Chapter 1

General Introduction

1.1 The unknown unknowns

It is widely agreed that human activities have left, to put it mildly, a soiled footprint around the globe, with widespread deforestation (Barbosa, Nabout, and Cunha 2023; FAO 2022; Pacheco et al. 2021), resource depletion (Oberle et al. 2019), soil erosion (Van Oost et al. 2007), water and air pollution (Fuller et al. 2022), as examples of extensively documented current environmental crises (Ceballos et al. 2015; Waters et al. 2016). Besides these strains, the total global population is expected to surpass 9 billion people by 2030, with an estimated increase of 30% of the population moving to urban areas (United Nations 2022). These growing demands from an increasing global population have not only exacerbated the degradation of natural resources, but also the challenges for a transition towards sustainable development (Kaiho 2023; Khorram-Manesh 2023). As a result, the United Nations published in 2015 a universal call for action focusing on a set of identified global goals, commonly known as the sustainable development goals (United Nations 2015). These goals describe humanity targets that aim at global peace, end of poverty and hunger, and among these, several pillars that are directly linked to the protection of the environment and the quality of natural resources (Yang et al. 2020; Yin et al. 2021).

To embrace these challenges, several sectors of society have mobilized the demand for the assessment of environmental impacts as an integral part of decision- and policy- making (Khorram-Manesh 2023), modifying current production systems to minimize negative impacts, and designing more resilient systems (Fiksel 2003; Wood et al. 2018). Assessing environmental impacts involves identifying and evaluating the potential effects of human activities on the natural environment, including air, water, soil, and wildlife (Qiu, Yu, and Huang 2022). By conducting a comprehensive assessment of environmental impacts, individuals, organizations, and governments can better understand the potential consequences of their actions and make informed decisions to mitigate or avoid harmful effects. This can help ensure that human activities are conducted in a manner that is compatible with the long-term health and well-being of the natural environment, which is essential for preserving biodiversity, protecting ecosystems, and maintaining the natural resources that sustain life on Earth (Barnosky et al. 2011).

1.2 The Ecosystem Service approach

Nowadays, the term ecosystem service is commonly used to address the natural resources and ecological processes that have been identified as beneficial for the sustenance of human wellbeing and the general interests of societies (Ainscough et al. 2019; Potschin and Haines-Young 2018). Several conceptual frameworks have been proposed to visualize the beneficial relations between ecosystems and society (Boyd and Banzhaf 2007; Fisher, Turner, and Morling 2009; Olander et al. 2021). In this thesis, I will refer to ecosystem services following the classification system proposed by the Common International Classification for Ecosystem Services (CICES), which presents three main categories (Figure 1.1): (i) provisioning services, which include resources directly obtained from ecosystems, such as biomass for food or materials, genetic resources and water; (ii) regulating services and maintenance, which are benefits obtained from the supporting ecological processes, including, but not limited to, climate regulation, soil erosion resistance, crop pollination, nutrient and water cycling; and (iii) cultural services, which includes the well-being and recreational benefits that people obtain from natural systems, such as knowledgeable systems, health and social relations, and aesthetic values (Haines-Young and Potschin 2018). Under other classification systems, such as the one presented by the Millenium Ecosystem Assessment (MEA 2005), regulatory and maintenance services are commonly listed as separate categories, with 'maintenance' services presented instead as 'supporting' services. However, these all refer to the same ecosystem services listed under a single category in the CICES classification, and which correspond to regulatory services, which help maintain and support human well-being and livable conditions (Mengist, Soromessa, and Feyisa 2020).

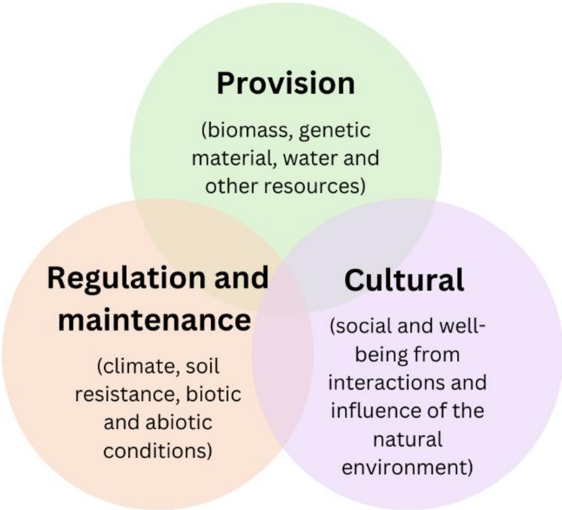

Figure 1.1 *The three main categories of ecosystem services as defined by the Common International Classification System of Ecosystem Services (CICES).*

The Millennium Ecosystem Assessment (MEA) was published in 2005 by the United Nations, as a result of a major international cooperation aimed at identifying, inventorying, and quantifying the state of multiple ecosystem services (Primmer et al. 2015). The results presented by the MEA report indicate that the majority of the ecosystem services identified showed severe degradation due to human activities, some to the degree of permanent or irreparable damage, and more than half currently managed under unsustainable practices (MEA

2005). In turn, the degradation of ecosystem services, which can be translated into social and economic damage, poses global risks for societies and human well-being (Costanza et al. 1998, 2014; Díaz et al. 2018; Van der Ploeg, De Groot, and Wang 2010).

Several factors, such as high demand of resources from an increasing urban population and short-term economic driven industrialization, have resulted in the unsustainable use and management of ecosystem services of all around the world (de Groot et al. 2012; Maes et al. 2012). Since the publication of the MEA report and the early economic valuation studies of ecosystem services (Costanza et al. 2014; de Groot et al. 2012; Reynaud and Lanzanova 2017) numerous efforts ranging from scientific to legislative, have focused on their assessment and protection (Aragão, Jacobs, and Cliquet 2016; McDonough et al. 2017), with the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) as a clear example of an initiative aimed at improving the understanding of ecosystem services and their interrelation with human activities and societies (Díaz et al. 2018; IPBES 2019).

To better assess impacts on ecosystem services, it is necessary to identify and quantify the mechanisms that drive them and the synergetic effects that can stimulate or hinder their provision (Perschke et al. 2023). Given the associated complexity of identifying key ecosystem services, and in particular those that are not directly linked to economic activities, many ecosystem services remain unaccounted for (Carrasco et al. 2014), with limited data available for a great part of those evaluated, and with several questions unresolved on the underlying processes that influence them and their effects on current and future generational needs (Ceballos et al. 2015).

Hence, effective policy and decision-making requires the aid of scientific research, by providing a better understanding and assessment of ecosystem services, and support guidelines towards more environmentally sustainable practices (Yin et al. 2021). In order to avoid 'cherry-picking' from a small sub-set of ecosystem services that could lead to negative consequences and unintended tradeoffs, it is important to concentrate efforts on the continuous identification and study of ecosystem services, and take on the challenge to model and quantify these systems and their dynamics (Schröter et al. 2017).

1.3 Life Cycle Assessment

Worldwide efforts to protect our natural environment have led to the development of different tools and environmental impact methods that allows us to estimate at different degrees, the direct and indirect effects of human activities. With most of the current production and trading systems operating with stakeholders located all around the world, this increasing trend highlights the need for methods that can operate with a global and systems level perspective. While several methods exist to trace material-flows linked to economic activities per country, such as material flow analysis (MFA) and environmental economic input output analysis (EEIOA), the method used worldwide to compare environmental impacts of product systems is known as Life Cycle Assessment (LCA). LCA has become an internationally standardized method that allows to estimate the environmental interventions throughout the life cycle of a product or service and translates them into potential impacts (ISO 2006). LCA is used across several sectors, to provide insights for decision making aiming at more sustainable consumption and production systems (O'Shea, Golden, and Olander 2013).

LCA studies can help determine and compare the environmental implications of systems that can range from technological advances to conventional practices looking to improve their environmental performance, and are increasingly being required by legislative bodies in the EU to be presented as part of the environmental profile for new products (European Commission 2021). Furthermore, the results can help identify 'hotspots' within a studied system (i.e., processes that contribute the most to a set of environmental impacts), providing opportunities to address polluting or highly impactful activities, as well as to compare and select alternatives that present a relatively better environmental performance (Heijungs et al. 2019; Mendoza Beltran et al. 2018). These advantages along with its systems thinking approach, has made LCA a valuable method in the transition towards sustainable practices.

As a direct result of its increased use worldwide, LCA has been the subject of intense and constant research over the last decade (Bare 2011; Curran et al. 2016; Koellner et al. 2013; Mutel et al. 2019; Nordborg et al. 2017; Yi, Kurisu, and Hanaki 2014). Although its framework has been standardized, the operationalization of LCA is constantly evolving along with the capacity of LCA software and the improvement of the LCI databases and impact methods used to estimate environmental impacts (O'Shea et al. 2013). This constant development is driven by the need to improve our understanding of environmental dynamics, as well as the reach and interpretation of LCA results to support decision-making.

1.3.1 Brief overview of the framework

The current LCA framework, as standardized by the ISO 14040 and ISO 14044 (ISO 2006a,b), is an environmental analysis that comprises four main phases (Figure 1.2): i) the goal and scope definition, in which the purpose of the study and the basis for comparison, i.e. the functional unit, are specified along with the system boundaries; ii) the inventory analysis, in which all processes needed to fulfill the functional unit and their inputs and outputs are determined along with the total emissions and resources associated with the product system(s) are compiled for each alternative; iii) the impact assessment, in which the resources and emissions are translated into environmental impacts, additional measures such as weighting or normalization can take place in this phase; and iv) the interpretation step, in which the robustness and completeness of the results can be analyzed, as well as measures of uncertainty and sensitivity analyses to help in the interpretation of results.

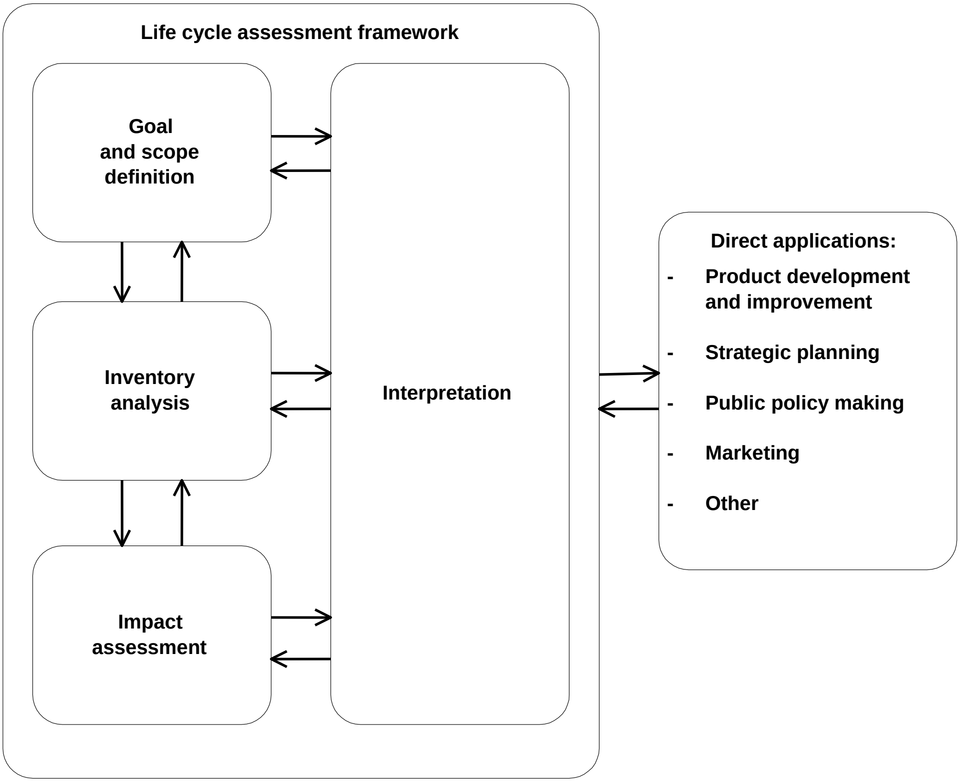

Figure 1.2 *The life cycle assessment framework. (ISO 2006)*

LCA studies consists of iterative rounds in which the goal and scope are constantly revisited, for example, to determine which processes are considered within system boundaries and to select the appropriate allocation methods to solve multifunctional processes (Guinée et al. 2002). Once the goal and scope have been determined, the inventory analysis takes place. ISO defines the inventory analysis (LCI) as the "phase of life cycle assessment involving the compilation and quantification of inputs and outputs for a product throughout its life cycle". To do this, quantitative data is compiled for each unit process relevant to the system assessed and within the selected system boundaries. According to ISO, a unit process is the "smallest element considered in the life cycle inventory analysis for which input, and output data are quantified". To compile life cycle inventory data, studies rely most of the times on third-party databases in an attempt to create a complete overview of the processes involved. An example of a widely used LCI database is ecoinvent, which contains information on over 18,000 activities related to diverse manufacturing practices, construction processes, energy systems, food production, transportation, among many other categories, covering globally generic and country specific scopes (Althaus et al. 2007; Wernet et al. 2016).

The LCI results contains information regarding all the elementary flows (i.e., environmental flows that correspond to resource inputs and emissions) associated with a functional unit. Thus, the relative output of each unit process is scaled for a whole system based on the functional unit defined. To further elaborate on this, we follow the standard computational nomenclature of the LCA matrix (Heijungs and Suh 2002):

$$\mathbf{A}\mathbf{s} = \mathbf{f}$$

and

$$\mathbf{s} = \mathbf{A}^{-1}\,\mathbf{f}$$

Where $\mathbf{A}$ is the technology matrix representing the flows within the economic system (with $\mathbf{A}^{-1}$ as its inverse), $\mathbf{f}$ is the final demand vector (which represents the reference flow of the system, i.e., the amount of product needed per functional unit) and $\mathbf{s}$ correspond to the scaling vector, which allows to determine vector $\mathbf{g}$, that relates the environmental flows and the economic system to its final demand, expressed as:

$$\mathbf{g} = \mathbf{B}\mathbf{s}$$

where $\mathbf{B}$ is the intervention matrix representing the environmental flows of all unit processes associated with a product system (Heijungs and Suh 2002). The expression to calculate the final inventory results, aggregated over the entire product system and across a life cycle, is the following:

$$\mathbf{g} = (\mathbf{B}\mathbf{A}^{-1})\,\mathbf{f}$$

where the inventory may be solved for a variety of final demands $\mathbf{f}$ (Heijungs and Suh 2002). Following the inventory analysis, the impact assessment phase takes place, which is aimed at “understanding and evaluating the magnitude and significance of the potential environmental impacts for a product system”(ISO 2006). To do this, the LCI results (i.e., the inventory of emissions and resources compiled for each system and scaled per functional unit) are usually converted into potential impacts by an impact characterization step:

$$\mathbf{h} = \mathbf{Q}\mathbf{g}$$

where $\mathbf{h}$ is the impact vector, and $\mathbf{Q}$ is the matrix of characterization factors (CFs). This linear expression represents the contribution of $\mathbf{g}$ to a given impact category (Heijungs and Suh 2002). More recent studies are targeting the development of non-linear approaches to introduce more complex dynamics within the common LCIA framework (Arbault et al. 2014; Li et al. 2020; Pizzol et al. 2020). As addressed in Heijungs and Suh (2022): "the matrix-based approach should be regarded as a convenient and simplified approach, which is subject to further innovation and added complexity as necessary". Although non-linear and dynamic approaches present promising though also challenging avenues for future research, they remain a minority in current LCIA literature and limited to niche applications in LCA studies. Therefore, this thesis will focus for now on the common linear use of characterization factors (Heijungs 2020).

1.3.2 Characterization factors, what are they?

As previously mentioned, characterization factors (CFs) are used to convert the LCI results into indicator results. CFs are numerical values derived from characterization models that quantify the potential effect of environmental interventions to a certain impact category (e.g., climate change, eutrophication). The CFs are usually provided by developers in the form of an ordered list of data (or a spatially explicit map), specific to the LCI results assigned to impact categories in the classification step (Heijungs and Suh 2002). They are typically derived from models that take into account the environmental fate and effects of a substance, emission or resource use. To provide a brief example, to calculate the global warming result of a product system, the relevant LCI data of the system would be multiplied by the corresponding characterization factors for each greenhouse gas emitted (the so-called global warming potentials or GWPs), such as carbon dioxide, methane, and nitrous oxide. The resulting values would be summed to obtain a single value for the impact category of climate change.

Overall, characterization factors play a critical role in enabling the comparison of the potential environmental impacts that can help inform decision-making towards more sustainable options. CFs for LCA studies can be found for either one of two types, firstly for ‘midpoints’, where the characterized impact lies somewhere along the ecological cause-effect pathway, or secondly, for ‘endpoints’, where damages linked to at least one of the three areas of protection is assessed (i.e., human health, ecosystem quality and resource scarcity) (See Figure 1.3). The selection between midpoint or endpoints to compare between product systems will largely depend on the purpose of the study and the preference of the LCA practitioner. While the midpoint approach usually involves less debatable assumptions and targets ecological effects, the end-point approach can involve higher uncertainties but provides more ‘intuitive’ metrics that can be more easily interpreted for decision making (Guinée and Heijungs 2017). However, both levels of characterization can complement each other and provide information regarding the ecological effects of the studied system and their influence on human health and environmental quality (Hacikamiloglu 2007).

CFs are usually compiled in families of impact assessment methods such as Recipe2016 (Huijbregts et al. 2016) and IMPACT world+ (Bulle et al. 2019), some of which present both midpoint and endpoint CFs, with a baseline covering approximately 17 midpoint categories, and endpoints that link to at least one of the 3 AoP: human health, natural environment, and natural resources (See Figure 1.3).

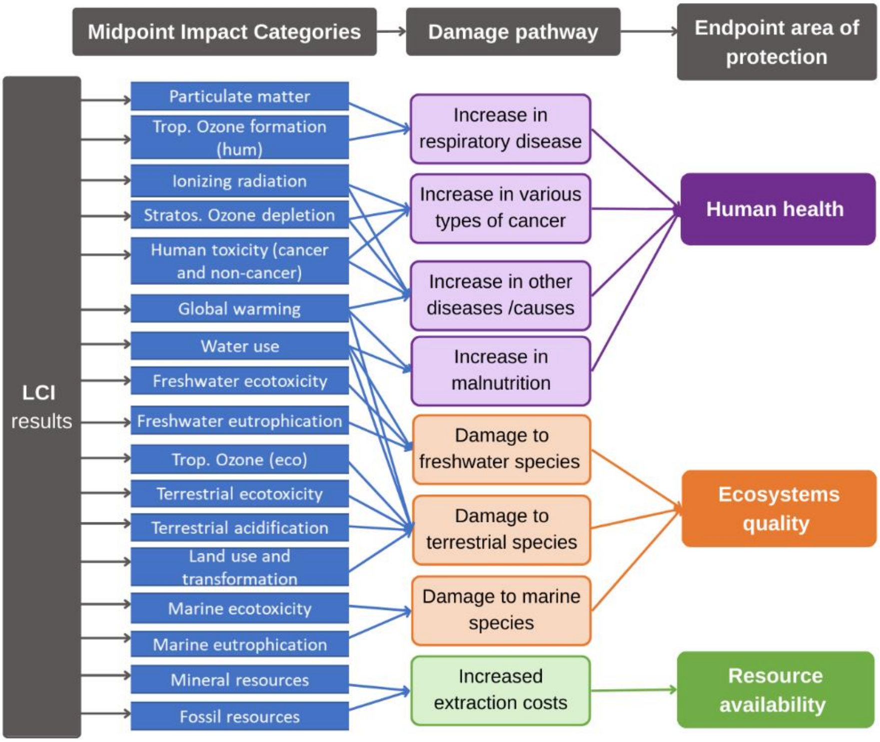

Figure 1.3 *Example of impact categories from ReCiPe2016* (Huijbregts et al. 2016). *During the impact assessment phase of LCA, LCI results are translated into midpoint impact indicator results, and through endpoints linked to at least one of the Areas of Protection.*

1.4 Assessing ecosystem service impacts in LCA

Until this past decade, the explicit mention of ecosystem services was completely missing from the LCA literature. This changed after the first review published by Zhang et al. (2009), who presented an overview of the possibilities for integrating the concept of ecosystem services in LCA studies (Zhang, Sing, and Bakshi 2010; Zhang, Singh, and Bakshi 2009). Since then, several authors have proposed different recommendations, some analyzing the opportunities from a conceptual approach (Bakshi and Small 2011; Dewulf et al. 2015; Maia de Souza et al. 2018; Rugani et al. 2019; Xinyu, Ziv, and Bakshi 2018), while others have worked on methods that can combine in some cases economic data (Cao et al. 2015) as well as emergy and exergy values (Rugani et al. 2013). Furthermore, several studies have tried to tackle the spatial variation aspects that influence ecosystem services, especially those related to land use (Arbault et al. 2014; Chaplin-Kramer et al. 2017; Karabulut et al. 2016). Unfortunately, a lot of these approaches are limited in applicability, sometimes due to the fact that they deviate significantly from the LCA framework or common LCA practices, which can ultimately hinder a wide implementation of the methods proposed.

Aiming for the assessment of ecosystem services within common LCA practices, another approach has been followed by studies that focused on the development of characterization factors that can be used in the impact assessment phase of LCA (Brandão and I Canals 2013; Saad, Koellner, and Margni 2013; Schmidt 2008; Seppälä et al. 2006). Addressing the impact assessment phase, Othoniel et al. (2016) presented a comprehensive overview of the challenges of incorporating the explicit assessment of ecosystem service within LCA, and clearly explained some of the limitations encountered for the adaptability of existing methods and incompatibilities of jargon that can lead to divergent views on ecosystem service analysis and interpretation.

The debate around jargon incompatibilities seems to center most of the times on whether we are assessing impacts on ecological processes linked to the supply of ecosystem services, or assessing the impacts on the supply of the benefits themselves (Othoniel et al. 2016). The first case presents more compatibilities for incorporation within common LCA practices, while the second one is challenging due to intrinsic characteristics of supply and demand functions, such as site dependency and temporal dynamics (Othoniel et al. 2019). For the first, where the impact is assessed at one point within the ecological cause-effect chain, the development of characterization factors presents valuable opportunities to incorporate new impact categories in LCA that can be directly linked to key ecosystem services (Kumar, Esen, and Yashiro 2013). International initiatives, such as the UNEP-SETAC, have proposed general guidelines and recommendations for the characterization of ecosystem services and biodiversity impacts to promote harmonized efforts (Koellner et al. 2013; Rugani et al. 2019; Verones et al. 2017). However, further research is needed to achieve an extended and successful incorporation of new impact categories that can be directly linked to ecosystem services, allowing for a more comprehensive overview of key environmental impacts (Callesen 2016) .

1.5 Problem identification

Despite the increasing evidence on the relevance of ecosystem services, their assessment in LCA studies remains limited to a handful of categories, most of them assessing indirectly, the potential impacts on identified ecosystem services. In order to increase their coverage in LCA studies, further development of impact assessment methods is needed. As mentioned in previous sections, two main challenges have been identified as hindering the development and successful implementation of new impact categories targeting ecosystem services in LCA. Both challenges are related to the compatibility with common LCI data and conventional LCIA practices. Impact assessment models targeting ecosystem services are usually complex, non-linear models that require high spatially detailed input data, while LCI data is often geographically coarse, with countries as the maximum level of geographical specificity presented for most unit processes. Reconciling both the compatibility of characterization models and characterization factors with common LCI data and LCIA practices, is of high relevance to allow for a practical implementation of the methods proposed. Although the development of characterization factors and the application of the LCA method are usually independent activities, these should not be carried out in disregard of each other, as these crucial mismatches between the specificity of the CFs and the available inventory data can limit the application of new impact categories to only a few specific LCA studies. Furthermore, there is yet no clear guidance on which ecosystem services should be targeted for incorporation in LCA studies, leading to the overarching questions: what would a comprehensive coverage of ecosystem services in LCA entail, and how can we overcome the identified challenges?

1.6 Research objectives of the thesis

This thesis extends the body of knowledge aimed at the incorporation of ecosystem service impacts in LCA studies. The objectives of this thesis are addressed in the following research questions:

RQ1: *Based on the current impact assessment methods available and the existing ecosystem services identified, what would be the optimal coverage of ecosystem services in LCA studies?*

RQ2: *To incorporate the assessment of new ecosystem services in LCA, how can we reconcile the differences that exist between ecosystem service methods, the LCA framework and available LCI data?*

RQ3: *How can we address key data gaps to produce readily applicable characterization factors for the assessment of ecosystem services in LCA?*

RQ4: *How can we increase the representation of intra-national differences that are relevant for ecosystem services, in country-specific characterization factors?*

1.7 Outline of the thesis

Following the research questions (Figure 1.4), this thesis has been organized starting with one introductory chapter (Chapter 1), followed by four content chapters (Chapters 2-5), and one concluding chapter (Chapter 6). In Chapter 2, I present an overview of the impact categories included in common impact assessment methods, identifying the ecosystem services that are directly and indirectly included. Parting from there, I compared the results with those ecosystem services included in inventories by CICES to identify the ones currently missing, and that should be the target for an optimal coverage in environmental LCA studies. From the ecosystem services identified as missing, I used available economic valuation data to provide a sense of perspective on the potential costs of neglecting their assessment and protection.

In Chapter 3, I tackled one of the key ecosystem services identified as missing from commonly used impact methods, to propose an approach that can allow for the characterization of impacts in a compatible way with available LCI data.

A review of impact assessment models from diverse disciplines targeting the selected ecosystem services was conducted in order to determine, from those available, which ones could be applicable for LCA and propose the required adaptations. The impact assessment model proposed is illustrated first with exemplary characterization factors produced in conjunction with an expert from the field of the ecosystem services studied.

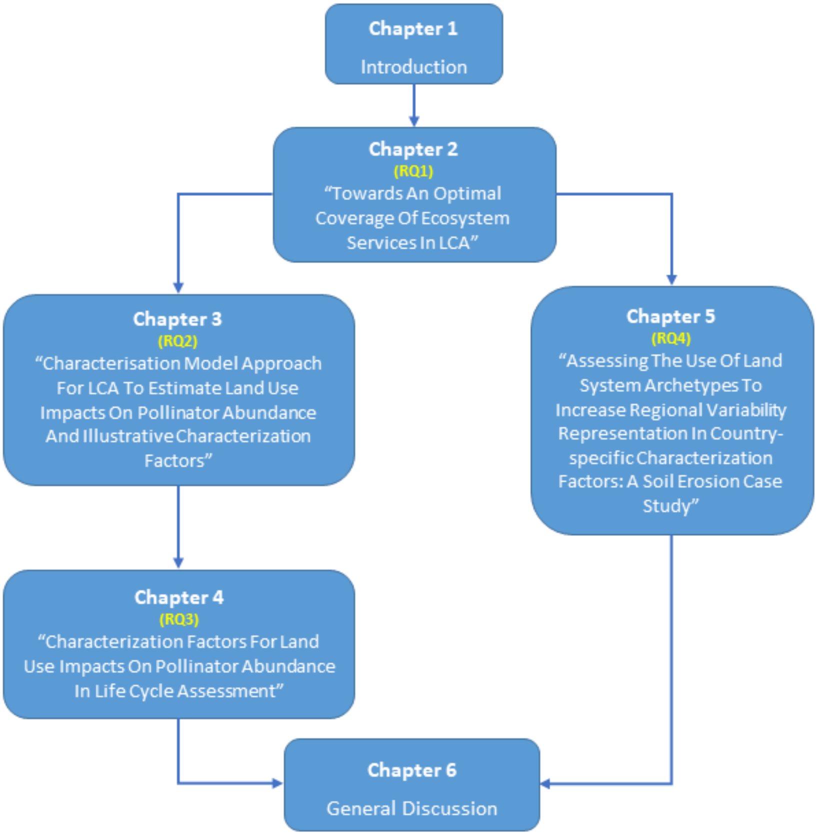

Figure 1.4 *Outline of this thesis.*

To move from illustrative to readily applicable characterization factors, I present in **Chapter 4** the procedure and results of an interdisciplinary collaboration with 25 expert researchers from around the world, that resulted in the derivation of the first set of characterization factors that allows to translate land use impacts on wild pollinators. Through the use of an expert elicitation method, I retrieved representative data pertaining to the field of the ecosystem service studied, by presenting a useful way to fill in knowledge gaps for characterization of impacts.

Lastly, given the current limitations on geographical specificity in LCA studies and the high relevance of biogeographical differences for many ecosystem services, I explored in **Chapter 5** how to improve the representation of intranational differences when producing country-specific characterization factors. This was done by applying land system archetypes derived from clustering techniques that combine both biogeographical and socioeconomic factors, to produce CFs that can represent the high diversity of impacts associated with site-dependent ecosystem services, such as is the case for the soil erosion resistance capacity. Previous studies had produced country-specific CFs for soil erosion based solely on biogeographical parameters and using the potential natural vegetation (PNV) as a reference state. In this chapter I produced CFs using information from land system archetypes as an alternative reference state, to compare our results with previous studies and challenge common practices that could potentially hinder the representation of key intranational variations.

In **Chapter 6**, I present a general discussion highlighting the main findings, addressing the limitations of our research as well as the challenges and opportunities for future researchers looking to dive into the topic of impact characterization of ecosystem services. The outcomes of this thesis are expected to provide useful insights not only on a viable way to expand the coverage of key environmental impacts in LCA studies, but also on the importance of interdisciplinary collaboration as an essential pillar for environmental research.

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