment and collected reference macroinvertebrate richness data (RICHNESS; resolved using the four previously identified categories), a unique identifier (UID), latitude (LAT), longitude (LON) and ecoregion (ECO) for 2062 rivers and streams (Figure 6.1; shown using white circles). We used the USEPA 2012 National Lakes Assessment (NLA) and collected impacted macroinvertebrate richness (RICHNESS) data, a unique identifier (UID), latitude (LAT), longitude (LON) and ecoregion (ECO), as well as a suite of environmental variables (spatial, physical, chemical and human; Table C.1), related to 134 reservoirs (Figure 6.1; shown using black circles).

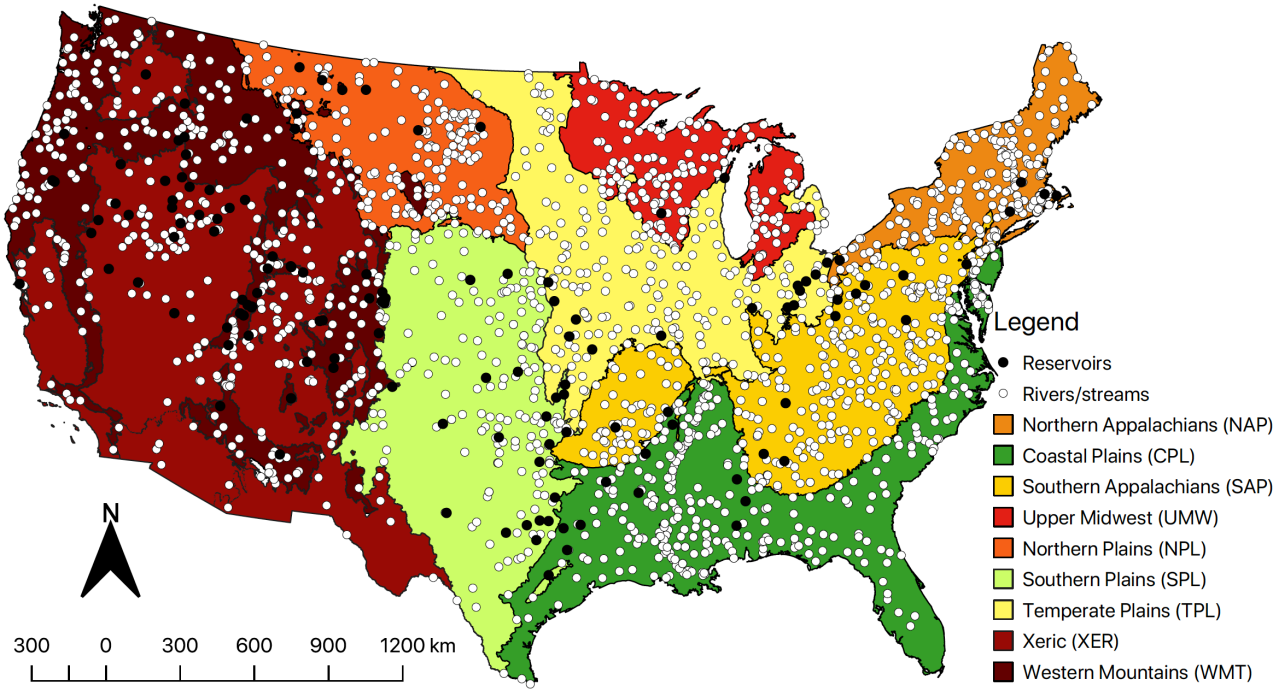

Figure 6.1 Map of the distribution of National River and Streams Assessment (NRSA) rivers and streams (n = 2062; white dots) and National Lake Assessment (NLA) reservoirs (n = 134; black dots) from the United States Environmental Protection Agency (USEPA), as well as the nine color-coded ecoregions. From Trottier et al. (2021) with permission of the authors and the publisher under the Creative Commons Attribution License.

6.2.3.1.4 Environmental variables

Through the NRSA dataset, we had access to summary environmental variables for rivers and streams, namely latitude, longitude and ecoregion. For reservoirs, there were a lot more environmental variables available and potentially useful to explain the observed variation in PDFs. Thus, we developed four environmental matrices, namely spatial, physical, chemical and human matrices, selecting a set of environmental variables potentially influencing macroinvertebrate richness, based on expert judgment, from the NLA dataset. See Table C.1 for a complete description of the variables included in each matrix. The spatial matrix included variables describing the location of the reservoirs; latitude, longitude, ecoregion, mean annual air temperature, mean annual precipitation and types of land covers (forested, cultivated pastures, wetlands, grasslands and shrubs, developed and water or barren). The physical matrix included variables describing the reservoir itself; reservoir area, elevation, shallow water habitat condition and riparian vegetation. The chemical matrix included variables that described the biochemical state of the reservoir; trophic state, water transparency (using a Secchi disk), dissolved organic carbon (DOC), total phosphorus level (TPL), water color, conductivity, total nitrogen level (TNL), pH, methylmercury and chlorophyll-α (Chl-α). The human matrix included variables describing the human activity, impact or influence around the reservoir shoreline; influence of buildings, commercial activities, crops, docks, landfills, lawns, parks, pastures, powerlines, roads, walls and others.

6.2.3.1.5 Ecoregions

Rivers and reservoirs were spread out across nine ecoregions, which was the result of a common aggregation of ecoregions adopted by both the NRSA and NLA datasets. Ecoregions were defined *a priori* by Omernick (1987) and Herlihy et al. (2008), where they used a classification based on similar environmental characteristics and macroinvertebrate assemblages. These nine ecoregions include: Coastal Plains (CPL), Northern Appalachians (NAP), Northern Plains (NPL), Southern Appalachians (SAP), Southern Plains (SPL) Temperate Plains (TPL), Upper Midwest (UMW), Western Mountains (WMT) and Xeric (XER; Figure 6.1).

6.2.3.1.6 PDF calculation

PDF was computed using only native riverine macroinvertebrate taxa and excluded all new taxa that would be encountered in a lake-like habitat (reservoir), as they would most likely not be present in a pre-reservoir occupation, river-like habitat. PDF was calculated as the difference in richness between river (reference; x) and reservoir richness (impacted; y), divided by river richness (reference; x). This general equation was applied to three spatial scales, the United States, nine ecoregions and 134 reservoirs, and repeated for each of the four previously identified categories of taxonomic resolution, namely genus-exclusive, genus-inclusive, family-exclusive and family-inclusive.

For the United States spatial scale (PDF_{USA}), we used the mean reference macroinvertebrate richness in rivers and streams for the whole country (one observation of richness per river or stream averaged over the United States; $\bar{x}_{USA}$; n = 2062) and the mean impacted macroinvertebrate richness in reservoirs for the whole United States (one observation of richness per reservoir averaged over the United States; $\bar{y}_{USA}$; n = 134) to obtain a United States specific change in macroinvertebrate richness, as per equation 6.1;

$$PDF_{USA} = \frac{\bar{x}_{USA} - \bar{y}_{USA}}{\bar{x}_{USA}} \tag{6.1}$$

For the ecoregion spatial scale (PDF_{ECO}), we used ecoregion mean reference macroinvertebrate richness of all rivers and streams (one observation of richness per river or stream averaged over each ecoregion; $\bar{x}_{ECO}$) and the ecoregion mean impacted macroinvertebrate richness in reservoirs (one observation of richness per reservoir averaged over each ecoregion; $\bar{y}_{ECO}$) to obtain an ecoregion specific change in macroinvertebrate richness, as per equation 6.2;

$$PDF_{ECO} = \frac{\bar{x}_{ECO} - \bar{y}_{ECO}}{\bar{x}_{ECO}} \tag{6.2}$$

For the reservoir spatial scale (PDF_{RES}), we assumed that macroinvertebrate richness in rivers and streams in the surrounding area of a reservoir from the NLA dataset would be comparable to what would have been found in a river prior to its transformation and occupation by a reservoir and could be used to derive PDF (see Trottier et al., [2021] for details). We used ecoregion mean reference macroinvertebrate richness of all rivers and streams (one observation of richness per river or stream averaged over each ecoregion; $\bar{x}_{ECO}$) and the impacted macroinvertebrate richness of a specific reservoir within the same ecoregion (one specific richness observation per reservoir, no averaging; y_{RES}) to obtain a reservoir specific change in macroinvertebrate richness, as per equation 6.3;

$$PDF_{RES} = \frac{\bar{x}_{ECO} - y_{RES}}{\bar{x}_{ECO}} \tag{6.3}$$

6.2.3.2 Data analysis, empirical model selection and cross-validation

6.2.3.2.1 ANOVA and regionalization

We used a one-way randomized-group analysis of variance (ANOVA) to determine whether there was a statistically significant difference between the nine PDF_{ECO} for each of the four taxonomic resolutions. If statistically significant, we pursued the analysis and identified which PDF_{ECO} were significantly different from each other based on their standardized mean difference and their Confidence Intervals (CI), as well as assessed differences or similarities in PDF_{ECO} rankings across taxonomic resolutions. All statistical analyses were done using R version 3.0.2 (R Core Team, 2017). We conducted the ANOVA with the *ind.oneway.second* function in the *rpsychi* R package version 0.8 (Okumura, 2012).

6.2.3.2.2 Empirical models variation partitioning and variables selection

To assess the influence of taxonomic resolution on empirical model variable selection, we performed four parallel empirical model variable selections, one for each of the four taxonomic resolutions, namely genus-exclusive, genus-inclusive, family-exclusive and family-inclusive.

We first performed variation partitioning for the four taxonomic resolutions. Using a variation partitioning analysis, which describes how a set of environmental matrices (*e.g.*, spatial, physical, chemical and human matrices; Table C.1) explains the shared variation observed in a response variable (*e.g.*, PDF_{RES}; Legendre, 2008), we identified if there were any differences in the variables significantly explaining the variation observed in PDF_{RES} between taxonomic resolutions. To achieve the most parsimonious analysis, we performed a stepwise selection procedure on each environmental matrix to identify which variables best explained the variation in PDF_{RES}. Variables selection was performed with the function *ordiR2step* and variation partitioning was conducted with the *varpart* function in the *vegan* R package version 2.5-2 (Oksanen et al., 2019).

Second, we used a multiple linear regression (*lm* function in the *stat* R package version 3.4.2; R Core Team, 2017) to build an empirical model specific to each taxonomic resolution. We used the variables identified by the variation partitioning analysis (which were specific to each of the four taxonomic resolutions) as the most influential environmental variables to explain the variation observed in PDF_{RES}. We checked whether assumptions associated with multiple linear regression were violated (dependency, normality, mean of zero and homoscedasticity of residuals) and deleted a few outliers. Model selection procedure was done with a common sample size of 78 reservoirs

for each of the four taxonomic resolutions. We applied a manual backward selection procedure, used the recommended information theoretic approach based on Akaike Information Criterion (AIC; Burnham & Anderson, 2002) and the Bayesian Information Criterion (BIC; Schwarz, 1978) to compare potential models within a taxonomic resolution (full model to null model), and selected the model with the highest support. For each environmental variable selected in final models, we extracted estimates and Standard Errors (SE), where the estimates represent the direction and magnitude of PDF_{RES}, specific to a taxonomic resolution.

6.2.3.2.3 Cross-validation and observed versus predicted plots

Cross-validation (CV) is an ensemble of methods used to measure the performance of a model on new data (Kassambara, 2018). In its simplest form, CV resides in setting aside a fraction of a complete dataset, the validation set (generally between 20 to 25%), building a model using the leftover fraction, the modeling set (generally between 75 to 80%) and testing its performance on the set aside fraction of the dataset (validation set; Kassambara, 2018). Briefly, if the model predictions are similar to the observed values from the validation set, the empirical model is considered robust (Kassambara, 2018). We combined cross-validation with the observed versus predicted (OP) plots approach. An OP plot assess the ability of an empirical model to predict PDF_{RES} in a reservoir. In these plots, PDF_{RES} observed ($PDF_{RES.OBS}$) is regressed against PDF_{RES} predicted ($PDF_{RES.PRED}$) and the intercept, the slope and the R^2_{ADJ} of the linear regression is analyzed to evaluate the prediction error of a model (Piñeiro et al., 2008). The intercept describes the model bias, the slope describes the model consistency, whereas the R^2_{ADJ} represents the proportion of the total variance that can be explained by the linear regression, that is how much of the linear variation in the observed data can be explained by the predicted data (Mesplé et al., 1996; Piñeiro et al., 2008; Smith & Rose, 1995). The regression slope is also compared to a 1:1 regression line (intercept = 0 and slope = 1), which translate perfect accuracy between the observed and the predicted values and allow the identification of overestimation versus underestimation zones of the predictive model. If the slope of the model is significantly different from 1, there is a difference between the observed and predicted values. If the intercept of the model is different from 0, there is a systematic and constant difference between the observed and predicted values. At last, we also computed a Root Mean Squared Error (RMSE), a prediction error metric that measures the prediction error between a real observation and a model prediction – lowest is best. Linear

regressions were performed using the *lm* function from the base package *stats (R Core Team, 2017)*.

6.2.3.2.4 Repeated k-fold cross-validation and generalization

Repeated k-fold Cross-validation (RKFCV) consists in assessing a model's predictive performance with different combinations of datasets splits (which samples are assigned to validation and modeling sets), and calculates an average RMSE for said model (Kassambara, 2018). Thus, it ensures the researcher that the results obtained through a specific random CV split are not only due to pure luck and that, using a different split of dataset would also lead to similar results. To assess this similarity (or dissimilarity), we compared the RMSE obtained through the initial CV split to the average RMSE obtained through RKFCV. If the former is within the error measure (standard deviation [SD] and CI) of the latter, generalizations can be made to the complete dataset with confidence, not only to the 20-25% of reservoirs from the validation set. The RKFCV procedure is as follows; 1) randomly split the modeling set into k-subsets – hereafter called folds, 2) set aside one fold, which will act as a validation set and run the model using the leftover folds, 3) validate the model using the set aside validation fold and record prediction error, 4) repeat this process until each fold has served for validation, 5) calculate the average of the k RMSE and 6) repeat this procedures x number of times, the final model RMSE is the mean RMSE of x repeats (Kassambara, 2018). As a rule of thumb, five or ten folds are suggested for k, which have been empirically shown to lead to neither high bias nor variance (Kassambara, 2018). We computed five folds – three repeats RKFCV on each of the four taxonomic resolution model that were used to make predictions (*i.e.*, model with highest support).

6.2.4 Results and discussion

6.2.4.1 Taxonomic resolution and regionalization

6.2.4.1.1 Taxonomic resolution did not influence the significance of regionalization

The ANOVA suggested a significant influence of regionalization on PDFs for all taxonomic resolutions (Figure 6.2). However, the influence of regionalization was not identical across taxonomic resolutions. There was a variation in PDF_{ECO} ranking position across taxonomic resolutions (Figure 6.2), especially between taxonomic resolutions that are not using the same level

of taxonomy (*i.e.*, genus-inclusive versus family-inclusive). In all taxonomic resolutions, $PDF_{ECO.CPL}$, $PDF_{ECO.SPL}$ and $PDF_{ECO.WMT}$ were ranked at the same position (see section 6.2.3.1.5 in the methodology for ecoregions' acronym definition). The most mobile PDF_{ECO}, in terms of rank, were $PDF_{ECO.TPL}$, $PDF_{ECO.SAP}$, $PDF_{ECO.NAP}$ and $PDF_{ECO.NPL}$. The latter are also considered more ambiguous in terms of statistical difference and do not have a definite ranking position across taxonomic resolutions. This may suggest that PDF_{ECO} ranking position in taxonomic resolutions sharing the same level of taxonomy (*i.e.*, genus-exclusive and genus-inclusive OR family-exclusive and family-inclusive) are more similar when compared to each other, especially for family-based taxonomic resolutions (Figure 6.2c-d). Even though we observed some minor changes in PDF_{ECO} ranking across taxonomic resolutions, these changes do not suggest that taxonomic resolution has a drastic impact regionalization and PDF_{ECO}.

6.2.4.1.2 Taxonomic resolution mildly influenced CFs' direction and magnitude

The direction and magnitude of some PDF_{ECO} were mildly influenced by taxonomic resolution. In terms of direction, only $PDF_{ECO.SPL}$ changed direction across taxonomic resolutions, where genus-based taxonomic resolutions $PDF_{ECO.SPL}$ were negative (gains in macroinvertebrate genera) and family-based taxonomic resolutions $PDF_{ECO.SPL}$ were positive (losses in macroinvertebrate families; Figures 6.2 and C.Sl, Table C.S1).

In terms of magnitude, taxonomic resolution did not significantly influence the magnitude in $PDF_{ECO.CPL}$, $PDF_{ECO.NAP}$, $PDF_{ECO.UMW}$, $PDF_{ECO.XER}$ and $PDF_{ECO.WMT}$ (Figures 6.2 and C.S1, Table C.S1), according to the F-statistic. Six out of nine (67%) PDF_{ECO} did not significantly differ from each other between genus-exclusive and genus-inclusive taxonomic resolutions (Figure C.S1). In general, PDF_{ECO} magnitudes at the genus-inclusive taxonomic resolution seemed to be higher than those at the genus-exclusive taxonomic resolution, but this observation is not statistically supported. Eight out of nine (89%) PDF_{ECO} did not significantly differ between family-exclusive and family-inclusive taxonomic resolutions (Figure C.S1). No trends could be identified for family-based taxonomic resolutions, family-exclusive and family-inclusive taxonomic resolutions were extremely similar in terms of magnitude. In general, PDF_{ECO} with similar taxonomic resolutions (*i.e.*, genus-exclusive OR family-exclusive) shared similar magnitudes. Taxonomic resolution did significantly influence magnitude in $PDF_{ECO.SPL}$, $PDF_{ECO.TPL}$, $PDF_{ECO.SAP}$ and $PDF_{ECO.NPL}$ (Figures 6.2 and C.S1, Table C.S1), according to the F-statistic. Across these four PDF_{ECO}, no common trend could be identified. The four PDF_{ECO} mostly influenced by taxonomic resolution did not have

especially low sample sizes and appeared to be closer in terms of climatic conditions – they are all located in the central part of the United States and three of them share a common landscape characteristic, namely the presence of plains. There is no apparent reason explaining why taxonomic resolution would have an influence in the presence of this specific type of landscape and it is most likely the result of a statistical artefact.[GT26] At this point, apart from these four PDF_{ECO}, where it was not possible to suggest one taxonomic resolution over the other, using any of the taxonomic resolutions would lead to similar PDF_{ECO}. In light of this information, we cannot draw clear conclusions about a potentially ideal taxonomic resolution.

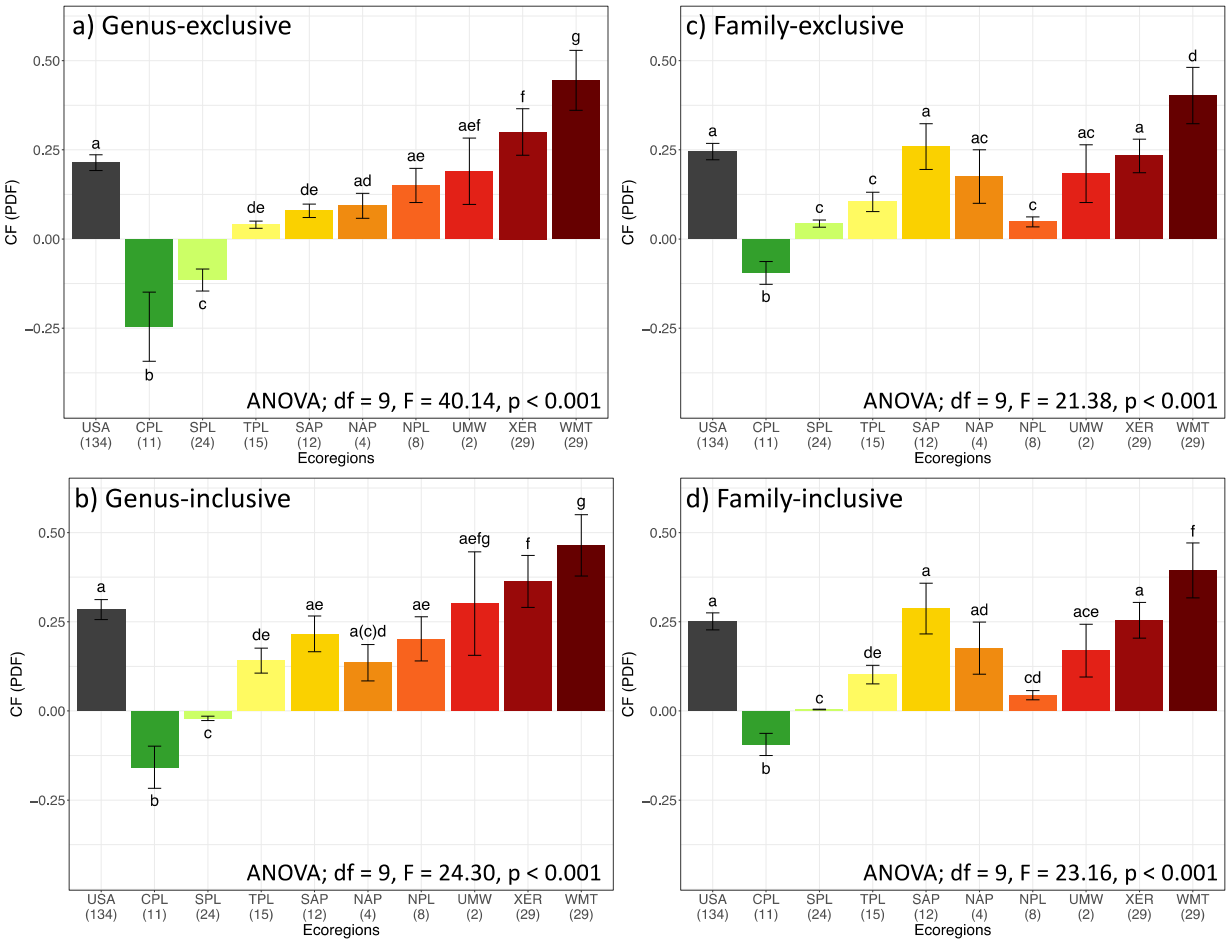

Figure 6.2 Barplot panels showing ecoregion specific Potentially Disappeared Fraction (PDF_{ECO}) with different taxonomic resolutions; genus-exclusive (a), genus-inclusive (b), family-exclusive (c) and family-inclusive (d). Ecoregions are color-coded based on PDF_{ECO} magnitudes (gains in taxa in green, to loss in taxa in red) observed at the genus-exclusive taxonomic resolution (reference) and ANOVA output for the influence of ecoregions on PDF_{ECO}, at four different levels of taxonomic resolution including the numerator (df_{num}) used to attribute F-statistics (a = 0.05), F-values obtained through statistical testing and significance level. If F-value is bigger than respective F-statistics, there is a difference in PDF_{ECO} at the specified level of taxonomic resolution. From Trottier et al. (2021) with permission of the authors and the publisher under the Creative Commons Attribution License.

6.2.4.1.3 Implications of the inclusiveness criterion

Despite the fact that we did not observe significant differences between exclusive and inclusive taxonomic resolutions,[GT27] we felt it was important to discuss the implications of the inclusiveness

criterion on the generation of CFs and their representativeness of the impacts to ecosystems. When favoring an exclusive taxonomic resolution (*e.g.*, genus only), the taxonomic resolution is homogeneous, very consistent and only carries little uncertainty, with respect to taxonomic identification. However, this type of taxonomic resolution most likely excludes some taxa, that were not identified up to the genus level, bringing a non-negligeable amount of information about the actual richness of an ecosystem. Thus, CFs generated with an exclusive taxonomic resolution, might not be as accurate and ecosystem impacts portrayed by the CFs might be biased. When an inclusive taxonomic resolution is favored, the richness depicted is a lot more faithful to the reality, which translates into more accurate CFs. The only downside is that the taxonomic resolution is not as coherent (multiple levels of resolution) and that there is considerable taxa identification uncertainty, since some taxa have not been identified up to most specific level taxonomic resolution. We also acknowledge that since it is easier to lose a specific genus than entire family, intuitively CFs resolved at the genus level of taxonomic resolution should demonstrate higher loss of taxa than CFs resolved at the family level of taxonomic resolution. However, in this study, because some taxa are not identified up to genus, it is possible to observe higher losses of taxa using CFs resolved at the family level of taxonomic resolution. [GT28]This is further highlighting the need to include as much data as possible in taxonomic resolution, that is favoring inclusive taxonomic resolution for a more faithful portrait of the ecosystem richness and sensitivity, and thus more accurate CFs.

6.2.4.2 Taxonomic resolution and empirical models

6.2.4.2.1 Variation partitioning was not influenced by taxonomic resolution

Ecoregions, elevation, area, trophic state and influence of lawns were environmental variables that explained significant amount of the total variation in PDF_{RES} across all taxonomic resolutions (Figure 6.3). Shallow water habitat condition, pH, influence of roads and parks explained additional variation for genus-based taxonomic resolutions, whereas it was solely the influence of powerlines that explained additional variation in both family-based taxonomic resolutions. At all taxonomic resolutions, the four environmental matrices (spatial, physical, chemical and human) explained a similar amount of the total variation in PDF_{RES} (Figure 6.3). At the genus-exclusive taxonomic resolution, ecoregion, elevation, area, trophic state, influence of lawns, macrophytes, pH and the influence of parks explained 47% of the total variation in PDF_{RES}. At the genus-inclusive

taxonomic resolution, ecoregion, elevation, area, trophic state, influence of lawns, pH and the influence of roads explained 51% of the total variation in PDF_{RES}. At the family-exclusive and family-inclusive taxonomic resolutions, ecoregion, elevation, area, trophic state, influence of lawns and powerlines respectively explained 43 and 41% of the total variation in PDF_{RES}. The additional fraction of the total variation explained in PDF_{RES} with genus-based taxonomic resolutions was normal considering the additional amount of resolution, but this difference was negligeable.

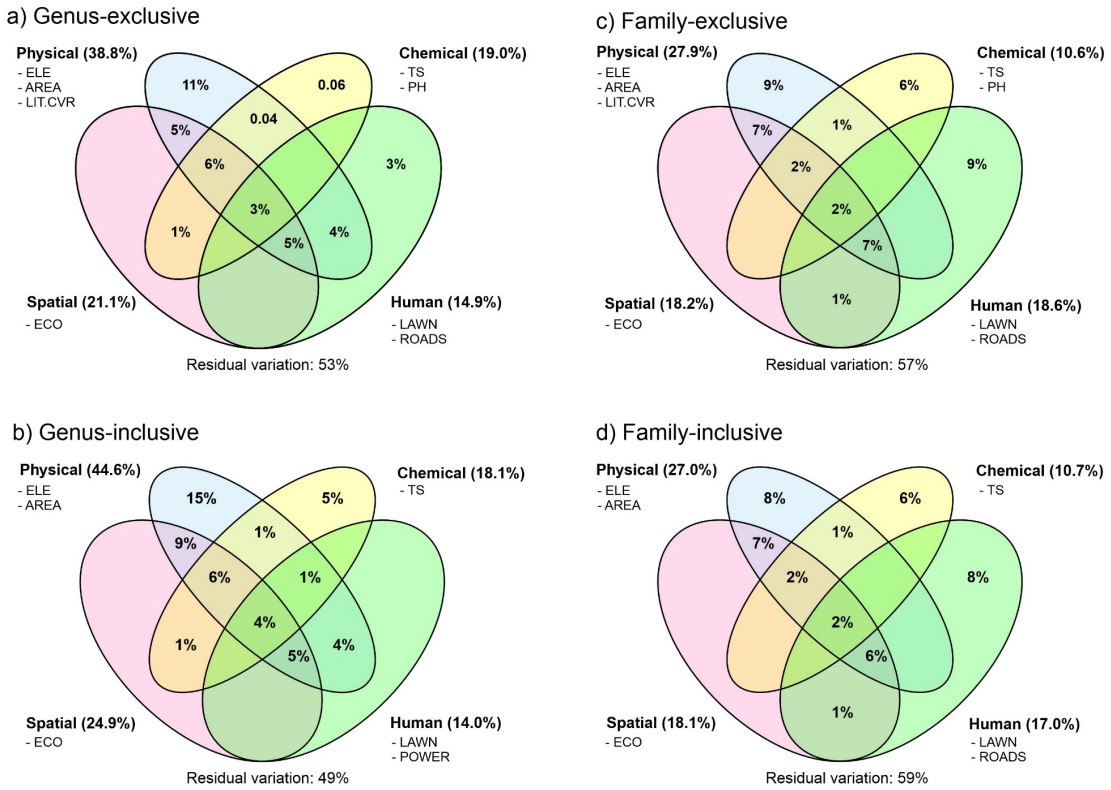

Figure 6.3 Venn diagram showing variation partitioning of a response matrix (Potentially Disappeared Fraction [PDF] of genera or families) explained by four matrices, that is spatial matrix, physical matrix, chemical matrix, and human matrix. The four panels represent the four categories of taxonomic resolution, where a) is genus-exclusive taxonomic resolution (47% of total variation in PDF_{RES} explained), b) is genus-inclusive taxonomic resolution (51% of total variation in PDF_{RES} explained), c) is family-exclusive taxonomic resolution (43% of total variation in PDF_{RES} explained) and d) is family-inclusive taxonomic resolution (; 41% of total variation in PDF_{RES} explained). ECO is for ecoregion, ELE is for elevation, AREA is for surface area, LIT.CVR is for shallow water habitat condition, TS is for trophic state, PH is pH level, LAWN is for the influence of lawn, ROADS is for the influence of roads and POWER is for the influence of powerlines. Values < 0 not shown.

Fractions of the total variation in PDF_{RES} explained by each matrix were similar between taxonomic resolutions sharing the same level of resolution. There was also minimal difference between genus- and family-based taxonomic resolutions. Physical and spatial matrices explained most of the total variation explained in PDF_{RES} across all taxonomic resolutions (Figure 6.3). Chemical and human matrices subsequently explained additional variation for genus-based taxonomic resolutions, whereas it was the other way around for family-based taxonomic resolutions (*i.e.*, subsequently human and chemical matrices). Variation partitioning seemed to be relatively unimpacted by taxonomic resolution.

6.2.4.2.2 Variables selection was influenced by taxonomic resolution

Taxonomic resolution influenced variables included in model with highest support but caused little variation in explanatory power. There was a differentiation in model selection between genus- and family-based taxonomic resolution. Models best explaining variation in PDF_{RES} were most similar between genus-based taxonomic resolutions (Table C.2, equations 6.4 and 6.5), as well as between family-based taxonomic resolutions (Table C.2, equations 6.6 and 6.7). Elevation and trophic state significantly explained variation in PDF_{RES} at all taxonomic resolutions. Area and pH explained additional variation at the genus-based taxonomic resolutions, whereas it was the influence of powerlines and lawns that explained additional variation at the family-based taxonomic resolutions (Table C.2).

$$PDF_{RES.GEN.EXC} = -1.159(\pm 0.413) + 0.011(\pm 0.002) \cdot \mathrm{sqrt}(ELE) + 0.134(\pm 0.046) \cdot \log_{10}(AREA) + 0.392(\pm 0.125) \cdot TS.OLI + 0.986(\pm 0.049) \cdot PH \quad (6.4)$$

$$PDF_{RES.GEN.INC} = -0.454(\pm 0.102) + 0.013(\pm 0.002) \cdot \mathrm{sqrt}(ELE) + 0.170(\pm 0.044) \cdot \log_{10}(AREA) + 0.325(\pm 0.115) \cdot TS.OLI \quad (6.5)$$

$$PDF_{RES.FAM.EXC} = 0.007(\pm 0.069) + 0.009(\pm 0.002) \cdot \mathrm{sqrt}(ELE) + 0.080(\pm 0.019) \cdot \log_{10}(POW + 1) + 0.348(\pm 0.118) \cdot TS.OLI - 0.059(\pm 0.028) \cdot \log_{10}(LAWN + 1) \quad (6.6)$$

$$PDF_{RES.FAM.INC} = 0.010(\pm 0.072) + 0.011(\pm 0.002) \cdot \mathrm{sqrt}(ELE) + 0.066(\pm 0.019) \cdot \log_{10}(POW + 1) + 0.381(\pm 0.123) \cdot TS.OLI \quad (6.7)$$

GEN and FAM abbreviations respectively means genus and family, whereas EXC and INC respectively means exclusive and inclusive. ELE stands for elevation, AREA for reservoir surface area, TS.OLI means an oligotrophic trophic state, POW is for the influence of powerlines and LAWN is for the influence of lawns.

Across all taxonomic resolutions, the model with highest support and best explaining variation in PDF_{RES} used a genus-inclusive taxonomic resolution (Model K: R^2_{adj} = 0.49, AIC=8, ΔAIC = 2, BIC = 20, p-value = 0.000; Table C.2). Models H, I and J (Table C.2), also at the genus-inclusive taxonomic resolution, were equally plausible and had high explanatory powers (R^2_{adj}). However, they contained marginally significant or non-significant environmental variables and were considered less parsimonious (*i.e.*, higher BIC). Thus, they were not chosen as best model, although they had lower ΔAIC.

6.2.4.2.3 Explanatory power and estimates were not influenced by taxonomic resolution

Variations in models' explanatory power were minimal across taxonomic resolutions. Genus-based models explained a little more variation in PDF_{RES} and family-based models explained a little less variation in PDF_{RES}. Estimates from variables present in more than one model had identical directionality and similar magnitude (when standardized; Table C.3), estimates did not statistically differ from one another (Figure C.S1). Although there were some differences in terms of variables and explanatory power, models with highest support across taxonomic resolutions were more similar than dissimilar, which once again did not point in any direction about which taxonomic resolution to favor.

6.2.4.3 Taxonomic resolution and prediction errors

6.2.4.3.1 Taxonomic resolution did not influence the ability of a predictive model

Previously identified models with highest support (section 6.2.4.2 of the discussion; $PDF_{RES.PRED}$) significantly predicted observed PDF_{RES} ($PDF_{RES.OBS}$) in the United States across all taxonomic resolutions (Figure 6.4). All observed versus predicted (OP) plots slopes were not significantly different from 1 (where observed values matched predicted values) and all intercepts were not significantly different from 0 (there were no model bias).

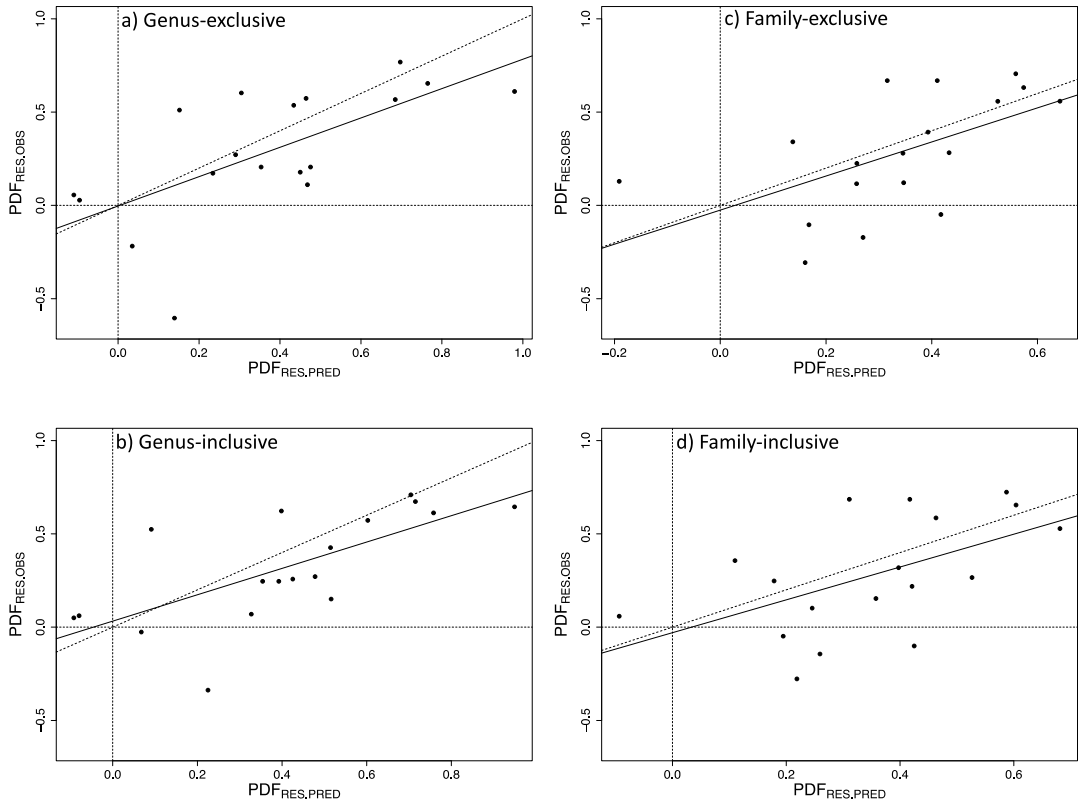

Figure 6.4 Observed versus predicted plot for each of the taxonomic resolution models with highest support. Top left if genus-exclusive (a), top right is family-exclusive (c), bottom left is genus-inclusive (b) and bottom right is family-inclusive (d). Dotted line is the 1:1 line, black solid line is the model slope. Models at all taxonomic resolutions can significantly predict observed PDF_{RES} within the United States.

6.2.4.3.2 Taxonomic resolution did not affect the prediction error of a model

For all taxonomic resolutions, the prediction error metric, RMSE, is quite high. At the spatial scale of the United States, regardless of taxonomic resolution, PDF_{RES} scored around 0.25, approximately 25% of macroinvertebrate taxa (genera or families) disappeared following reservoir transformation and occupation in the United States. Our RMSE values ranged from 0.24 to 0.28. Prediction errors were roughly the same magnitude as our values and thus, because RMSE values were expressed in the same units as the response variable (PDF_{RES}), the potential for error in prediction was high. This was further corroborated by the goodness-of-fit (R^2_{adj}) measures between observed and predicted values, which was not particularly high (25% to 44% of the variation in the $PDF_{RES.OBS}$ was explained by $PDF_{RES.PRED}$; Table 6.1). Thus, using genus- or family-based models

generated models that could statistically predict PDF_{RES}, but these models were not especially good at giving accurate predictions. These observations and conclusions are true for the validation set of 18 reservoirs. RKFCV can help evaluate if these outcomes generalize to the complete dataset. The RMSE and R^2_{ADJ} observed for each of the taxonomic resolution models were within the distribution of averaged RMSE and R^2_{ADJ} that were obtained from the RKFCV (Table 6.1; Figure 6.5). Thus, we can conclude that the validation exercise and its outcomes were not only true for the 18 reservoirs used for validation and that the conclusions generalized to the complete dataset of 78 reservoirs.

Table 6.1 Summary of models with highest support and observed versus predicted models, at each taxonomic resolution (genus-exclusive [D and D'], genus-inclusive [K and K'], family-exclusive [P and P'] and family-inclusive [W and W']). Akaike Information Criterion is ΔAIC, and Bayesian Information Criterion is BIC, PDF_{RES} stands for Potentially Disappeared Fraction (PDF) of genera or families. $PDF_{RES.OBS}$ is for observed PDF_{RES} from validation set, $PDF_{RES.PRED}$ is PDF_{RES} predicted for the modeling set, ELE for elevation, AREA for surface area, T.S. for trophic state, PH for pH level, POW for the influence of powerlines and LAWN for influence of lawns. For each model, the estimate for the intercept is labelled b_{int} and all other bs (b_{ELE}, b_{AREA}, $b_{T.S}$, b_{PH}, b_{POW} and b_{LAWN}), estimate for the slope of their respective variables. See Table C.1 for full description of the variables used and Table C.3 for variables estimates of models with highest support, Standard Errors (SE) and p-values. Delta AIC and BIC are calculated within each taxonomic resolution. RMSE stands for Root Mean Squared Error and RMSE.SD for its Standard Deviation (SD). MAE stands for Mean Absolute Error and MAE.SD for its SD. R^2_{ADJ} stands for the mean goodness-of-fit and R^2_{ADJ}.SD for its SD. N is the sample size, MODEL in the DATASET column stands for modeling set and VALID for validation set. *Marginally significant.

Models	Non-significant variables	AIC	BIC	R^2_{ADJ}	P-VALUE	RMSE	RMSE.SD	MAE	MAE.SD	R^2_{ADJ}	R^2_{ADJ}.SD	N	DATASET
(D) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH	PH*	15	28	0,44	0,000	0,28	0,04	0,23	0,04	0,41	0,19	60	MODEL
(D') $PDF_{RES.OBS} \sim b_{int} + b_{PDF.RES.PRED}$*$PDF_{RES.PRED}$	None	8	11	0,40	0,003	0,28						18	VALID
(K) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS	None	11	22	0,50	0,000	0,27	0,05	0,23	0,05	0,51	0,10	60	MODEL
(K') $PDF_{RES.OBS} \sim b_{int} + b_{PDF.RES.PRED}$*$PDF_{RES.PRED}$	None	0	3	0,44	0,002	0,24						18	VALID
(P) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS + b_{LAWN}*log10(LAWN)	TS* and LAWN*	10	22	0,45	0,000	0,26	0,03	0,22	0,03	0,42	0,12	60	MODEL
(P') $PDF_{RES.OBS} \sim b_{int} + b_{PDF.RES.PRED}$*$PDF_{RES.PRED}$	None	7	10	0,29	0,013	0,26						18	VALID
(W) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS	TS*	14	25	0,41	0,000	0,26	0,05	0,22	0,05	0,44	0,18	60	MODEL

(W') $PDF_{RES.OBS} \sim b_{int} + b_{PDF.RES.PRED}*PDF_{RES.PRED}$	None	8	11	0,25	0,020	0,27	18	VALID

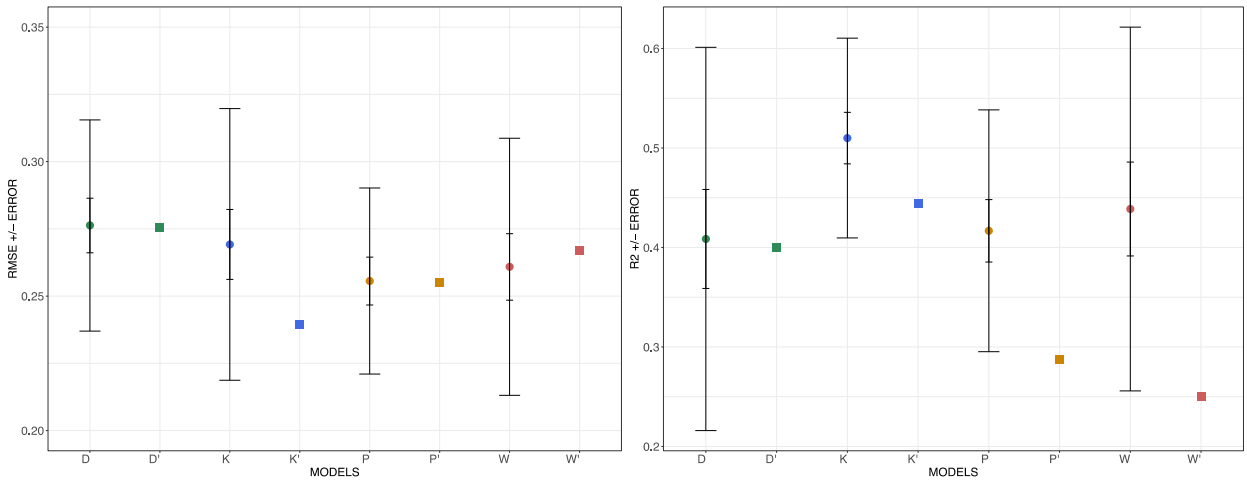

Figure 6.5 Plot showing the Root Mean Squared Error (RMSE) on the left and R^2_{ADJ} (R2) on the right, across taxonomic resolutions. Circle points represent average RMSE from the training part repeated k-fold cross-validation using modeling set (MODEL; n = 60) and square points represent the single RMSE from the testing part of the RKFCV using validation set (VALID; n = 18). Big error bars are Standard Deviations (SD) and small error bars are Confidence Intervals (CI), which were only available for the training part of the RKFCV. Model D and D' are genus-exclusive, K and K' are genus-inclusive, P and P' are family-exclusive and, W and W' are family-inclusive. Refer to Table 6.1 for further model identification and information.

6.2.4.3.3 Contextualization within the literature and earlier studies

To our knowledge, three studies touched basis with the notion of evaluating the impacts of environmental interventions without using species richness as a proxy for biodiversity in LCA. Lalande et al. (2013), were able to use a genetic fingerprinting technique (*i.e.*, denaturing gradient gel electrophoresis; DGGE) to estimate the biodiversity of soil microbial communities. The outputs of this study were more methodological and did not go as far as translating biodiversity estimates into CFs. Lessard et al. (2014) attempted to generate realistic toxicity thresholds (*i.e.*, EC_{50} values) using enzyme-based functional biodiversity index in contaminated soils. Similar to Lalande et al. (2013), they did not use their biodiversity index in CFs. Thus, both these studies tackled the issue of taxonomic resolution in a different way (*i.e.*, genetic fingerprinting and functional biodiversity) but none of them evaluated how these changes in biodiversity indices affected the generation of CFs neither did they compare how multiple types of biodiversity indices, or varying taxonomic

resolutions, could influence CFs generated using said indices or taxonomic resolutions. One study did propose to use functional diversity, instead of species richness, as an index to calculate CFs assessing land use impacts (de Souza et al., 2013), and showed that there was significant differences between CFs calculated with functional richness and species richness. However, assessing functional diversity requires species richness data and as well as organism specifics traits, which is even more challenging in terms if taxonomic identification than just species richness (de Souza et al., 2013). To our knowledge, with our study, it is the first time that the influence of coarser taxonomic resolutions (*i.e.*, genus and family) is evaluated in LCA.

6.2.5 Conclusion

In this study, we evaluated how taxonomic resolution influenced CFs assessing the impacts of reservoirs occupation on macroinvertebrate richness in the United States. Our results suggested that using either genus- or family-based taxonomic resolutions conveyed similar messages: [GT29] regionalization of CFs is important, the direction and magnitude of CFs are very similar among taxonomic resolutions, and empirical predictive models are also similar in terms of model selection and predictive abilities. If available, we suggest using a genus-based taxonomic resolution because this finer scale resolution explains additional variation in CFs and is considered more sensitive to the loss of taxa (*i.e.*, it is easier to lose a genus, than a whole family).

In terms of inclusiveness (exclusive versus inclusive taxonomic resolutions), we generally observed non-significant differences between genus-exclusive and genus-inclusive or family-exclusive and family-inclusive. Thus, including taxa that carry some ambiguity in terms of identification (*i.e.*, resolved up to the order, class or phylum or even family) does not seem to strongly bias CFs generation. In light of this information, we would advise LCA practitioners and modelers to use inclusive taxonomic resolutions, when available, as they are more representative of the richness of the community that is sampled, which translate into how potential impacts are portrayed within CFs.

There was not a lot of difference observed when comparing genus- and family-based taxonomic resolutions. Thus, it would be even more interesting to compare these two taxonomic resolutions to a species-based taxonomic resolution, which was unfortunately not available in the datasets used in this study. It would also be interesting to compare the influence of these taxonomic resolution at different spatial scales given that, especially for macroinvertebrates, it is recommended that genus-

or family-based taxonomic resolutions be used for large spatial scale bioassessments (*i.e.*, across a country) and that species-based taxonomic resolution may be more appropriate for smaller scale bioassessments (*i.e.*, within a single river; Jones, 2008; Marchant et al., 1995; Marshall et al., 2006; Verdonschot, 2006; Waite et al., 2004).

6.2.6 Acknowledgments

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CHAPTER 7 GENERAL DISCUSSION[GT30]

7.1 Achievement of research objectives

This thesis aimed at assessing the impacts of hydropower, dams and reservoirs on macroinvertebrate richness, both as a general portrait and within the LCA framework. This chapter provides a general discussion on how three specific research objectives helped achieve the main objective of this doctoral thesis. It also identifies contributions, limitations, recommendations and potential future research.

7.1.1 Objective 1: Literature review and meta-analysis on the global impacts of hydropower, dams and reservoirs on macroinvertebrate richness

Presented in chapter 4 (article 1), the literature review and meta-analysis allowed to further our understanding of the global impacts of hydropower, dams and reservoirs on macroinvertebrate richness. Our results showed a wide range of individual study responses (effect sizes) of macroinvertebrate richness to hydropower impacts, both in terms of direction (positive or negative impact) and magnitude. Indeed, the majority of the studies showed effect sizes that significantly differed from 0, whereas a minority of studies showed effect sizes that did not significantly differ from 0. This large variability of responses prevented the meta-analysis, which aims to average all studies' effect sizes into one mean effect size, from showing a statistically significant global impact of hydropower, dams and reservoirs on macroinvertebrate richness (mean effect size). Lots of environmental and methodological noise might have hindered the detection of a statistically significant impact, which is why we also investigated the influence of variables such as biomes, impact type, study design, as well as sampling season and gear. None of these variables showed a statistically significant effect. Thus, we suggested that we may not have targeted the appropriate variables influencing richness of macroinvertebrates and that these organisms may be regulated by much finer-scale processes associated with their immediate environment, such as granulometry, temperature, wave disturbance and macrophytes.

7.1.1.1 Contributions and limitations

In this first article, the scope of the outcomes is global and variables analyzed, with regards to their influence on macroinvertebrate richness and the impacts of hydropower, dams and reservoirs are macro- and meso-scaled. Even if the outcomes are not as clear as we would have hoped, they contribute to both the biology and LCA communities, each in their own way. For the biology community, although we haven't reached a clear consensus (not statistically significant mean effect size), the meta-analysis allowed us to observe that a majority of studies did show statistically significant impacts of hydropower, dams and reservoirs that had a negative effect on macroinvertebrate richness. Despite the lack of statistical significance of the meta-analysis, it helped us draw a nuanced portrait of the impacts of hydropower, dams and reservoirs on macroinvertebrate richness. These observations could eventually join forces with actual conclusions about the impacts of hydropower on other groups of organisms (*e.g.*, fish; Turgeon et al., 2019) and further our comprehension of the impacts of hydropower, dams and reservoirs on freshwater biodiversity in general. Our analysis suggested that macro-scale variables did not have a significant influence on the detection of hydropower impacts on macroinvertebrate richness and that we may have to look into more micro-scale variables in order to understand how environmental variables influences both the richness of macroinvertebrates and the detection of hydropower impacts.[GT31] For the LCA community, even though the main outcome of the meta-analysis is that there is no significant impact of hydropower, dams and reservoirs, because we observed that a majority of studies did indeed show significant impacts of hydropower, dams and reservoir on macroinvertebrate richness, we think it is important to think about integrating these impacts in LCA, namely in the form of CFs. This first thesis article sets the table for the following articles and gives some pointers regarding what type of environmental variables (*i.e.*, micro-scale) should be included to adequately represent the impacts of hydropower, dams and reservoirs on macroinvertebrate richness.

We believe that the main limitation in this study was the difficulty to find suitable studies that provided the required data to include into the meta-analysis. In many cases, studies that could have been included were excluded due to a lack of access to raw data, which prevented us from obtaining richness measures and their associated error. The relatively small number of studies included (*i.e.*, 17 studies) and the inability to control for methodological and environmental noise with the

available variables most likely prevented this meta-analysis from showing a statistically significant conclusion.

7.1.1.2 Recommendations and future research

As no statistically significant conclusions could be drawn from this meta-analysis, we strongly recommend enriching this meta-analysis with new studies that fit the criteria as they become available. With a larger sample size and a larger range of environmental variables (macro- to micro-scale), we might be able to identify which environmental variables (and what scale) influence macroinvertebrate richness and how it is impacted by hydropower, dams and reservoirs. We suggest that macro- and meso-scale environmental variables like biomes, type of study, type of impact, sampling season and gear continue to be collected, and that additional micro-scale environmental variables (*e.g.*, granulometry, temperature, wave disturbance and macrophytes) be added to the lot. Additionally, we recommend that studies be more transparent about the calculation of richness, error measure and sample size used, as well as share their raw datasets when possible. Accessing raw datasets could allow the calculation of macroinvertebrate richness, even if it was not the primary purpose of a study. This could help including more studies in the meta-analysis and potentially lead to more impactful conclusions (*i.e.*, statistically significant results). One of these recommendations was addressed in the subsequent chapter (5; article 2), that is to collect micro-scale environmental variables. Indeed, in the context of the second article, we had access to environmental variables at a finer scale, the scale singular reservoirs, and used them to explain how macroinvertebrate richness, impacted by reservoir occupation, varied with these variables.

7.1.2 Objective 2: Regionalized CFs and empirical model assessing the impacts of reservoir occupation on macroinvertebrate richness at three spatial scales

Presented in chapter 5 (article 2), the CFs and empirical model developed allowed the assessment of the impacts of reservoir occupation on macroinvertebrate richness at three spatial scales; reservoirs, ecoregions and the United States, and the possibility to explain the variability observed in reservoir specific CF with only a few reservoir-related, meso- and micro-scale environmental [GT32]variables. In this article, we observed a significant loss of macroinvertebrate richness of 28% following reservoir occupation at the United States spatial scale (*i.e.*, country). We also observed

the presence of a longitudinal gradient of impact (CFs), from lowest in the east to highest in the west and identified a few key environmental variables, namely elevation, trophic state and reservoir surface area, that explained a non-negligeable amount of the variation observed in reservoir-specific CFs. Large oligotrophic reservoirs at high elevation had higher CFs. Our results provided evidences regarding significant impacts of reservoir occupation on macroinvertebrate richness and a strong support for regionalized CFs. We showed that CFs were uniform and consistent across scales (*i.e.*, reservoir, ecoregion and country scales) and offered a simple empirical model, based on reservoir elevation, trophic state and surface area, to be used by LCA modelers and practitioners to estimate CFs where macroinvertebrate richness data is not available.

7.1.2.1 Contributions and limitations

In this second article, the scope of the outcomes is more targeted than in the first article, namely either at the scale of the country, ecoregions or reservoirs, and environmental variables analyzed, with regards to their influence on macroinvertebrates, are also at a smaller scale than previously (*i.e.*, meso- and micro-scale). The main contribution of this article is mostly directed towards LCA modellers and practitioners. It offers new, empirical-based CFs assessing the impacts of reservoir occupation on macroinvertebrate richness, that can be used for either the whole United States or nine ecoregions, as well as a model to estimate CFs specific to reservoirs. With these macroinvertebrate-based CFs, we could eventually combine them with fish-based CFs from Turgeon et al. (2021) and obtain more robust and representative CFs that integrate the impacts of reservoir occupation on more than one group of organisms, namely macroinvertebrates and fish. Another advantage of our CFs is that they were generated with empirical data, not theoretical curves assessing changes in richness based on water discharge (*i.e.*, Species-discharge Relationships; SDRs) or area (*i.e.*, Species-area Relationships; SARs). Because theoretical curves reflect ecological and evolutionary outcomes from ecosystems that are considered in equilibrium with natural discharge, it is risky to use these curves to assess species richness in impacted reservoirs where the discharge is likely not considered natural or in equilibrium. Thus, empirical-based CFs are more representative of the biological reality in a reservoir impacted environment.

The major limitation of this article is that the usage of CFs is restrained to the United States and thus, because of the global nature of supply chains, our CFs are not yet usable in LCA. However, our CFs are paving the way for future robust CFs integrating multiple groups of organisms and

applicable for the whole world, not just the United States. Maybe in few years, we will be able to accurately assess products, processes and services that encounter reservoir occupation in their life cycle (whether it is through hydropower, irrigation or recreation), especially with regards to the impact that reservoir occupation has on the ecosystem quality AoP, which was, just until recently (Turgeon et al., 2021), considered null. Some methodological limitations also come with the generation of these CFs. Gains of lentic taxa were not included in richness counts. Thus, our CFs are considered conservative and practitioners should keep this in mind when using these CFs. One last important limitation is that CFs were computed using genus richness because macroinvertebrate data resolved at the species level of taxonomic resolution was not available for both USEPA datasets. In LCA, CFs for the ecosystem quality AoP are predominantly calculated in terms of Potentially Disappeared Fraction of species ($PDF \cdot m^2 \cdot yr/m^2 \cdot yr$, which represents the fraction of species that disappeared following an impact for given occupied surface area). However, in this study, we use this framework and applied it to macroinvertebrate richness data resolved at the genus level of taxonomic resolution, not species. On one side, because we are working with proportions, we are still able to measure an impact on richness, just not in terms of species. On the other hand, it would be difficult to merge our macroinvertebrate CFs in PDF of macroinvertebrate genera to CFs in PDF of fish species and generate multi-groups CFs, because the two CFs are not using the same taxonomic resolution (a genus is comprised of one or multiple species and thus, losing a genus implies losing most likely more than one species).

7.1.2.2 Recommendations and future research

The empirical-based CFs developed in this research work are valid within a specific geographical context, that is assessing the impacts of reservoir occupation on macroinvertebrate richness in the United States. The validity of the CFs was not tested outside of this geographical context. As future research, it would be particularly interesting to see if the CFs issued from this study could be used outside the United States. A good first attempt to test the robustness of the CFs would be to use them in Canada, as many ecoregions that were in the United States frame of work did cross over the geographical border. Provided that one would access macroinvertebrate richness data across Canada, resolved at the genus level of taxonomic resolution, it would be possible to test if CFs are significantly different from CFs generated for the United States and if the empirical model provided in this article does give good estimated CFs in Canadian reservoirs. Finally, based on the last identified limitation, we strongly recommend assessing the influence of different taxonomic

resolutions on CFs generation. Subsequently, we may be able to advise that our macroinvertebrate-based CFs be coupled with fish-based CFs from Turgeon et al. (2021), provided that there is no significant influence, to generate multi-groups CFs that would be more representative of the ecosystem biodiversity. This taxonomic resolution issue gave rise to the following chapter (6; article 3).

7.1.3 Objective 3: Influence of taxonomic resolution on CFs assessing the impacts of reservoir occupation on macroinvertebrate richness

Presented in chapter 6 (article 3), the analysis of the influence of taxonomic resolution on CFs suggested that using genus- or family-based taxonomic resolutions lead to similar conclusions; 1) regionalization of CFs is statistically significant using all taxonomic resolutions, meaning that there is a spatial differentiation in CFs for all taxonomic resolutions, 2) the directionality and magnitude of CFs are similar among taxonomic resolutions, and 3) model selection and predictive abilities of the developed empirical models are also very similar among taxonomic resolutions. We also assessed the influence of an exclusive taxonomic resolution versus an inclusive taxonomic resolution. In an exclusive taxonomic resolution, taxa that are not resolved up to a targeted level of taxonomic resolution are not included as contributors to richness (*e.g.*, only genus for a taxonomic resolution up to genus), whereas in an inclusive taxonomic resolution, taxa that are not resolved up to a targeted level of taxonomic resolution are included as contributors to richness (*e.g.*, genus, family, order and class for a taxonomic resolution up to genus). There was generally no significant difference in CFs between exclusive and inclusive taxonomic resolutions. This suggests that inclusive taxonomic resolutions, where some taxa are more ambiguous in terms of identification, do not seem to bias CFs generation and thus, should definitely be included in richness counts. Indeed, these ambiguous taxa bring a non-negligeable amount of information regarding the richness of an ecosystem and would generate CFs that portray the impacts of reservoir occupation on macroinvertebrate richness more accurately.

7.1.3.1 Contributions and limitations

For the first time, our analyses and results provided the LCA community with some groundwork regarding the influence of taxonomic resolution on CFs. When computing CFs, specifically in the form of PDF, LCA practitioners can either use genus- or family-based taxonomic resolution

without significantly affecting their results, this is the major contribution stemming from this article. The other contribution is that we can now also say that the noise introduced from an inclusive taxonomic resolution is not skewing the computation of CFs. While those contributions are on the theoretical and methodological side of research, they have a clear practical output that sets a precedent in the scientific community: CFs computed with different taxonomic resolutions lead to similar results. However, there are some limitations associated with this research. First, we still do not know if CFs computed with genus- or family-based taxonomic resolutions are statistically similar to CFs computed with species-based taxonomic resolution, which is the predominantly used taxonomic resolution when it comes to CF calculated in PDF. Our data did not allow for such an analysis. Second, the contributions of the article are only true for macroinvertebrates and should not yet be applied to other groups of organisms, and only valid in the United States.

7.1.3.2 Recommendations and future research

Based on the results from this last article, we recommend that LCA modellers use genus-based, as opposed to family-based, taxonomic resolution since additional variation in CFs is explained by this finer scale of resolution. We also advise that inclusive taxonomic resolution should be favored as they are more representative of the taxa community inhabiting ecosystems and contributing to their biodiversity. As identified in the last paragraph, we strongly suggest that future research be conducted to evaluate the influence of finer (*i.e.*, species) and coarser (*i.e.*, order) taxonomic resolutions on CFs. More specifically, we are interested in the comparison of species-based taxonomic resolution with genus- and family-based taxonomic resolutions. Finally, we would also recommend that this type of methodological work be conducted using other groups of organisms and outside the United States. Testing the validity of this article's outcomes on zooplankton, fish, plants or vertebrates in Canada would constitute a good follow-up to this study.

7.2 Three articles, one thesis

The three articles are intimately linked and contributed to the wholesomeness of this thesis. In the first article, even if the global outcome of the meta-analysis was not statistically significant, the analysis allowed the observation a majority of studies were the impacts of hydropower, dams and reservoirs on macroinvertebrate richness were statistically significant. Within this majority, most of the studies showed significant negative impacts of hydropower, dams and reservoirs, few

showed positive impacts. Despite the non-significant meta-analysis, which was most likely the results of a wide range of significant effect sizes and not a majority of non-significant effect sizes, the outcomes of this article justified the need to work on integrating the impacts of hydropower, dams and reservoirs on macroinvertebrate richness through CFs, which directly led to the second article. The first article also allowed the identification of a potentially more accurate scale of environmental variables (*i.e.*, micro-scale versus macro-scale), which is an observation that was also taken into account in the second article. One assumption that was made in this second article, which also constituted a limitation of this work, is that we used macroinvertebrate genera richness instead on macroinvertebrate species richness to generate CFs in PDF. Because PDF are predominantly calculated with species richness in the LCA community, the implications of our CFs are limited, especially with regards to combining them with other CFs (based on species richness) to generate multi-groups CFs that are more representative of the impacts to the whole ecosystem. This assumption/limitation brought us to question the influence of taxonomic resolution (species, genera and families) on the generation of CFs. This became the main objective in the third article. Given that richness data resolved up to the species level of taxonomic resolution is not always available, especially for larger scale bioassessment and macroinvertebrates, being able to generate CFs with genus or family richness would unfold the number of CFs and facilitate the calculation of multi-groups CFs. In this third article, we came out with the conclusion that using either genus- or family-based taxonomic resolutions generated CFs (assessing the impacts of reservoir occupation on macroinvertebrate richness) that were not significantly different from each other. Moreover, results also suggested that inclusive taxonomic resolutions, despite being less coherent and more ambiguous in terms of identification, should be privileged over exclusive taxonomic resolution because they portray a more faithful representation of the ecosystem biodiversity under study. Given the choice, LCA practitioners should favor a genus-inclusive taxonomic resolution over family-based and/or exclusive taxonomic resolutions to assess macroinvertebrate richness and subsequently CFs. This article outcomes partially validated the choice that was made regarding the use of a genus-inclusive taxonomic resolution to assess macroinvertebrate richness and generate CFs in the second article. However, because we did not have access to macroinvertebrate richness data resolved at the species level of taxonomic resolution, it still remains unclear if CFs generated with genus or family richness are similar to CFs generated with species richness.

CHAPTER 8 CONCLUSION AND RECOMMENDATIONS

The heart of this thesis resides in three scientific articles. The first article was oriented towards the biological community, while also serving the LCA community. It provided a global portrait of the impacts of hydropower, dams and reservoirs on macroinvertebrate richness. The second article proposed practical impact indicators (CFs) that are aimed at the LCA community. It demonstrated that it is possible to generate empirical CFs, the need to develop regionalized CFs and provided a simple empirical model to estimate a CF specific to singular American reservoirs based on few micro-scale environmental variables. The last article brought in a research study that is considered more theoretical and methodological in its nature. It suggested that taxonomic resolution (genus- and family-based), did not influence CFs, whether it be in terms of CF regionalization, directionality, magnitude, model selection and predictive abilities, and that inclusive taxonomic resolutions should be favored over exclusive taxonomic resolutions.

We consider our work both fundamental and practical, as well as original and novel. CFs proposed in the second article contribute to the originality of this doctoral research as choosing macroinvertebrates as research taxa has almost never been done. Most studies used fish as research organism while macroinvertebrates have only been used once to quantify the impacts of river water consumption on biodiversity in LCA (Tendall et al., 2014). The third article brings in novelty to this doctoral research. To our knowledge, evaluating the influence of taxonomic resolution in CFs had never been done before in LCA.

This doctoral research's main attributes are its multi-disciplinarity and versatility. It combined biological data into an impact assessment tool aimed for decision making in the context of environmental sustainability of products, processes and services. It provided the LCA community with multi-scale CFs assessing the impact of reservoir occupation on macroinvertebrate richness across the United States. Eventually, when the scope of these CFs transcends macroinvertebrates in the United States (*i.e.*, multi-group and worldwide), they can be integrated in LCA methodologies (*i.e.*, IMPACT World+), where they will greatly improve how the impacts of reservoir occupation on biodiversity are accounted for within those methodologies, as well as providing a more faithful portrait of the impacts associated with products, processes and services that encompass reservoir occupation (*i.e.*, hydropower) in their life cycle.

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APPENDIX A ARTICLE 1

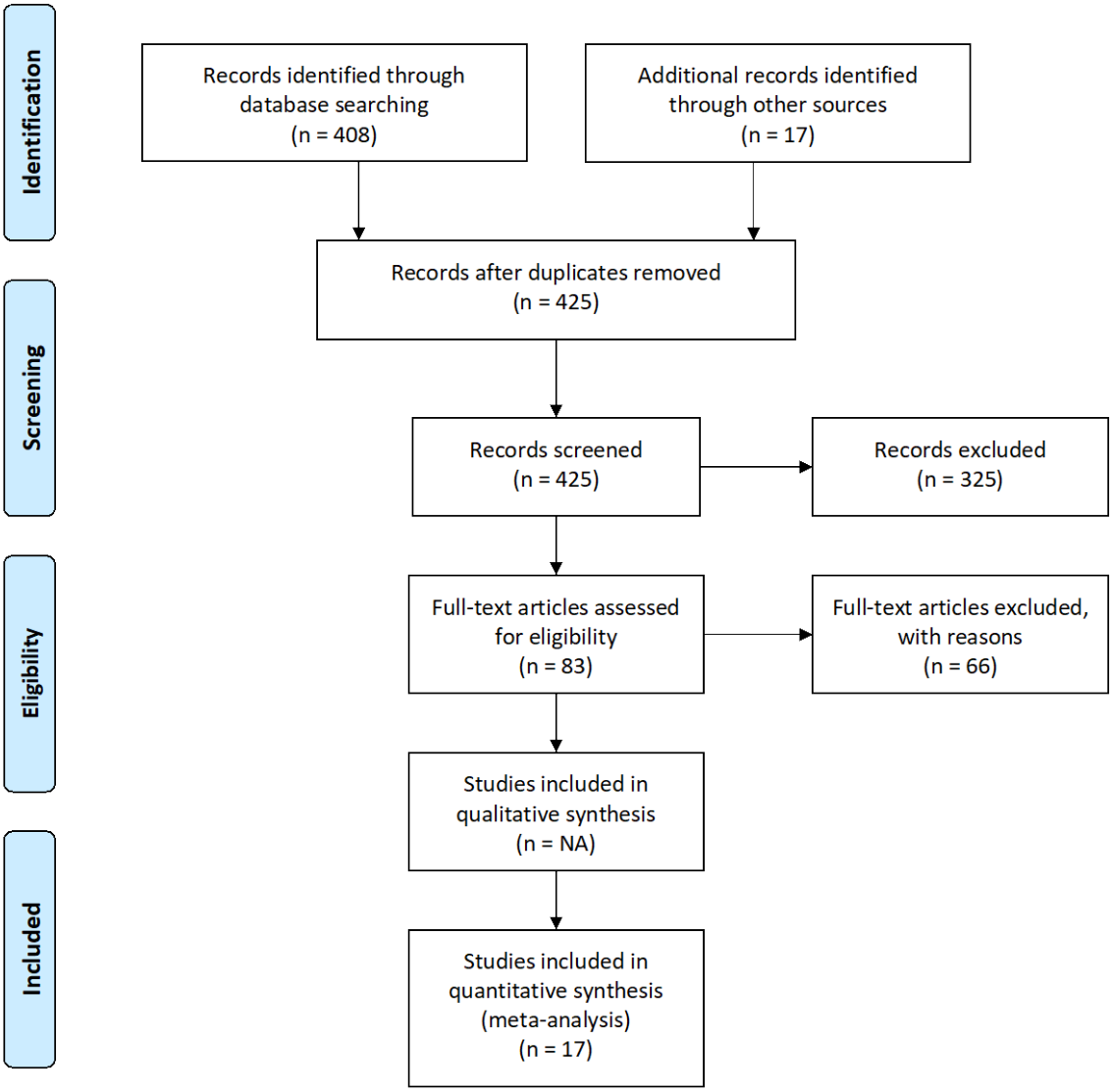

Fig A.1 PRISMA flow diagram for this meta-analysis from Moher et al. (2009).

Table A.1 PRISMA checklist for this meta-analysis, from Moher et al. (2009).

Section/topic	#	Checklist item	Reported on page #
TITLE			
Title	1	Identify the report as a systematic review, meta-analysis, or both.	1
ABSTRACT			
Structured summary	2	Provide a structured summary including, as applicable: background; objectives; data sources; study eligibility criteria, participants, and interventions; study appraisal and synthesis methods; results; limitations; conclusions and implications of key findings; systematic review registration number.	2
INTRODUCTION			
Rationale	3	Describe the rationale for the review in the context of what is already known.	3-6
Objectives	4	Provide an explicit statement of questions being addressed with reference to participants, interventions, comparisons, outcomes, and study design (PICOS).	4, 6
METHODS			
Protocol and registration	5	Indicate if a review protocol exists, if and where it can be accessed (e.g., Web address), and, if available, provide registration information including registration number.	7-8
Eligibility criteria	6	Specify study characteristics (e.g., PICOS, length of follow-up) and report characteristics (e.g., years considered, language, publication status) used as criteria for eligibility, giving rationale.	8-9
Information sources	7	Describe all information sources (e.g., databases with dates of coverage, contact with study authors to identify additional studies) in the search and date last searched.	8
Search	8	Present full electronic search strategy for at least one database, including any limits used, such that it could be repeated.	8
Study selection	9	State the process for selecting studies (i.e., screening, eligibility, included in systematic review, and, if applicable, included in the meta-analysis).	8-9
Data collection process	10	Describe method of data extraction from reports (e.g., piloted forms, independently, in duplicate) and any processes for obtaining and confirming data from investigators.	9-10
Data items	11	List and define all variables for which data were sought (e.g., PICOS, funding sources) and any assumptions and simplifications made.	9-10
Risk of bias in individual studies	12	Describe methods used for assessing risk of bias of individual studies (including specification of whether this was done at the study or outcome level), and how this information is to be used in any data synthesis.	NA
Summary measures	13	State the principal summary measures (e.g., risk ratio, difference in means).	10
Synthesis of results	14	Describe the methods of handling data and combining results of studies, if done, including measures of consistency (e.g., I^2) for each meta-analysis.	12

Table A.1 (continued) PRISMA checklist for this meta-analysis, from Moher et al. (2009).

Section/topic	#	Checklist item	Reported on page #
Risk of bias across studies	15	Specify any assessment of risk of bias that may affect the cumulative evidence (e.g., publication bias, selective reporting within studies).	12
Additional analyses	16	Describe methods of additional analyses (e.g., sensitivity or subgroup analyses, meta-regression), if done, indicating which were pre-specified.	14
RESULTS			
Study selection	17	Give numbers of studies screened, assessed for eligibility, and included in the review, with reasons for exclusions at each stage, ideally with a flow diagram.	15
Study characteristics	18	For each study, present characteristics for which data were extracted (e.g., study size, PICOS, follow-up period) and provide the citations.	15, 33
Risk of bias within studies	19	Present data on risk of bias of each study and, if available, any outcome level assessment (see item 12).	NA
Results of individual studies	20	For all outcomes considered (benefits or harms), present, for each study: (a) simple summary data for each intervention group (b) effect estimates and confidence intervals, ideally with a forest plot.	16
Synthesis of results	21	Present results of each meta-analysis done, including confidence intervals and measures of consistency.	15
Risk of bias across studies	22	Present results of any assessment of risk of bias across studies (see Item 15).	14
Additional analysis	23	Give results of additional analyses, if done (e.g., sensitivity or subgroup analyses, meta-regression [see Item 16]).	15-16
DISCUSSION			
Summary of evidence	24	Summarize the main findings including the strength of evidence for each main outcome; consider their relevance to key groups (e.g., healthcare providers, users, and policy makers).	17-18
Limitations	25	Discuss limitations at study and outcome level (e.g., risk of bias), and at review-level (e.g., incomplete retrieval of identified research, reporting bias).	18-22
Conclusions	26	Provide a general interpretation of the results in the context of other evidence, and implications for future research.	23
FUNDING			
Funding	27	Describe sources of funding for the systematic review and other support (e.g., supply of data); role of funders for the systematic review.	24

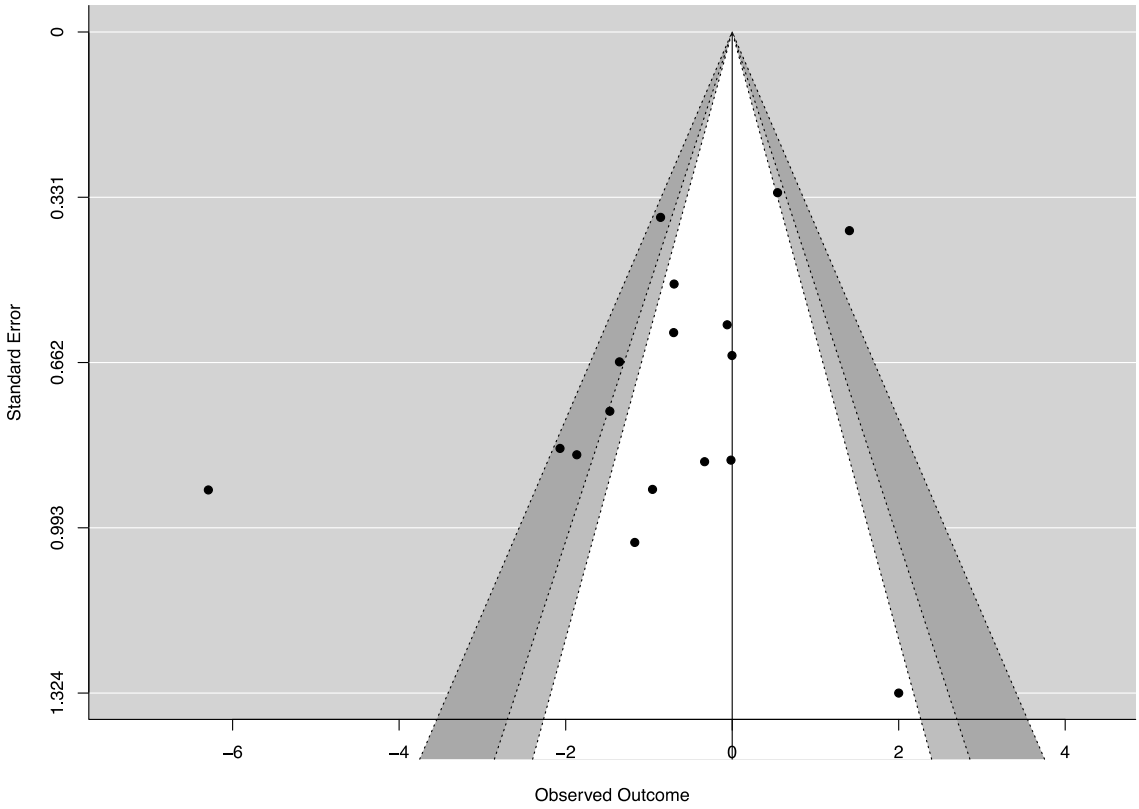

Fig A.2 Funnel plot for this meta-analysis, where no statistically significant asymmetry is observed (z = -1.0707, p = 0.2843).

Table A.2 Metadata table (1 of 3) showing all variables for each study included in this meta-analysis (Aroviita & Hämäläinen, 2008; Bruno et al., 2019; Englund & Malmqvist, 1996; Jackson et al., 2007; Kraft, 1988; Kullasoot et al., 2017; Marchetti et al., 2011; Mellado-Dìaz et al., 2019; Milner et al., 2019; Molozzi et al., 2013; Schneider & Petrin, 2017; Smokorowski et al., 2011; Steel et al., 2018; Takao et al., 2008; Valdovinos et al., 2007; White et al., 2011). Use Table A.3 as a companion table to get more information on each variable.

UID	AUTHORS	YEAR	OBS	BIOME	IMP	STUDY	SEASON	GEAR	REF.MEAN	REF.SD	REF.N	IMP.MEAN	IMP.SD	IMP.N	YI	VI	WI	CI.LOW	CI.UP	NOTES
6	Aroviita and Hamalainen	2008	1	BOR	WLF	NAT.IMP	FALL	NET	35,83	4,22	6	27,80	8,64	5	-1,118	0,423	0,054	-1,542	-0,695	UPPER.LITTORAL.VS.LESS.THAN.3M.REG.AMP
	Aroviita and Hamalainen	2008	2	BOR	WLF	NAT.IMP	FALL	NET	35,83	4,22	6	24,20	3,90	5	-2,604	0,675	0,054	-3,279	-1,929	UPPER.LITTORAL.VS.MORE.THAN.3M.REG.AMP
	Aroviita and Hamalainen	2008	3	BOR	WLF	NAT.IMP	FALL	GRAB	18,80	5,39	11	14,17	5,74	6	-0,798	0,276	0,054	-1,074	-0,521	LOWER.LITTORAL.VS.LESS.THAN.3M.REG.AMP
	Aroviita and Hamalainen	2008	4	BOR	WLF	NAT.IMP	FALL	GRAB	18,80	5,39	11	10,80	1,92	5	-1,619	0,373	0,054	-1,992	-1,247	LOWER.LITTORAL.VS.MORE.THAN.3M.REG.AMP
18	Valdovinos et al.	2007	1	TEM	WLF	NAT.IMP	SUMMER	GRAB	19,00	3,00	16	1,50	0,60	16	-7,885	1,096	0,099	-8,982	-6,789	REF.UNREG.VS.REG.2001
	Valdovinos et al.	2007	2	TEM	WLF	NAT.IMP	SUMMER	GRAB	16,70	3,50	16	2,75	0,50	16	-5,439	0,587	0,099	-6,026	-4,852	REF.UNREG.VS.REG.2002
87	Marchetti et al.	2011	1	TEM	FR	GRAD	WINTER	NET	17,50	0,70	4	16,50	1,50	4	-0,742	0,534	0,034	-1,277	-0,208	REF.VS.IMP.HIGH.VS.LOW.JANUARY.BENTHIC
	Marchetti et al.	2011	2	TEM	FR	GRAD	SPRING	NET	20,00	0,70	4	19,00	0,40	4	-1,524	0,645	0,034	-2,169	-0,879	REF.VS.IMP.HIGH.VS.LOW.APRIL.BENTHIC
	Marchetti et al.	2011	3	TEM	FR	GRAD	SUMMER	NET	19,50	0,30	4	21,00	0,40	4	3,685	1,349	0,034	2,336	5,034	REF.VS.IMP.HIGH.VS.LOW.JULY.BENTHIC
	Marchetti et al.	2011	4	TEM	FR	GRAD	WINTER	NET	13,60	1,60	4	15,00	1,60	4	0,760	0,536	0,034	0,224	1,296	REF.VS.IMP.HIGH.VS.LOW.JANUARY.DRIFT
	Marchetti et al.	2011	5	TEM	FR	GRAD	SPRING	NET	18,40	1,30	4	14,80	1,80	4	-1,992	0,748	0,034	-2,740	-1,244	REF.VS.IMP.HIGH.VS.LOW.APRIL.DRIFT
	Marchetti et al.	2011	6	TEM	FR	GRAD	SUMMER	NET	16,60	2,10	4	20,10	2,60	4	1,286	0,603	0,034	0,683	1,890	REF.VS.IMP.HIGH.VS.LOW.JULY.DRIFT
94	Molozzi et al.	2013	1	TRO	WLF	NAT.IMP	NA	GRAB	20,50	2,12	2	27,50	3,54	2	1,354	1,229	0,084	0,125	2,583	REF.IBIRITE.0.70M.VS.IMP.VARGEM.DAS.FLORES.2.54M
	Molozzi et al.	2013	2	TRO	WLF	NAT.IMP	NA	GRAB	20,50	2,12	2	29,00	0,00	2	3,199	2,279	0,084	0,920	5,478	REF.IBIRITE.0.70M.VS.IMP.SERRA.AZUL.5.71M
163	Takao et al.	2008	1	TEM	FR	NAT.IMP	WINTER	NET	79,88	8,50	4	62,74	7,42	4	-1,866	0,718	0,203	-2,584	-1,148	REF.TRIBUTARY.VS.IMP.DAM
203	Kullasoot et al.	2017	2	TRO	FR	GRAD	NA	NET	10,33	4,92	6	12,83	5,86	6	0,426	0,341	0,037	0,085	0,767	REF.S2.TRIBUTARY.VS.IMP.S1
	Kullasoot et al.	2017	3	TRO	FR	GRAD	NA	NET	10,33	4,92	6	6,06	6,11	6	-0,710	0,354	0,037	-1,065	-0,356	REF.S2.TRIBUTARY.VS.IMP.S3
	Kullasoot et al.	2017	4	TRO	FR	GRAD	NA	NET	10,33	4,92	6	11,61	3,77	6	0,269	0,336	0,037	-0,067	0,606	REF.S2.TRIBUTARY.VS.IMP.S4
	Kullasoot et al.	2017	5	TRO	FR	GRAD	NA	NET	10,33	4,92	6	10,17	5,96	6	-0,027	0,333	0,037	-0,360	0,306	REF.S2.TRIBUTARY.VS.IMP.S5
	Kullasoot et al.	2017	6	TRO	FR	GRAD	NA	NET	10,33	4,92	6	6,78	3,19	6	-0,790	0,359	0,037	-1,149	-0,431	REF.S2.TRIBUTARY.VS.IMP.S6
	Kullasoot et al.	2017	7	TRO	FR	GRAD	NA	NET	10,33	4,92	6	13,00	7,01	6	0,407	0,340	0,037	0,067	0,747	REF.S2.TRIBUTARY.VS.IMP.S7
330	White et al.	2011	1	BOR	WLF	NAT.IMP	SUMMER	NET	24,86	4,38	14	26,88	5,72	8	0,397	0,200	0,075	0,197	0,597	REF.NAT.VS.IMP.LOW.WLF
	White et al.	2011	2	BOR	WLF	NAT.IMP	SUMMER	NET	24,86	4,38	14	21,44	5,35	8	-0,694	0,207	0,075	-0,901	-0,486	REF.NAT.VS.IMP.MOD.WLF
	White et al.	2011	3	BOR	WLF	NAT.IMP	SUMMER	NET	24,86	4,38	14	13,88	3,04	8	-2,665	0,358	0,075	-3,023	-2,307	REF.NAT.VS.IMP.HIGH.WLF

Table A.2 (continued) Metadata table (2 of 3) showing all variables for each study included in this meta-analysis (Aroviita & Hämäläinen, 2008; Bruno et al., 2019; Englund & Malmqvist, 1996; Jackson et al., 2007; Kraft, 1988; Kullasoot et al., 2017; Marchetti et al., 2011; Mellado-Dìaz et al., 2019; Milner et al., 2019; Molozzi et al., 2013; Schneider & Petrin, 2017; Smokorowski et al., 2011; Steel et al., 2018; Takao et al., 2008; Valdovinos et al., 2007; White et al., 2011). Use Table A.3 as a companion table to get more information on each variable.

UID	AUTHORS	YEAR	OBS	BIOME	IMP	STUDY	SEASON	GEAR	REF.MEAN	REF.SD	REF.N	IMP.MEAN	IMP.SD	IMP.N	YI	VI	WI	CI.LOW	CI.UP	NOTES
331	Smokorowski et al.	2011	1	TEM	FR	NAT.IMP	SUMMER	CB	14,00	0,73	30	18,00	0,77	30	5,262	0,297	0,076	4,965	5,560	REF.BATCHAWANA.VS.IMP.MAGPIE.2002
	Smokorowski et al.	2011	2	TEM	FR	NAT.IMP	SUMMER	CB	17,00	0,78	30	17,00	0,78	30	0,000	0,067	0,076	-0,067	0,067	REF.BATCHAWANA.VS.IMP.MAGPIE.2003
	Smokorowski et al.	2011	3	TEM	FR	NAT.IMP	SUMMER	CB	15,00	0,90	30	17,00	0,80	30	2,318	0,111	0,076	2,207	2,430	REF.BATCHAWANA.VS.IMP.MAGPIE.2004
335	Englund and Malmqvist	1996	1	BOR	FR	NAT.IMP	NA	NET	80,00	4,46	14	74,00	11,20	16	-0,668	0,141	0,115	-0,809	-0,526	REF.UNREG.VS.IMP.REG
	Englund and Malmqvist	1996	2	BOR	FR	NAT.IMP	NA	NET	80,00	4,46	14	71,00	10,22	21	-1,043	0,135	0,115	-1,177	-0,908	REF.UNREG.VS.IMP.REG.RED.FLOWS
	Jackson et al.	2007	1	TEM	FR	GRAD	SPRING	CB	19,12	2,98	10	3,24	1,92	5	-5,537	1,322	0,023	-6,859	-4,215	REF.SPRING.POOLED.VS.IMP.S1
	Jackson et al.	2007	2	TEM	FR	GRAD	SPRING	CB	19,12	2,98	10	9,03	1,62	5	-3,600	0,732	0,023	-4,332	-2,868	REF.SPRING.POOLED.VS.IMP.S2
	Jackson et al.	2007	3	TEM	FR	GRAD	SPRING	CB	19,12	2,98	10	14,97	4,35	5	-1,129	0,342	0,023	-1,471	-0,786	REF.SPRING.POOLED.VS.IMP.S3
	Jackson et al.	2007	4	TEM	FR	GRAD	FALL	CB	23,19	3,02	10	10,32	2,37	5	-4,270	0,908	0,023	-5,178	-3,363	REF.AUTUMN.POOLED.VS.IMP.S1
341	Jackson et al.	2007	5	TEM	FR	GRAD	FALL	CB	23,19	3,02	10	13,13	4,38	5	-2,708	0,544	0,023	-3,253	-2,164	REF.AUTUMN.POOLED.VS.IMP.S2
	Jackson et al.	2007	6	TEM	FR	GRAD	FALL	CB	23,19	3,02	10	21,00	3,68	5	-0,637	0,314	0,023	-0,950	-0,323	REF.AUTUMN.POOLED.VS.IMP.S3
	Jackson et al.	2007	7	TEM	FR	GRAD	WINTER	CB	14,77	3,16	10	10,20	3,85	5	-1,270	0,354	0,023	-1,623	-0,916	REF.WINTER.POOLED.VS.IMP.S1
	Jackson et al.	2007	8	TEM	FR	GRAD	WINTER	CB	14,77	3,16	10	9,91	3,58	5	-1,388	0,364	0,023	-1,752	-1,024	REF.WINTER.POOLED.VS.IMP.S2
	Jackson et al.	2007	9	TEM	FR	GRAD	WINTER	CB	14,77	3,16	10	17,42	4,06	5	0,720	0,317	0,023	0,403	1,038	REF.WINTER.POOLED.VS.IMP.S3
	Kraft	1988	1	TEM	WLF	NAT.IMP	SUMMER	GRAB	5,50	1,70	5	4,30	0,60	5	-0,850	0,436	0,036	-1,286	-0,414	REF.RED.WLF.VS.IMP.UNNAT.WLF.JUN84
	Kraft	1988	2	TEM	WLF	NAT.IMP	SUMMER	GRAB	5,30	1,60	5	4,60	0,90	5	-0,487	0,412	0,036	-0,899	-0,075	REF.RED.WLF.VS.IMP.UNNAT.WLF.JUL84
345	Kraft	1988	3	TEM	WLF	NAT.IMP	SUMMER	GRAB	5,80	2,00	5	6,90	1,90	5	0,509	0,413	0,036	0,096	0,922	REF.RED.WLF.VS.IMP.UNNAT.WLF.AUG84
	Kraft	1988	4	TEM	WLF	NAT.IMP	SUMMER	GRAB	7,90	1,60	5	9,10	1,30	5	0,743	0,428	0,036	0,315	1,171	REF.RED.WLF.VS.IMP.UNNAT.WLF.JUN85
	Kraft	1988	5	TEM	WLF	NAT.IMP	SUMMER	GRAB	8,70	2,60	5	10,30	2,20	5	0,600	0,418	0,036	0,182	1,018	REF.RED.WLF.VS.IMP.UNNAT.WLF.AUG85
	Kraft	1988	6	TEM	WLF	NAT.IMP	SUMMER	GRAB	8,20	2,70	5	6,90	1,50	5	-0,537	0,414	0,036	-0,952	-0,123	REF.RED.WLF.VS.IMP.UNNAT.WLF.JUN86
405	Schneider and Petrin	2017	1	BOR	FR	NAT.IMP	SUMMER	NET	10,82	3,89	20	14,01	7,11	20	0,546	0,104	0,232	0,442	0,649	REF.UNREG.NAT.FLOW.VS.IMP.REG.MOD.FLOW
406	Vaikasas et al.	2013	1	TEM	FR	GRAD	SPRING	NET	28,00	1,73	3	17,00	3,16	10	-3,465	0,895	0,102	-4,361	-2,570	REF.CONTROL.VS.IMP.IN.HPP.DAM
	Vaikasas et al.	2013	2	TEM	FR	GRAD	SPRING	NET	28,00	1,73	3	20,00	6,32	10	-1,291	0,497	0,102	-1,788	-0,793	REF.CONTROL.VS.IMP.BELOW.HPP.DAM

Table A.2 (continued) Metadata table (3 of 3) showing all variables for each study included in this meta-analysis (Aroviita & Hämäläinen, 2008; Bruno et al., 2019; Englund & Malmqvist, 1996; Jackson et al., 2007; Kraft, 1988; Kullasoot et al., 2017; Marchetti et al., 2011; Mellado-Dìaz et al., 2019; Milner et al., 2019; Molozzi et al., 2013; Schneider & Petrin, 2017; Smokorowski et al., 2011; Steel et al., 2018; Takao et al., 2008; Valdovinos et al., 2007; White et al., 2011). Use Table A.3 as a companion table to get more information on each variable.

UID	AUTHORS	YEAR	OBS	BIOME	IMP	STUDY	SEASON	GEAR	REF.MEAN	REF.SD	REF.N	IMP.MEAN	IMP.SD	IMP.N	YI	VI	WI	CI.LOW	CI.UP	NOTES
401	Mellado-Diaz et al.	2019	1	TEM	FR	GRAD	NA	NET	43,22	2,28	4	33,67	4,39	3	-2,440	1,009	0,011	-3,449	-1,431	CABRIEL.REF.VS.CLASS5.0-4KM
	Mellado-Diaz et al.	2019	2	TEM	FR	GRAD	NA	NET	43,22	2,28	4	31,32	2,73	3	-4,051	1,755	0,011	-5,806	-2,295	CABRIEL.REF.VS.CLASS4.4-10KM
	Mellado-Diaz et al.	2019	3	TEM	FR	GRAD	NA	NET	43,22	2,28	4	28,62	2,95	3	-4,778	2,214	0,011	-6,992	-2,564	CABRIEL.REF.VS.CLASS2.20-30KM
	Mellado-Diaz et al.	2019	4	TEM	FR	GRAD	NA	NET	43,22	2,28	4	34,27	4,20	3	-2,359	0,981	0,011	-3,340	-1,378	CABRIEL.REF.VS.CLASS1.OVER30KM
	Mellado-Diaz et al.	2019	5	TEM	FR	GRAD	NA	NET	46,53	2,59	4	44,37	3,79	2,5	-0,583	0,676	0,011	-1,260	0,093	GUADA.REF.VS.CLASS5.0-4KM
	Mellado-Diaz et al.	2019	6	TEM	FR	GRAD	NA	NET	46,53	2,59	4	39,53	0,54	2,5	-2,691	1,207	0,011	-3,899	-1,484	GUADA.REF.VS.CLASS4.4-10KM
	Mellado-Diaz et al.	2019	7	TEM	FR	GRAD	NA	NET	46,53	2,59	4	44,02	1,01	2,5	-0,940	0,718	0,011	-1,658	-0,222	GUADA.REF.VS.CLASS3.10-20KM
	Mellado-Diaz et al.	2019	8	TEM	FR	GRAD	NA	NET	46,53	2,59	4	35,01	2,29	2,5	-3,796	1,759	0,011	-5,555	-2,038	GUADA.REF.VS.CLASS2.20-30KM
	Mellado-Diaz et al.	2019	9	TEM	FR	GRAD	NA	NET	49,69	1,75	9	55,37	0,69	2,67	3,231	0,933	0,011	2,298	4,164	AVIA.REF.VS.CLASS5.0-4KM
	Mellado-Diaz et al.	2019	10	TEM	FR	GRAD	NA	NET	49,69	1,75	9	45,49	2,51	2,67	-2,030	0,662	0,011	-2,693	-1,368	AVIA.REF.VS.CLASS4.4-10KM
	Mellado-Diaz et al.	2019	11	TEM	FR	GRAD	NA	NET	49,69	1,75	9	41,99	1,59	2,67	-4,111	1,210	0,011	-5,320	-2,901	AVIA.REF.VS.CLASS3.10-20KM
	Mellado-Diaz et al.	2019	12	TEM	FR	GRAD	NA	NET	41,64	4,07	9	32,98	3,04	3	-2,056	0,621	0,011	-2,677	-1,436	NAJERILLA.REF.VS.CLASS5.0-4KM
	Mellado-Diaz et al.	2019	13	TEM	FR	GRAD	NA	NET	41,64	4,07	9	42,35	3,82	3	0,163	0,446	0,011	-0,283	0,608	NAJERILLA.REF.VS.CLASS3.10-20KM
	Mellado-Diaz et al.	2019	14	TEM	FR	GRAD	NA	NET	41,64	4,07	9	52,34	2,90	3	2,555	0,716	0,011	1,838	3,271	NAJERILLA.REF.VS.CLASS2.20-30KM
	Mellado-Diaz et al.	2019	15	TEM	FR	GRAD	NA	NET	39,02	1,45	7	32,96	2,05	2,33	-3,436	1,205	0,011	-4,640	-2,231	PORMA.REF.VS.CLASS5.0-4KM
	Mellado-Diaz et al.	2019	16	TEM	FR	GRAD	NA	NET	39,02	1,45	7	34,48	0,50	2,33	-3,052	1,071	0,011	-4,123	-1,981	PORMA.REF.VS.CLASS3.10-20KM
	Mellado-Diaz et al.	2019	17	TEM	FR	GRAD	NA	NET	39,02	1,45	7	40,30	1,78	2,33	0,755	0,603	0,011	0,152	1,357	PORMA.REF.VS.CLASS1.OVER30KM
402	Bruno et al.	2019	1	TEM	FR	NAT.IMP	NA	NET	35,60	1,90	5	36,20	3,80	10	0,169	0,301	0,109	-0,132	0,470	1970.FREE.FLOW.CATCH.VS.REG.CATCH
	Bruno et al.	2019	2	TEM	FR	NAT.IMP	NA	NET	38,60	5,10	5	28,40	4,90	10	-1,934	0,425	0,109	-2,359	-1,509	2010.FREE.FLOW.CATCH.VS.REG.CATCH.PLUS.CC
403	Milner et al.	2019	1	TEM	FR	NAT.IMP	SUMMER	NET	31,00	1,00	3	19,67	9,45	3	-1,345	0,817	0,067	-2,163	-0,528	US.OF.JUNCTION.UNREG.CATCH.VS.REG.CATCH
	Milner et al.	2019	2	TEM	FR	NAT.IMP	SUMMER	NET	31,30	10,69	3	40,33	7,77	3	0,771	0,716	0,067	0,055	1,487	TRIBUTARY.OF.JUNCTION.UNREG.CATCH.VS.REG.CATCH
	Milner et al.	2019	3	TEM	FR	NAT.IMP	SUMMER	NET	31,33	2,31	3	28,33	5,86	3	-0,537	0,691	0,067	-1,228	0,153	DS.OF.JUNCTION.UNREG.CATCH.VS.REG.CATCH
404	Steel et al.	2018	1	TEM	FR	NAT.IMP	SUMMER	NET	36,67	3,06	3	35,00	2,65	3	-0,466	0,685	0,066	-1,150	0,219	REF.NFA.VS.IMP.RUBICON.BYPASS
	Steel et al.	2018	2	TEM	FR	NAT.IMP	SUMMER	NET	31,33	9,45	3	21,33	4,16	3	-1,093	0,766	0,066	-1,859	-0,327	REF.NFY.VS.IMP.SFY.AUGMENTED
	Steel et al.	2018	3	TEM	FR	NAT.IMP	SUMMER	NET	36,67	3,06	3	25,00	8,49	2	-1,535	1,069	0,066	-2,604	-0,466	REF.NFA.VS.IMP.MFA.HYDROPEAKING

Table A.3 Companion table describing all variables in Table A.2.

VARIABLES	DESCRIPTION	TYPE
UID	Study/paper unique identifier	Numerical
AUTHORS	List of authors, written as if they were cited in a paper	Character string
YEAR	Publication year	Numerical
OBS	Identification number for each observation in the study	Numerical
BIOME	Biomes derived from terrestrial ecoregions as defined by the World Wildlife Fund (WWF; boreal [BOR], temperate [TEM] and tropical [TRO])	Categorical
IMP	Type of impact from hydropower, either flow regulation [FLOW.REG] or water level fluctuation [WLF]	Categorical
STUDY	Type of study, either longitudinal gradient [GRAD] or cross-sectional (natural vs impacted [NAT.IMP])	Categorical
SEASON	Season samples were collected [SPRING, SUMMER, FALL, WINTER]. Samples that were collected over multiple seasons (composite) are referred to as NA	Categorical
GEAR	Sampling gear used to sample macroinvertebrates; d-nets, kick-sample and Surber [NET], Ponar and Ekman [GRAB] and colonization baskets [COL.BASKET]	Categorical
REF.MEAN	Mean richness in reference sample (number of taxa)	Numerical
REF.SD	Standard deviation of mean richness in reference sample	Numerical
REF.N	Sample number for mean richness in reference sample	Numerical
IMP.MEAN	Mean richness in impacted sample (number of taxa)	Numerical
IMP.SD	Standard deviation of mean richness in impacted sample	Numerical
IMP.N	Sample number for mean richness in impacted sample	Numerical
YI	Observed effect sizes (g)	Numerical
VI	Sampling variance corresponding to respective effect sizes (V_g)	Numerical
WI	Inverse variance as weight associated to each study	Numerical
CI.LOW	Lower confidence interval	Numerical
CI.UP	Upper confidence interval	Numerical
NOTES	Description of impact the effect size represents	Character string

Table A.4 Table showing the sensitive analysis outputs. Rho values (ρ) ranges from 0 to 1 and mean Effect Size (ES), Standard Error (SE) and between study variance (τ^2) estimates are relatively insensitive to these varying ρ values.

Estimate type	$\rho = 0$	$\rho = 0.2$	$\rho = 0.4$	$\rho = 0.6$	$\rho = 0.8$	$\rho = 1$
Mean Effect Size (ES)	-0.864	-0.864	-0.864	-0.864	-0.864	-0.864
Standard Error (SE)	0.475	0.475	0.475	0.475	0.475	0.475
Between-study variance (τ^2)	4.198	4.200	4.202	4.205	4.207	4.209

APPENDIX B ARTICLE 2

Table B.1 Raw data table for our study of reservoir richness in the United States including variables such as the unique identifier (UID) for each reservoir, latitude (LAT), longitude (LON), ecoregion (ECO), elevation (ELE; in meters), area (AREA; in hectares), trophic state (TS), number of river and stream samples in the ecoregion (N.REF), mean native riverine richness per ecoregion (MEAN.REF.S), standard deviation of the mean native riverine richness per ecoregion(SD.REF.S), number of reservoir samples (N.IMP; at the reservoir level hence always one), reservoir richness (IMP.S; specific to each reservoir, not a mean), standard deviation of the reservoir richness (SD.IMP.S; one reservoir, thus standard deviation always zero), Potentially Disappeared Fraction of species (PDF), standard deviation associated to the Potentially Disappeared Fraction of species (SD.PDF), lower confidence interval (LOW.CI) and higher confidence interval (UP.CI).

UID	LAT	LON	ECO	ELE	AREA	TS	N.REF	MEAN.REF.S	SD.REF.S	N.IMP	IMP.S	SD.IMP.S	PDF	SD.PDF	LOW.CI	UP.CI
6243	38.507965	-94.673265	TPL	295.8	110.0	EUT	209	32.0	12.9	1	42	0	-0.314	0.126	-0.562	-0.066
6252	40.111767	-75.861530	SAP	186.7	63.2	EUT	344	45.8	15.1	1	35	0	0.236	0.078	0.083	0.388
6267	31.621546	-88.353739	CPL	50.8	32.6	EUT	327	28.4	16.6	1	32	0	-0.126	0.074	-0.271	0.019
6270	35.174388	-99.077489	SPL	499.9	141.5	EUT	176	26.1	13.1	1	14	0	0.465	0.233	0.007	0.922
6281	35.285013	-112.154163	WMT	2072.1	24.8	EUT	222	39.8	12.0	1	15	0	0.623	0.188	0.254	0.992
6319	36.831633	-104.226303	WMT	2066.2	44.2	EUT	222	39.8	12.0	1	16	0	0.598	0.181	0.243	0.952
6342	39.994962	-105.112227	SPL	1620.9	18.2	EUT	176	26.1	13.1	1	26	0	0.006	0.003	0.000	0.011
6404	39.638354	-95.456811	TPL	321.7	26.9	EUT	209	32.0	12.9	1	22	0	0.312	0.126	0.066	0.558
6437	39.000154	-95.779648	TPL	350.6	101.9	EUT	209	32.0	12.9	1	34	0	-0.064	0.026	-0.114	-0.013
6451	40.161787	-79.052384	SAP	551.9	18.2	EUT	344	45.8	15.1	1	43	0	0.061	0.020	0.022	0.101
6481	39.484355	-118.723571	XER	1202.0	164.1	EUT	213	31.0	11.6	1	19	0	0.387	0.144	0.104	0.670
6482	41.928985	-119.179002	XER	1678.2	100.3	EUT	213	31.0	11.6	1	16	0	0.484	0.181	0.129	0.838
6501	35.980828	-108.931643	WMT	2290.2	15.4	EUT	222	39.8	12.0	1	14	0	0.648	0.196	0.264	1.032
6525	35.992932	-96.873677	SPL	255.6	177.7	EUT	176	26.1	13.1	1	40	0	-0.530	0.266	-1.051	-0.009
6550	34.953616	-96.718159	SPL	281.1	533.1	EUT	176	26.1	13.1	1	27	0	-0.033	0.016	-0.065	-0.001

Table B.1 (continued)

UID	LAT	LON	ECO	ELE	AREA	TS	N.REF	MEAN.REF.S	SD.REF.S	N.IMP	IMP.S	SD.IMP.S	PDF	SD.PDF	LOW.CI	UP.CI
6556	35.412203	-95.929276	SPL	201.3	204.3	EUT	176	26.1	13.1	1	25	0	0.044	0.022	0.001	0.087
6570	37.655565	-98.260986	SPL	479.3	56.8	EUT	176	26.1	13.1	1	39	0	-0.492	0.247	-0.975	-0.008
6575	40.723819	-109.183908	XER	2184.0	43.3	EUT	213	31.0	11.6	1	21	0	0.322	0.120	0.086	0.558
6586	31.787274	-96.064492	CPL	91.4	852.5	EUT	327	28.4	16.6	1	28	0	0.015	0.009	-0.002	0.031
6599	46.543922	-104.028258	NPL	907.4	3.8	EUT	179	29.4	10.1	1	33	0	-0.121	0.042	-0.203	-0.040
6606	37.390887	-99.784838	SPL	685.8	127.3	EUT	176	26.1	13.1	1	28	0	-0.071	0.036	-0.141	-0.001
6617	41.757410	-115.722027	XER	2089.2	23.6	EUT	213	31.0	11.6	1	26	0	0.161	0.060	0.043	0.279
6618	41.198060	-115.892296	XER	1814.2	5.0	EUT	213	31.0	11.6	1	26	0	0.161	0.060	0.043	0.279
6622	31.889172	-97.702492	SPL	290.0	20.7	EUT	176	26.1	13.1	1	35	0	-0.339	0.170	-0.672	-0.005
6668	36.823001	-96.047588	SPL	230.6	103.5	EUT	176	26.1	13.1	1	38	0	-0.453	0.228	-0.899	-0.007
6695	36.705443	-96.419109	TPL	266.8	325.4	EUT	209	32.0	12.9	1	40	0	-0.251	0.101	-0.450	-0.053
6719	38.398162	-115.117053	XER	1574.3	72.3	EUT	213	31.0	11.6	1	23	0	0.258	0.096	0.069	0.446
6731	41.701690	-113.959671	XER	1622.6	10.3	EUT	213	31.0	11.6	1	26	0	0.161	0.060	0.043	0.279
6735	32.944430	-96.453752	SPL	146.0	13.8	EUT	176	26.1	13.1	1	34	0	-0.300	0.151	-0.596	-0.005
6742	47.761716	-108.432829	NPL	912.2	4.3	EUT	179	29.4	10.1	1	30	0	-0.019	0.007	-0.032	-0.006
6753	39.931042	-104.973296	SPL	1600.5	9.2	EUT	176	26.1	13.1	1	20	0	0.235	0.118	0.004	0.466
6762	33.516175	-94.125132	CPL	82.4	17.1	EUT	327	28.4	16.6	1	43	0	-0.513	0.301	-1.103	0.076
6774	46.826042	-100.634208	NPL	523.4	3.9	EUT	179	29.4	10.1	1	16	0	0.456	0.157	0.149	0.764
6795	38.997241	-108.051180	WMT	3070.9	15.0	EUT	222	39.8	12.0	1	32	0	0.195	0.059	0.080	0.311
6796	37.193346	-95.988976	SPL	252.1	13.5	EUT	176	26.1	13.1	1	36	0	-0.377	0.189	-0.748	-0.006
6806	38.491412	-79.314781	SAP	604.1	3.8	EUT	344	45.8	15.1	1	43	0	0.061	0.020	0.022	0.101
6823	43.165878	-115.652476	XER	997.2	163.9	EUT	213	31.0	11.6	1	24	0	0.225	0.084	0.060	0.390
6868	38.235087	-112.463009	WMT	2680.7	9.6	EUT	222	39.8	12.0	1	26	0	0.346	0.105	0.141	0.551
6869	38.847537	-111.961390	XER	1589.5	93.2	EUT	213	31.0	11.6	1	16	0	0.484	0.181	0.129	0.838
6874	39.036703	-107.911131	WMT	3105.2	5.7	EUT	222	39.8	12.0	1	30	0	0.246	0.074	0.100	0.391
6875	40.944919	-106.011968	XER	2410.4	17.4	EUT	213	31.0	11.6	1	26	0	0.161	0.060	0.043	0.279

Table B.1 (continued)

UID	LAT	LON	ECO	ELE	AREA	TS	N.REF	MEAN.REF.S	SD.REF.S	N.IMP	IMP.S	SD.IMP.S	PDF	SD.PDF	LOW.CI	UP.CI
6923	40.039991	-81.013888	SAP	322.7	43.0	EUT	344	45.8	15.1	1	24	0	0.476	0.157	0.168	0.784
6940	44.329096	-116.184107	WMT	1507.0	71.6	EUT	222	39.8	12.0	1	29	0	0.271	0.082	0.110	0.431
6944	39.169507	-111.450721	WMT	2837.7	18.8	EUT	222	39.8	12.0	1	29	0	0.271	0.082	0.110	0.431
6959	44.964115	-116.463019	WMT	1453.0	211.4	EUT	222	39.8	12.0	1	32	0	0.195	0.059	0.080	0.311
6966	39.142411	-111.452546	WMT	2889.5	27.9	EUT	222	39.8	12.0	1	9	0	0.774	0.234	0.315	1.232
6970	44.796705	-116.732688	WMT	2154.4	12.4	OLI	222	39.8	12.0	1	13	0	0.673	0.204	0.274	1.072
6971	43.191413	-116.959804	XER	1399.8	73.0	EUT	213	31.0	11.6	1	8	0	0.742	0.277	0.199	1.285
6976	38.791149	-105.106361	WMT	3147.0	10.2	EUT	222	39.8	12.0	1	4	0	0.899	0.272	0.366	1.433
7020	38.078326	-122.743359	XER	51.1	335.3	EUT	213	31.0	11.6	1	36	0	-0.162	0.061	-0.281	-0.043
7057	39.204737	-111.668912	WMT	1789.8	24.8	EUT	222	39.8	12.0	1	30	0	0.246	0.074	0.100	0.391
7097	38.788187	-111.774878	WMT	2203.2	6.7	EUT	222	39.8	12.0	1	15	0	0.623	0.188	0.254	0.992
7100	43.965218	-122.683968	WMT	255.3	709.5	EUT	222	39.8	12.0	1	37	0	0.070	0.021	0.028	0.111
7105	41.110516	-82.083872	NAP	258.0	21.0	EUT	225	46.0	13.8	1	28	0	0.391	0.118	0.160	0.622
7108	30.963438	-95.903504	CPL	86.9	27.3	EUT	327	28.4	16.6	1	27	0	0.050	0.029	-0.007	0.107
7109	32.072696	-97.129773	SPL	186.9	12.6	EUT	176	26.1	13.1	1	24	0	0.082	0.041	0.001	0.163
7136	41.633291	-118.389357	XER	1311.6	15.4	EUT	213	31.0	11.6	1	23	0	0.258	0.096	0.069	0.446
7205	32.240254	-101.313303	SPL	711.1	56.8	OLI	176	26.1	13.1	1	2	0	0.924	0.464	0.015	1.832
7207	42.157825	-122.607634	WMT	684.2	256.5	EUT	222	39.8	12.0	1	26	0	0.346	0.105	0.141	0.551
7226	42.130013	-122.478277	WMT	1344.3	4.4	EUT	222	39.8	12.0	1	19	0	0.522	0.158	0.213	0.832
7228	34.227816	-86.843449	SAP	247.0	73.0	EUT	344	45.8	15.1	1	44	0	0.039	0.013	0.014	0.065
7229	40.631585	-120.002870	XER	1329.5	37.1	EUT	213	31.0	11.6	1	20	0	0.354	0.132	0.095	0.614
7232	40.703407	-83.378745	TPL	269.7	102.7	EUT	209	32.0	12.9	1	17	0	0.468	0.189	0.099	0.838
7276	41.168583	-119.817451	XER	1560.8	28.9	OLI	213	31.0	11.6	1	9	0	0.710	0.265	0.190	1.229
7294	39.056042	-82.690673	SAP	211.5	65.1	EUT	344	45.8	15.1	1	47	0	-0.026	0.009	-0.043	-0.009
7304	31.587497	-98.622503	SPL	448.8	27.8	EUT	176	26.1	13.1	1	36	0	-0.377	0.189	-0.748	-0.006
7306	39.241149	-117.165818	XER	2255.5	5.8	EUT	213	31.0	11.6	1	19	0	0.387	0.144	0.104	0.670

Table B.1 (continued)

UID	LAT	LON	ECO	ELE	AREA	TS	N.REF	MEAN.REF.S	SD.REF.S	N.IMP	IMP.S	SD.IMP.S	PDF	SD.PDF	LOW.CI	UP.CI
7325	40.337080	-105.126694	SPL	1562.4	189.8	EUT	176	26.1	13.1	1	19	0	0.273	0.137	0.004	0.542
7368	32.515869	-87.861085	CPL	22.3	4731.5	EUT	327	28.4	16.6	1	29	0	-0.021	0.012	-0.044	0.003
7369	41.035420	-96.837727	TPL	391.7	29.2	EUT	209	32.0	12.9	1	26	0	0.187	0.075	0.039	0.334
7375	33.364862	-88.166880	CPL	78.1	5.6	EUT	327	28.4	16.6	1	45	0	-0.584	0.342	-1.254	0.086
7392	34.534351	-92.268826	CPL	74.1	105.8	EUT	327	28.4	16.6	1	26	0	0.085	0.050	-0.013	0.182
7402	34.284778	-97.170972	SPL	245.4	160.7	EUT	176	26.1	13.1	1	39	0	-0.492	0.247	-0.975	-0.008
7405	40.328099	-96.532001	TPL	425.9	32.1	EUT	209	32.0	12.9	1	25	0	0.218	0.088	0.046	0.390
7409	33.075563	-92.660596	CPL	55.4	7.3	EUT	327	28.4	16.6	1	31	0	-0.091	0.053	-0.196	0.013
7459	33.882010	-85.931618	SAP	171.0	18.1	EUT	344	45.8	15.1	1	39	0	0.148	0.049	0.052	0.244
7471	46.040623	-110.692175	NPL	1556.2	96.7	EUT	179	29.4	10.1	1	23	0	0.219	0.075	0.071	0.366
7472	46.624624	-110.738336	NPL	1672.9	150.7	EUT	179	29.4	10.1	1	25	0	0.151	0.052	0.049	0.252
7533	43.078399	-112.693659	XER	1338.8	21.3	EUT	213	31.0	11.6	1	14	0	0.548	0.205	0.147	0.949
7572	40.372482	-84.340110	TPL	291.8	327.1	EUT	209	32.0	12.9	1	22	0	0.312	0.126	0.066	0.558
7579	39.608156	-84.971507	TPL	254.8	72.8	EUT	209	32.0	12.9	1	21	0	0.343	0.138	0.072	0.614
7643	39.706824	-111.293369	WMT	2569.0	31.5	EUT	222	39.8	12.0	1	29	0	0.271	0.082	0.110	0.431
7652	40.176639	-84.265220	TPL	275.4	15.3	EUT	209	32.0	12.9	1	29	0	0.093	0.037	0.020	0.166
7684	37.673281	-107.112778	WMT	3530.5	2.1	EUT	222	39.8	12.0	1	19	0	0.522	0.158	0.213	0.832
7686	37.316232	-107.112994	WMT	2348.4	35.2	EUT	222	39.8	12.0	1	34	0	0.145	0.044	0.059	0.231
7698	41.152339	-110.824953	XER	2180.1	90.9	EUT	213	31.0	11.6	1	8	0	0.742	0.277	0.199	1.285
7713	46.118216	-113.374640	WMT	1847.6	152.3	EUT	222	39.8	12.0	1	26	0	0.346	0.105	0.141	0.551
7800	41.677573	-73.144698	NAP	198.8	56.2	EUT	225	46.0	13.8	1	51	0	-0.109	0.033	-0.174	-0.045
7810	43.413998	-119.410472	XER	1268.6	107.8	EUT	213	31.0	11.6	1	10	0	0.677	0.253	0.181	1.173
7812	35.562459	-93.637568	SAP	203.7	44.7	EUT	344	45.8	15.1	1	28	0	0.389	0.128	0.137	0.640
8016	33.829304	-109.090421	WMT	2403.4	48.4	EUT	222	39.8	12.0	1	17	0	0.573	0.173	0.233	0.912
8121	36.067203	-91.142428	SAP	82.7	222.6	EUT	344	45.8	15.1	1	26	0	0.432	0.143	0.153	0.712
8144	35.583189	-90.962941	CPL	69.0	95.9	EUT	327	28.4	16.6	1	21	0	0.261	0.153	-0.038	0.560

Table B.1 (continued)

UID	LAT	LON	ECO	ELE	AREA	TS	N.REF	MEAN.REF.S	SD.REF.S	N.IMP	IMP.S	SD.IMP.S	PDF	SD.PDF	LOW.CI	UP.CI
8151	41.088461	-82.729015	TPL	249.8	80.7	EUT	209	32.0	12.9	1	26	0	0.187	0.075	0.039	0.334
8184	40.055586	-105.747080	WMT	3029.0	51.0	EUT	222	39.8	12.0	1	17	0	0.573	0.173	0.233	0.912
8207	39.653475	-82.473781	SAP	240.2	47.7	EUT	344	45.8	15.1	1	44	0	0.039	0.013	0.014	0.065
8250	48.380621	-110.985266	NPL	913.5	9.5	EUT	179	29.4	10.1	1	20	0	0.321	0.110	0.105	0.537
8256	39.775971	-81.522472	SAP	241.9	24.2	EUT	344	45.8	15.1	1	36	0	0.214	0.071	0.076	0.352
8278	48.026569	-109.623760	NPL	1128.8	9.8	EUT	179	29.4	10.1	1	27	0	0.083	0.028	0.027	0.138
8325	37.416782	-108.405651	WMT	2213.0	65.4	EUT	222	39.8	12.0	1	14	0	0.648	0.196	0.264	1.032
8342	32.056309	-96.731783	SPL	162.6	5.5	EUT	176	26.1	13.1	1	34	0	-0.300	0.151	-0.596	-0.005
8360	40.674602	-110.970699	WMT	3043.5	39.4	OLI	222	39.8	12.0	1	15	0	0.623	0.188	0.254	0.992
8395	40.889268	-109.846108	WMT	2622.7	32.5	EUT	222	39.8	12.0	1	24	0	0.397	0.120	0.161	0.632
8409	39.720767	-86.720223	TPL	255.2	124.2	EUT	209	32.0	12.9	1	30	0	0.062	0.025	0.013	0.110
8413	31.910344	-95.301856	CPL	129.5	481.4	EUT	327	28.4	16.6	1	38	0	-0.337	0.198	-0.725	0.050
8414	40.121695	-104.945911	SPL	1509.8	24.7	EUT	176	26.1	13.1	1	6	0	0.771	0.387	0.012	1.529
8416	38.939348	-91.282857	TPL	226.5	4.3	EUT	209	32.0	12.9	1	37	0	-0.157	0.063	-0.282	-0.033
8427	44.698479	-87.499926	UMW	179.8	35.5	EUT	167	39.3	12.7	1	30	0	0.237	0.077	0.087	0.388
8435	47.876688	-107.125232	NPL	758.1	13.8	EUT	179	29.4	10.1	1	14	0	0.524	0.180	0.171	0.878
8437	42.646907	-72.218497	NAP	195.1	138.4	EUT	225	46.0	13.8	1	40	0	0.130	0.039	0.053	0.206
8443	41.812487	-70.638227	CPL	13.7	10.8	EUT	327	28.4	16.6	1	42	0	-0.478	0.280	-1.027	0.071
8480	30.006973	-96.709810	SPL	109.8	21.4	EUT	176	26.1	13.1	1	39	0	-0.492	0.247	-0.975	-0.008
8487	39.661693	-84.646207	TPL	287.3	7.1	EUT	209	32.0	12.9	1	20	0	0.374	0.151	0.079	0.670
8494	41.989559	-71.205066	NAP	30.9	202.4	EUT	225	46.0	13.8	1	40	0	0.130	0.039	0.053	0.206
8495	46.941268	-119.278530	XER	263.5	53.1	EUT	213	31.0	11.6	1	34	0	-0.097	0.036	-0.169	-0.026
8504	38.070619	-111.375127	WMT	3074.1	11.2	EUT	222	39.8	12.0	1	9	0	0.774	0.234	0.315	1.232
8614	36.044443	-85.586295	SAP	269.0	30.2	EUT	344	45.8	15.1	1	22	0	0.520	0.171	0.184	0.856
8635	40.495536	-83.899880	TPL	303.7	2018.9	EUT	209	32.0	12.9	1	21	0	0.343	0.138	0.072	0.614
8765	37.595562	-112.254669	WMT	2389.9	70.2	EUT	222	39.8	12.0	1	25	0	0.371	0.112	0.151	0.592

Table B.1 (continued)

UID	LAT	LON	ECO	ELE	AREA	TS	N.REF	MEAN.REF.S	SD.REF.S	N.IMP	IMP.S	SD.IMP.S	PDF	SD.PDF	LOW.CI	UP.CI
8766	40.863781	-109.811815	WMT	2528.1	18.1	EUT	222	39.8	12.0	1	13	0	0.673	0.204	0.274	1.072
8777	39.360351	-111.963708	XER	1521.4	3220.7	EUT	213	31.0	11.6	1	3	0	0.903	0.337	0.242	1.565
8811	31.331826	-97.268438	SPL	180.5	14.8	EUT	176	26.1	13.1	1	41	0	-0.568	0.285	-1.127	-0.009
1000019	43.460190	-116.141301	XER	973.1	33.4	EUT	213	31.0	11.6	1	25	0	0.193	0.072	0.052	0.334
1000025	43.198407	-114.599726	XER	1678.7	44.6	EUT	213	31.0	11.6	1	27	0	0.129	0.048	0.034	0.223
1000029	42.677045	-113.407296	XER	1278.7	3395.5	EUT	213	31.0	11.6	1	12	0	0.613	0.229	0.164	1.061
1000030	42.206372	-114.878852	XER	1593.0	393.0	EUT	213	31.0	11.6	1	20	0	0.354	0.132	0.095	0.614
1000068	46.206071	-116.834367	XER	1034.1	38.1	EUT	213	31.0	11.6	1	22	0	0.290	0.108	0.078	0.502
1000073	35.582330	-101.717139	SPL	894.9	6559.4	EUT	176	26.1	13.1	1	9	0	0.656	0.329	0.011	1.301
1000084	41.037407	-100.775775	SPL	916.0	641.0	EUT	176	26.1	13.1	1	15	0	0.426	0.214	0.007	0.846
1000086	41.321987	-98.900675	SPL	657.7	1117.9	EUT	176	26.1	13.1	1	15	0	0.426	0.214	0.007	0.846
1000122	42.264258	-116.310496	XER	1690.0	35.5	EUT	213	31.0	11.6	1	25	0	0.193	0.072	0.052	0.334
1000126	42.531475	-116.364513	XER	1773.6	29.4	EUT	213	31.0	11.6	1	23	0	0.258	0.096	0.069	0.446
1000137	42.187306	-113.924080	XER	1444.7	407.4	OLI	213	31.0	11.6	1	11	0	0.645	0.241	0.173	1.117
1000223	43.537123	-90.959353	UMW	277.4	14.8	EUT	167	39.3	12.7	1	25	0	0.364	0.118	0.133	0.595

Table B.2 Variation partitioning fractions (percentages) explained. [a] to [d] represent the fractions of variation explained uniquely by each of the four matrices, [a] being the spatial matrix, [b] the physical matrix, [c] the chemical matrix and [d] the human influence matrix. [e] to [j] are the joint fractions between two matrices ([e] is spatial and physical, [f] is physical and chemical, [g] is spatial and chemical, [h] is spatial and human, [i] is physical and human, [j] is chemical and human) and [k] to [n] the joint fractions between three matrices ([k] is spatial, physical and human, [l] is spatial, physical and chemical, [m] is physical, chemical and human, and [n] is spatial, chemical and human). Finally, [o] is the joint fraction between all four matrices.

%	[a]	[b]	[c]	[d]	[e]	[f]	[g]	[h]	[i]	[j]	[k]	[l]	[m]	[n]	[o]	Value
51	0.2	14.8	5.2	0.0	9.4	0.6	0.9	-0.2	3.8	0.4	4.8	5.9	1.3	-0.1	3.9	51.0
45	0.2	14.8			9.4	0.6	0.9	-0.2	3.8		4.8	5.9	1.3	-0.1	3.9	45.4
25	0.2				9.4		0.9	-0.2			4.8	5.9		-0.1	3.9	24.9
24					9.4						4.8	5.9			3.9	24.1
11							0.9					5.9		-0.1	3.9	10.7
8								-0.2			4.8			-0.1	3.9	8.4
45		14.8			9.4	0.6			3.8		4.8	5.9	1.3		3.9	44.6
15		14.8														14.8
18			5.2			0.6	0.9			0.4		5.9	1.3	-0.1	3.9	18.1
14				0.0				-0.2	3.8	0.4	4.8		1.3	-0.1	3.9	14.0

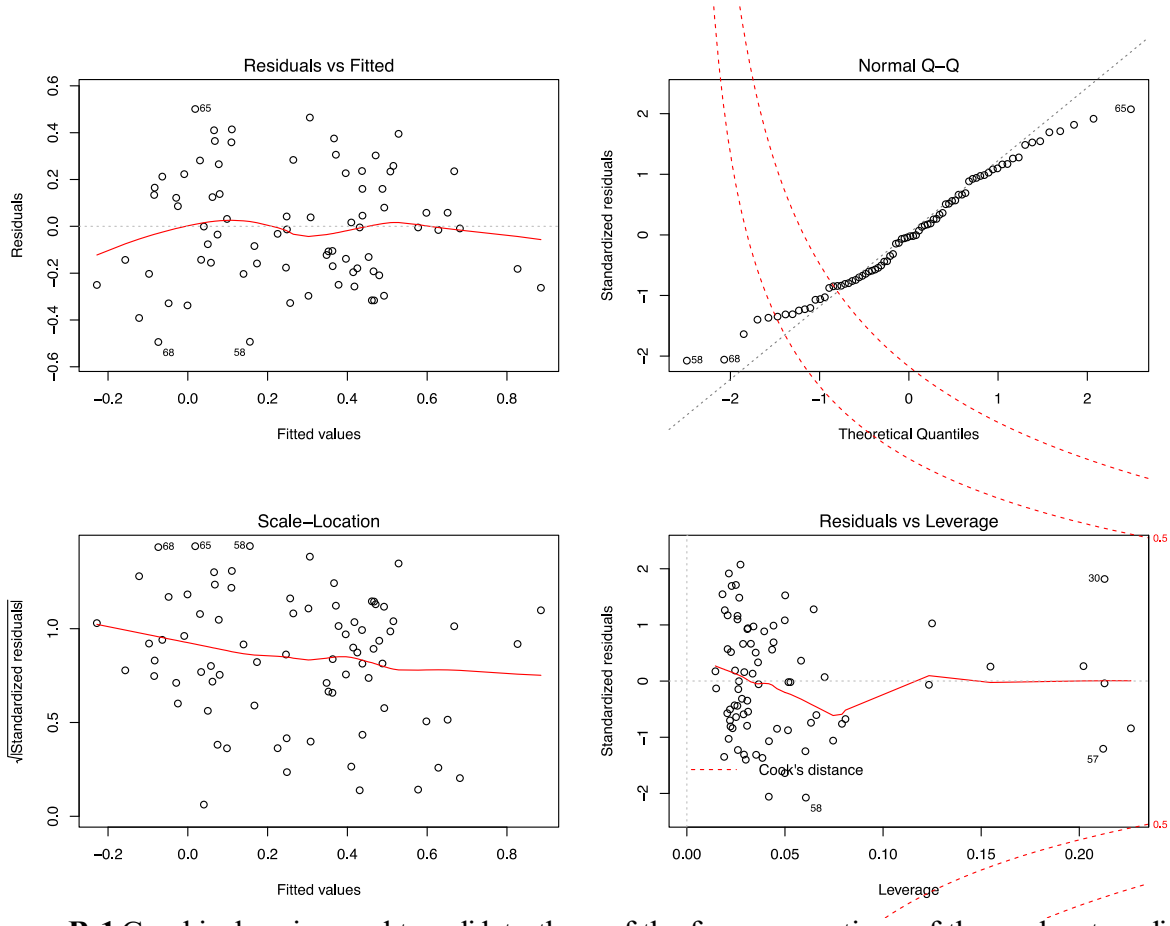

Figure B.1 Graphical series used to validate three of the four assumptions of the explanatory linear model multiple regression, that is the normality of the residuals (Normal Q-Q plot), residuals mean of 0 (Residuals vs Fitted plot), and the homoscedasticity of the residuals (Residuals vs Fitted plot and Scale-Location plot). In addition to the assumptions, we can also check for leverage points in the dataset using the Residuals vs Leverage plot.

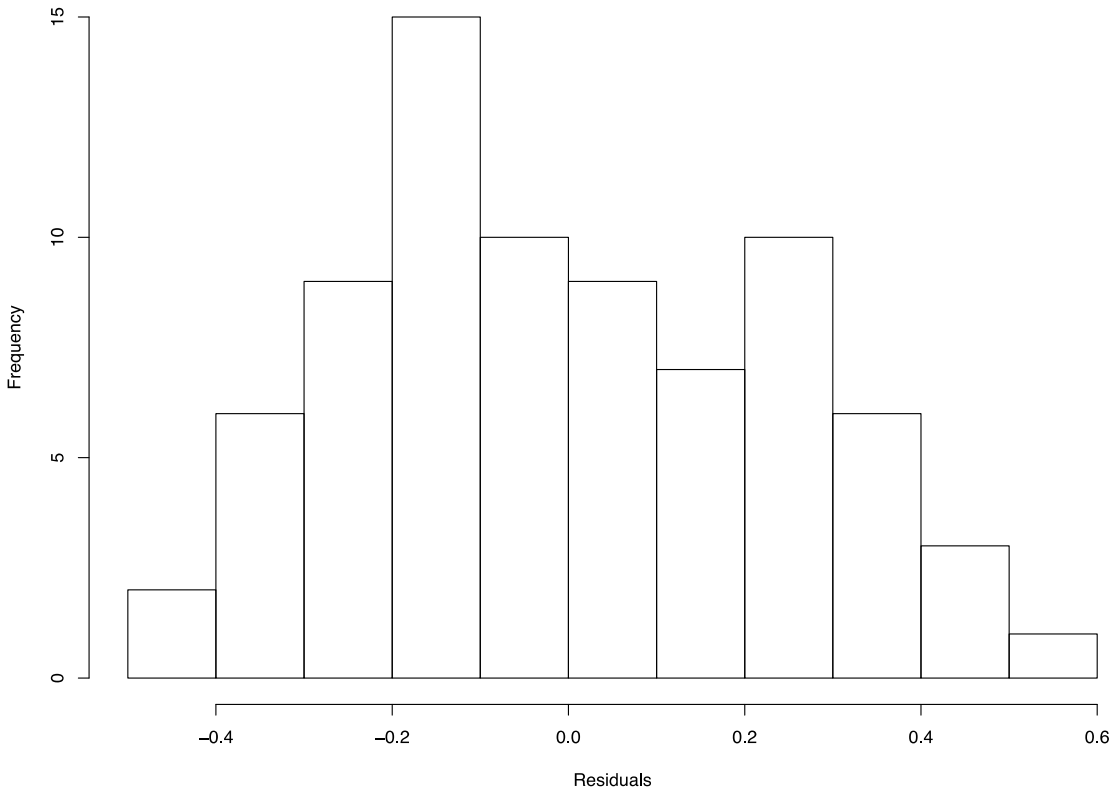

Figure B.2 Frequency histogram of the residuals to double check the normality of the model's residuals (A), further confirmed by a Shapiro-normality test (p-value = 0.216, thus considered normal).

APPENDIX C ARTICLE 3

Table C.1 Table showing the environmental variables from four matrices using the United States Environmental Protection Agency – National Lake Assessment (USEPA-NLA) dataset. The table shows the environmental variables used for forward selection and variation partitioning, a short definition of the variables, their respective units and the type of variable (N for numerical and F for categorical). From Trottier et al. (2021) with permission of the authors and the publisher under the Creative Commons Attribution License.

Matrix	Variable	Definition (ABBR.)	Units	Type
Spatial	Latitude	Latitude of reservoir (LAT)	Decimal degrees	N
	Longitude	Longitude of reservoir (LON)	Decimal degrees	N
	Ecoregion	National Aquatic Resources Surveys (NARS) 9-level reporting regions, based on aggregated Omernik (1987) level III ecoregions (ECO)	-	F
	Temperature	Annual mean air temperature, specific to ecoregion (TEMP)	°C	N
	Precipitation	Annual mean precipitations, specific to ecoregion (PREC)	mm	N
	Forested	Percentage of land cover in ecoregion that is forested (LC.FOR)	%	N
	Cultivated pasture	Percentage of land cover in ecoregion that is cultivated pastures (LC.CP)	%	N
	Wetlands	Percentage of land cover in ecoregion that is wetlands (LC.WET)	%	N
	Grassland and shrubs	Percentage of land cover in ecoregion that is grasslands and shrubs (LC.GS)	%	N
	Developed	Percentage of land cover in ecoregion that is developed (LC.DEV)	%	N
	Water or barren	Percentage of land cover in ecoregion that is water or barren (LC.WB)	%	N

Table C.1 (continued)

Matrix	Variable	Definition (ABBR.)	Units	Type
Physical	Area	Surface area of reservoir (AREA)	ha	N
	Elevation	Elevation of reservoir (ELE)	m	N
	Shallow water	Shallow water habitat condition indicator (quality of the shallow edge of the lake by using data on the presence of living and non-living features such as overhanging vegetation, aquatic plants, large woody snags, brush, boulders and rock ledges, based on aggregated visual observations; USEPA, 2016b) (LIT.CVR)	-	N
	Riparian vegetation	Riparian vegetation condition indicator (evaluation of riparian or lake based on observations of three layers of vegetation – understory grasses and forbs, mid-story non-woody and woody shrubs, and over story trees, based on aggregated visual observations; USEPA, 2016b) (RIP.VEG)	-	N
Chemical	Trophic.state	Trophic state of reservoir (oligotrophic and eutrophic) (TS)	-	F
	Secchi	Secchi depth (SECCHI)	m	N
	DOC	Dissolved organic carbon level (DOC)	mg/L	N
	PTL	Total phosphorus level (TPL)	µg/L	N
	Color	Water color (COLOR)	PCU	N
	Conductivity	Water conductivity level (COND)	µs/cm	N
	NTL	Total nitrogen level (TNL)	mg/L	N
	pH	pH level (PH)	pH scale	N
	Methylmercury	Top sediment methylmercury level (HG)	ng/L	N

Table C.1 (continued)

Matrix	Variable	Definition (ABBR.)	Units	Type
Chemical	Chl-α	Chlorophyll-α measurement result of reservoir (CHLA)	μg/L	N
Human	Buildings	Human influence by buildings around reservoir shoreline (if present in plot a value of 1 is attributed, if absent in plot a value of 0 is attributed and if adjacent to plot, a value of 0.5 is attributed and then summed for 10 sampling plots in each reservoir, same procedure for each type of human influence; BUILDINGS)	-	N
	Commercial	Human influence by commercial activities around reservoir shoreline (COMM)	-	N
	Crops	Human influence by row crops around reservoir shoreline (CROPS)	-	N
	Docks	Human influence by docks/boats around reservoir shoreline (DOCKS)	-	N
	Landfills	Human influence by landfill/trash around reservoir shoreline (LANDFILLS)	-	N
	Lawns	Human influence by lawn around reservoir shoreline (LAWNS)	-	N
	Parks	Human influence by parks around reservoir shoreline (PARKS)	-	N
	Pastures	Human influence by pastures around reservoir shoreline (PAST)	-	N
	Powerlines	Human influence by powerlines/utility lines around reservoir shoreline (POW)	-	N

Table C.1 (continued)

Matrix	Variable	Definition (ABBR.)	Units	Type
Human	Roads	Human influence by roads/railroads around reservoir shoreline (ROADS)	-	N
	Walls	Human influence by walls/bulkheads/revetments around reservoir shoreline (WALLS)	-	N
	Others	Human influence by other around reservoir shoreline (OTHERS)	-	N

Table C.2 Summary of statistical candidate models, at each taxonomic resolution (genus-exclusive [GEN.EXC], genus-inclusive [GEN.INC], family-exclusive [FAM.EXC] and family-inclusive [FAM.INC]). Akaike Information Criterion is ΔAIC, and Bayesian Information Criterion is BIC, PDF stands for Potentially Disappeared Fraction of genera or families, ELE for elevation, AREA for surface area, T.S. for trophic state, PARK for the influence of parks, PH for pH level, LAWN for influence of lawns, LIT.CVR for shallow water habitat condition, ROADS for influence of roads and POW for the influence of powerlines. For each candidate model, the estimate for the intercept is labelled b_{int} and all other bs (b_{ELE}, b_{AREA}, $b_{T.S}$, b_{PARK}, b_{PH}, b_{LAWN}, $b_{LIT.CVR}$, b_{ROAD}, b_{POW}), estimate for the slope of their respective variable. See Table C.1 for full description of the variables used and Table C.3 for variables estimates of models with highest support, Standard Error (SE) and p-values. Delta AIC calculated within each taxonomic resolution. Models in bold are model with highest support within their taxonomic resolution. *Marginally significant.

Taxonomic resolution	Models	Non-significant variables	AIC	deltaAIC	BIC	R^2_{ADJ}	P-VALUE	N
	(A) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PARK}*log10(PARK) + b_{PH}*PH + b_{LAWN}*log10(LAWN) + b_{LITCVR}*sqrt(LIT.CVR)	LAWN* and LIT.CVR	14	0	36	0,48	0	78
	(B) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PARK}*log10(PARK) + b_{PH}*PH + b_{LAWN}*log10(LAWN)	LAWN*	14	0	33	0,47	0	78
	(C) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PARK}*log10(PARK) + b_{PH}*PH	PH* and PARK*	16	1	32	0,46	0	78
GEN.EXC	**(D) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH**	**None**	**18**	**3**	**32**	**0,43**	**0**	**78**
	(E) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA)	None	25	11	35	0,36	0	78
	(F) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE)	None	30	16	38	0,31	0	78
	(G) $PDF_{RES} \sim b_{int}$	None	58	44	63	NA	NA	78
	(H) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH + b_{LAWN}*log10(LAWN) + b_{ROAD}*log10(ROAD)	PH*, LAWN and ROAD	8	2	27	0,51	0	78
	(I) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH + b_{LAWN}*log10(LAWN)	PH* and LAWN	8	1	24	0,51	0	78
	(J) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS + b_{PH}*PH	PH*	6	0	20	0,51	0	78
GEN.INC	**(K) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA) + b_{TS}*TS**	**None**	**8**	**2**	**20**	**0,49**	**0**	**78**
	(L) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{AREA}*log10(AREA)	None	14	8	23	0,45	0	78
	(M) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE)	None	25	19	32	0,35	0	78
	(N) $PDF_{RES} \sim b_{int}$	None	58	52	63	NA	NA	78

Table C.2 (continued)

Taxonomic resolution	Models	Non-significant variables	AIC	deltaAIC	BIC	R^2_{ADJ}	P-VALUE	N
FAM.EXC	(O) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + bPOW*log10(POW) + b_{TS}*TS + b_{LAWN}*log10(LAWN) + b_{AREA}*log10(AREA)	LAWN* and AREA	10	0	27	0,43	0	78
	(P) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS + b_{LAWN}*log10(LAWN)	**None**	**11**	**0**	**25**	**0,43**	**0**	**78**
	(Q) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS	None	13	3	25	0,4	0	78
	(R) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW)	None	21	10	30	0,33	0	78
	(S) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE)	None	29	19	36	0,25	0	78
	(T) $PDF_{RES} \sim b_{int}$	None	50	40	55	NA	NA	78
FAM.INC	(U) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS + b_{LAWN}*log10(LAWN) + b_{AREA}*log10(AREA)	LAWN* and AREA	14	0	31	0,42	0	78
	(V) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS + b_{LAWN}*log10(LAWN)	LAWN*	15	1	29	0,4	0	78
	(W) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW) + b_{TS}*TS	**None**	**17**	**3**	**29**	**0,38**	**0**	**78**
	(X) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE) + b_{POW}*log10(POW)	None	24	10	34	0,31	0	78
	(Y) $PDF_{RES} \sim b_{int} + b_{ELE}$*sqrt(ELE)	None	32	17	39	0,23	0	78
	(Z) $PDF_{RES} \sim b_{int}$	None	51	37	56	NA	NA	78

Table C.3 Variables estimates (EST) of models with highest support, Standard Errors (SE) and p-values (PVAL), for models built with raw data (which was transformed) or standardized data (which facilitates comparison of directionality and magnitude across models).

DATA TYPE	MODELS	INT			ELE			AREA			TS			PH			POW			LAWN		
		EST	*SE*	*PVAL*	*EST*	*SE*	*PVAL*	*EST*	*SE*	*PVAL*	*EST*	*SE*	*PVAL*	*EST*	*SE*	*PVAL*	*EST*	*SE*	*PVAL*	*EST*	*SE*	*PVAL*
Transformed	GEN.EXC (D)	-1,159	0,413	0,006	0,011	0,002	0,000	0,134	0,046	0,005	0,392	0,125	0,003	0,986	0,049	0,049						
Transformed	GEN.INC (K)	-0,454	0,102	0,000	0,013	0,002	0,000	0,170	0,044	0,000	0,325	0,115	0,006									
Transformed	FAM.EXC (P)	0,007	0,069	0,921	0,009	0,002	0,000				0,348	0,118	0,004				0,080	0,019	0,000	-0,059	0,028	0,040
Transformed	FAM.INC (W)	0,010	0,072	0,886	0,011	0,002	0,000				0,381	0,123	0,003				0,066	0,019	0,001			
Standardized	GEN.EXC (D)	0,269	0,029	0,000	0,167	0,032	0,000	0,086	0,030	0,005	0,097	0,031	0,003	0,063	0,031	0,049						
Standardized	GEN.INC (K)	0,270	0,028	0,000	0,199	0,029	0,000	0,109	0,028	0,000	0,080	0,028	0,006									
Standardized	FAM.EXC (P)	0,244	0,028	0,000	0,140	0,031	0,000				0,086	0,029	0,004				0,127	0,030	0,000	-0,069	0,033	0,040
Standardized	FAM.INC (W)	0,246	0,029	0,000	0,159	0,030	0,000				0,094	0,030	0,003				0,104	0,030	0,001			

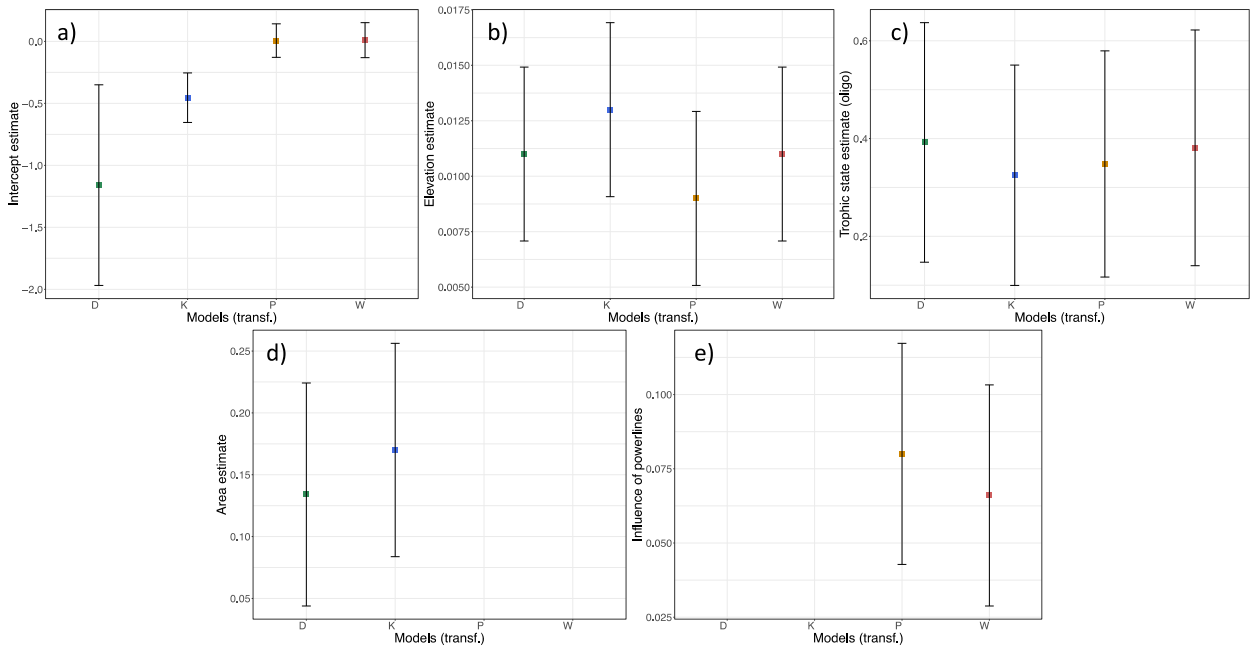

Figure C.1 Plots showing estimates (raw data transformed) from different variables across taxonomic resolutions models. Refer to Table C.2 for model identification (D, K, P and W). a) is intercept, b) is elevation, c) is trophic state, d) is area and e) is powerlines. In none of the cases are the estimates significantly different (ANOVA) between taxonomic resolutions.

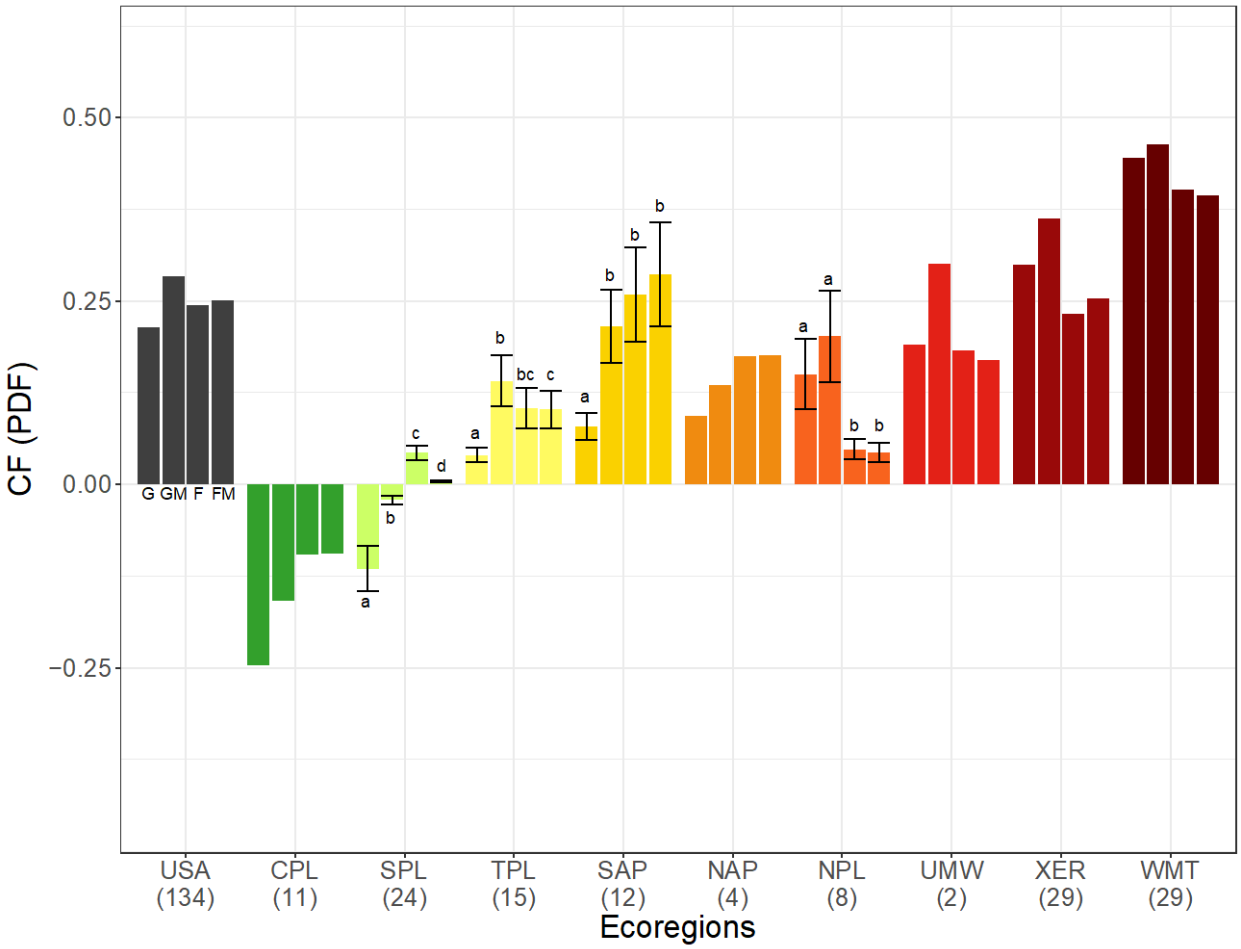

Figure C.S1 Barplot showing Potentially Disappeared Fraction (PDF) at the four categories taxonomic resolutions (respectively, genus-exclusive [G], genus-inclusive [GM], family-exclusive [F] and family-inclusive [FM]), for the United States (USA) and the nine ecoregions. PDF_{ECO} without error bars are not significantly different from each other, whereas ecoregion PDF_{ECO} with differing letters show statistical difference. Ecoregions are color-coded the same way as in the map, with the exception of USA in dark grey.

Table C.S1 Table showing the ANOVA output for the influence of taxonomic resolution on nine ecoregions PDF_{ECO} (United States [USA], Coastal Plains [CPL], Southern Plains [SPL], Temperate Plains [TPL], Southern Appalachians [SAP], Northern Appalachians [NAP], Northern Plains [NPL], Upper Midwest [UMW], Xeric [XER] and Western Mountains [WMT]), the numerator (df_{num}) and denominator (df_{denom}) dfs used to attribute F-statistics (a = 0.05), F-values obtained through statistical testing (ANOVA) and significance, where SD stands for significantly different and NSD for non-significantly different. If F-value is bigger than respective F-statistics, there is a significant influence of taxonomic resolution on PDF_{ECO} at the specified ecoregion.

Ecoregions	**df_{num}**	**df_{denom}**	**F-statistic**	**F-value**	**Significance**
USA	3	532	8.53	5.30	NSD
CPL	3	40	8.59	5.34	NSD
SPL	3	92	8.55	62.30	SD
TPL	3	56	8.57	9.95	SD
SAP	3	44	8.59	10.85	SD
NAP	3	12	8.74	1.60	NSD
NPL	3	28	8.62	14.54	SD
UMW	3	4	9.12	1.36	NSD
XER	3	112	8.55	3.57	NSD
WMT	3	112	8.55	0.66	NSD