Technische Universität Berlin

Master's Thesis

The Economics of Planetary Boundaries: The Role of Biodiversity Loss for the Integrated Assessment of Climate Change

Tommaso Ficara

supervised by:

Prof. Dr. Ottmar Edenhofer and Dr. Martin Hänsel

Abstract

In this thesis, we explore the role of climate change-driven biodiversity loss for optimal climate policy within a dynamic cost-benefit analysis framework and evaluate the sustainability of the novel identified trajectories by assessing their compliance with the Planetary Boundaries approach from Rockström et al. (2009). We conduct our analysis by extending the dual-good social welfare function from the Drupp and Hänsel (2021) amended version of the Dynamic Integrated Climate-Economy model (DICE) and disentangling the comprehensive non-market good consumption component via a constant elasticity of substitution (CES) utility function to include for the consumption of biodiversity-related ecosystem services and other non-market goods, such as health and culture explicitly. In the framework we propose, we include a climate change-driven biodiversity loss function calibrated according to species' extinction meta-analysis from Urban (2015). The decline in biodiversity due to global warming affects the capability of ecosystems to provide fundamental ecosystem services to society and therefore impact aggregated utility levels. Throughout this thesis, the analysis of relative price changes of the explicitly included non-market goods takes central importance. As relative prices changes between goods are given by the difference in growth rates times the elasticity of substitution, we investigate whether assuming limited substitutability between the non-market amenities and considering the heterogeneity in growth rates leads to more stringent climate policy outcomes compared to a case where the goods are perfectly substitutable or included in the model in an aggregated form. Given the importance of the substitutability assumption, we conduct a literature review to introduce arguments in support to both high and low substitution possibilities between biodiversity-related ecosystem services and other non-market goods. In the analysis, we find that considering the changes in relative scarcities and assuming limited substitutability between biodiversity-related ecosystem services and other non-market goods leads to 9.6% (11.6%) higher Social Costs of Carbon in 2020 (2100) compared to a case where substitution possibilities between these amenities are perfect. In the following, building on the dual discounting literature, to correct for the imbalance generated by relative price changes, we propose a three-good discounting approach. Concluding we evaluate to which extent the novel identified optimal climate policy trajectories comply with the climate change and biosphere integrity boundary by measuring the projected CO_2 concentration pathways and the estimated climate change-driven extinction rates. We find that acknowledging limited substitution possibilities between biodiversity-related ecosystem services and other non-market goods represents a step forward in meeting the constraints of the planetary boundaries, but it is not sufficient to lead society to the "safe operating space".

Zusammenfassung

In dieser Masterarbeit untersuchen wir die Rolle des durch den Klimawandel verursachten Biodiversitätsverlustes für eine optimale Klimapolitik im Rahmen einer dynamischen Kosten-Nutzen-Analyse und bewerten die Nachhaltigkeit der neu identifizierten Pfade, indem wir ihre Übereinstimmung mit dem Ansatz der planetarischen Grenzen von Rockström et al. (2009) bewerten.

In der vorliegenden Analyse erweitern wir die Zwei-Güter-Wohlfahrtsfunktion aus der überarbeiteten Version des Dynamic Integrated Climate-Economy model (DICE) von Drupp und Hänsel (2021) und entkoppeln die umfassende Nicht-Marktgüter-Konsumkomponente über eine Nutzenfunktion mit konstanter Substitutionselastizität, um den Konsum von biodiversitätsbezogenen Ökosystemleistungen und anderen Nicht-Marktgüter wie Gesundheit und Kultur explizit einzubeziehen. Darüber hinaus beziehen wir eine Biodiversitäts-Verlustfunktion in das Modell ein, die anhand der Meta-Analyse des Artensterbens von Urban (2015) kalibriert wurde. In dem von uns entwickelten Modell wirkt sich die durch die Klimaerwärmung bedingte Abnahme der Biodiversität auf die Fähigkeit der Ökosysteme aus, grundlegende Ökosystemleistungen für die Gesellschaft zu erbringen, und hat somit negative Auswirkungen auf das aggregierte Nutzenniveau.

In der zugrundeliegenden Arbeit nimmt die Analyse der relativen Preisänderungen der explizit einbezogenen, Nicht-Marktgüter zentrale Bedeutung ein. Da die relativen Preisänderungen zwischen den Gütern durch die Differenz der Wachstumsraten mal der Substitutionselastizität gegeben sind, untersuchen wir, ob die Annahme einer begrenzten Substituierbarkeit zwischen den Nichtmarktgütern und die Berücksichtigung der Heterogenität der Wachstumsraten zu strengeren klimapolitischen Ergebnissen führt als in dem Fall, in dem die Güter perfekt substituierbar oder in aggregierter Form in das Modell einbezogen sind. Angesichts der Bedeutung der Substituierbarkeitsannahme, führen wir einen Literaturüberblick durch, um Argumente für hohe und niedrige Substitutionsmöglichkeiten zwischen biodiversitätsbezogenen Ökosystemleistungen und anderen nicht-marktbestimmten Gütern zu finden. In unserer Analyse stellen wir fest, dass die Berücksichtigung der relativen Preisänderungen und die Annahme einer begrenzten Substituierbarkeit zwischen Biodiversitäts-bezogenen Ökosystemleistungen und anderen, nichtmarktbestimmten Gütern zu 9,6 (11.6) Prozent höheren sozialen Kohlenstoff-Kosten im Jahr 2020 (2100) führt, verglichen mit dem Fall, dass Substitutionsmöglichkeiten zwischen diesen Gütern perfekt sind. Um das durch die relativen Preisänderungen entstandene Ungleichgewicht zu korrigieren, schlagen wir im Folgenden einen Drei-Güter-Diskontierungsansatz vor. Abschließend bewerten wir, inwieweit die neu identifizierten optimalen klimapolitischen Pfade mit den planetarischen Grenzen des Klimawandels und der biosphärischen Integrität kompatibel sind, indem wir die prognostizierten CO_2-Konzentrationspfade und die geschätzten klimawandelbedingten Aussterberaten messen. Wir kommen zu dem Ergebnis, dass die Anerkennung begrenzter Substitutionsmöglichkeiten zwischen Ökosystemleistungen, die mit der biologischen Vielfalt zusammenhängen, und anderen nicht-marktbestimmten Gütern einen Schritt nach vorn darstellt, um die Beschränkungen der planetarischen Grenzen zu erfüllen, aber nicht ausreicht, um die Gesellschaft in die Sicherheitszone zu führen.

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Glossary

BII	Biosphere Intactness Index
CES	Constant Elasticity of Substitution
DICE	Dynamic Integrated Model of Climate and Economy
E/MSY	Extinctions per Million Species per Year
GHG	Greenhouse Gases
IAM	Integrated Assessment Model
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IPCC	Intergovernmental Panel on Climate Change
MEA	Millennium Ecosystem Assessment
RPE	Relative Price Effect
PPM	Parts per Million
SCC	Social costs of Carbon
WTP	Willingness to Pay

1 Introduction

Anthropogenic climate change represents one of the main drivers of biodiversity loss (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, IPBES 2019) and threatens the capability of ecosystems to provide fundamental services to society (Hoppe et al. 2018).

In this context, the influential *Dasgupta Review* (Dasgupta 2021) offers high-level policy guidance to contrast the disruption of the Biosphere and provides a disturbing picture of the current state of biodiversity. A key message is the boundness of economic growth in the planetary boundaries, a sustainability concept introduced by Rockström et al. (2009). The planetary boundaries framework identifies the critical thresholds of nine fundamental biophysical processes that define 'the safe operating space' where society can pursue economic growth and prosperity. Transgressing the thresholds might lead the Earth System to a much less stable environment and pose a severe threat to societies' development (Steffen et al. 2015).

In this framework, climate change and biodiversity loss are of central importance. In fact, due to their deep interrelatedness with all other biophysical processes, the climate change and biosphere integrity boundaries (the latter accounting for biodiversity loss), are defined as 'core boundaries' (Lade et al. 2020). Therefore, maintaining the stability of these processes is a key prerequisite for the prospering of societies (Steffen et al. 2015).

Despite the important contribution to the sustainability discipline, the planetary boundaries framework does not consider the economic aspects of maintaining society in the domains of the safe operating space. In this regard, a central tool for economists to evaluate the costs, the benefits, and the trade-offs of climate policies are Integrated Assessment Models (IAMs), which combine findings from economics, ecology, and physics into single analytical frameworks (Nordhaus 2013). The Dynamic Integrated Climate-Economy (DICE) model, built on an extended Ramsey neoclassical economic growth model, is one of the most used IAMs by governmental bodies and scholars in climate policy (Nordhaus 2017).

One key metric produced by DICE, used to inform carbon pricing initiatives, are the Social Costs of Carbon (SCC), defined as 'the change in the discounted value of economic welfare from an additional unit of CO_2 - equivalent emissions' (Nordhaus 2017, 1).

However, climate change impacts on biodiversity are underrepresented in IAMs (Brooks and Newbold 2014) and are therefore not accounted for in most of the SCC estimates. Biodiversity is not included in climate change cost-benefit analysis primarily for two reasons: firstly, the physical impacts of global warming are difficult to assess and secondly, these amenities do not have a market price and assigning an economic value to these amenities is therefore a challenging task (Tol 2002). Nevertheless, according to Kaushal and Navrud (2018), the omission of climate change-driven biodiversity loss damages in IAMs could lead to an underestimation of the SCC and the resulting regulatory measures.

A central aspect in intertemporal climate change cost-benefit analyses is that most of the damage from global warming will only occur in the future and, therefore, must be discounted to be comparable to today's costs and benefits (Stern 2007). In this context, recent studies have criticised the use of a uniform social discount rate to address the evaluation of intergenerational costs and benefits of climate change mitigation (Hoel and Sterner 2007; Sterner and Persson 2008; Drupp 2018; Drupp and Hänsel 2021). These studies argued that, if the output composition of the economies is destinated to change due to heterogeneous sectoral growth rates, and if future comprehensive consumption is not evaluated in terms of relative prices reflecting increases in relative scarcities between different goods, the choice of a single discount rate will result as inadequate. By amplifying the valuation of climate change damages, relative price increases might, in fact, nullify or even reverse the discounting effect (Sterner and Persson 2008; Drupp and Hänsel 2021). This has been proven to be true in the presence of limited substitutability between the goods, one of the main drivers of relative price increases (Drupp and Hänsel 2021). On the other hand, with perfect substitutability, relative prices would be constant over time, and economic damages from climate change, regardless of the kind, compensable with increases in consumption (Sterner and Persson 2008).

The literature has proposed two approaches to correct for the imbalance of relative price changes in the long run. The first one consists in applying good-specific discount rates, whilst the second approach computes comprehensive consumption by taking into account relative price changes and discounts the aggregated consumption bundle with a single discount rate (Drupp and Hänsel 2021; Baumgärtner et al. 2015).

The DICE model (Nordhaus 2017), featuring only a comprehensive consumption variable in the social welfare function, accounts only in an implicit form for relative price changes (Drupp and Hänsel 2021). Drupp and Hänsel (2021) extended the DICE model by disentangling the comprehensive consumption variable in the social welfare function via a CES utility function to account for the explicit consumption of market and non-market goods and to study the impacts of non-market amenities’ relative price changes on climate policy optimality. According to the study, accounting for non-market goods’ relative price changes leads to more than 56 percent (81) higher SCC in 2020 (2100).

In this thesis, we build on the model specification from Drupp and Hänsel (2021) and propose an extended model that further disaggregates the heterogeneous non-market consumption variable from Drupp and Hänsel (2021) into the explicit consumption of biodiversity-dependent ecosystem services and other non-market goods (health, leisure, culture) via a nested CES utility function and considers economic damages from biodiversity loss an important driver of climate policy optimality.

We integrate the negative impacts of temperature on biodiversity in DICE via a climate change-driven biodiversity loss function calibrated according to the species' extinction predictions from the Urban (2015) meta-analysis. In the model we present, biodiversity represents an enabling asset of ecosystems that provide essential services that benefit society according to the schemes of the cascade model (Potschin and Haines-Young 2010). In the study of relative price changes that we conduct in this thesis, the substitutability assumption between biodiversity-related ecosystem services and other non-market goods plays a central role. Thus, it is imperative to assess to which extent the anthropogenic disruption of biodiversity and ecosystem services can be tolerated due to their close interrelatedness with essential drivers of human well-being, such as health and culture (Millennium Ecosystem Assessment 2005). In this context, the recent COVID-19 pandemic seems to be related to the destruction of habitats where climate change can be considered one of the primary drivers (Lorentzen et al. 2020).

By disaggregating the comprehensive non-market good consumption variable in Drupp and Hänsel (2021) specification, we analyse the implications of limited substitutability between biodiversity-related ecosystem services and other non-market goods on optimal climate policy and provide updated SCC estimates. In addition, we evaluate the compliance of the novel identified optimal climate policy trajectories with the limits identified by the climate change and biosphere integrity boundaries.

The thesis is structured as follows: Section 2 presents the planetary boundaries framework with particular attention to the climate change and biodiversity loss boundaries and investigates their interrelationship from a physical basis. In Section 3, we introduce the ecological-economic aspects that link biodiversity and human well-being according to the “cascade model” schemes and conduct a literature review to attribute an economic value to biodiversity-related ecosystem services. Next, in Section 4, we introduce the importance of the substitutability concept in the context of climate change cost-benefit analyses and present a literature review offering arguments to support both a high and low substitutability between biodiversity-related ecosystem services other non-market goods. In Section 5, after introducing IAMs and their main features, we present the DICE with a focus on the model components that will be relevant for the implementation of the amendments. In Section 6, we present the model extensions building on Drupp and Hänsel (2021) and according to the findings of the previous sections. In Section 7, we analyse the sensitiveness of optimal climate policy output depending on the substitutability assumption between the explicitly represented non-market goods and evaluate the compliance of optimal climate policy trajectories with the climate change and biosphere integrity boundaries. After having presented a summary of the results, this work concludes with a discussion on the limitations of the analysis.

2 Climate Change-driven Biodiversity Loss in the Planetary Boundaries Framework

Since its introduction, the planetary boundaries concept has attracted much attention from global policymakers to inform efforts towards a more sustainable society (Lade et al. 2020). The sustainability approach is based on the precautionary principle, which in this framework discourages world societies from leading the biophysical processes away from the stability state known in the Holocene; crossing the boundaries might have catastrophic consequences for life on Earth (Rockström et al. 2009).

In the planetary boundaries framework, the climate change and biosphere integrity boundaries are defined as "core boundaries" (Lade et al. 2020). The health status of these biophysical processes reflects the general conditions of the planet, and abrupt changes in these processes alone would destabilize the Earth System (Steffen et al. 2015). Therefore, addressing the degradation of these systems becomes of key importance for society's safety.

As of today, the biosphere integrity boundary has been classified as 'beyond zone of uncertainty' (high risk) (European Environment Agency 2019). The boundary accounts for the fundamental role of the biosphere in preserving the functioning of the Earth System (Steffen et al. 2015), and is measured via a dual approach.

Firstly, the genetic uniqueness of species is accounted in the boundary by using extinction per million species per year (E/MSY) as a control variable, which indicates the estimated total number of extinctions per year if the totality of species were one million.

The boundary is set at <10 E/MSY, which corresponds to ten times the natural extinction rate, and the current value of the control variable is estimated between 100-1000 E/MSY (Steffen et al. 2015). In general, biodiversity is difficult to quantify, and the choice of species' extinction rate as a control variable has received a good amount of criticism (Mace et al. 2014). Indeed biodiversity can be measured in different ways: modern ecological research primarily employs the number of species, phylogenetic, genetic, and phenotypic diversity, abundance-weighted species richness or evenness of species distribution (Paul et al. 2020).

In a recent stage of the Holocene, often referred to as Anthropocene, humans have boosted the rate of species extinction between a hundred and a thousand times the rates that were normal over the History of Earth (Mace et al. 2005): this extinction event is known as the sixth largest event taking place on Earth (Ripple et al. 2017; Dirzo et al. 2014; Ceballos and Ehrlich 2018). Moreover, IPBES (2019) estimates that circa one million species could face extinction in the upcoming decades due to climate change, land and sea use change, pollution, and invasion of alien species.

Secondly, the Biosphere Intactness Index (BII) accounts for the role of the biosphere in the Earth System functioning and is currently under development (Steffen et al. 2015; Dasgupta 2021). Due to great gaps in the knowledge on the relationship between BII and Earth System responses, the boundary is set at 90% of BII with a great error range (ranging from 90% to 30%) (Steffen et al. 2015).

The climate change boundary is set at a collar between 350 and 550 parts per million (ppm) of carbon dioxide concentration in the atmosphere, and its objective is to prevent temperature increases higher than 2°C pre-industrial (Rockström et al. 2009). In 2017, anthropogenic induced warming reached 1°C (in the range of 0.8°C-1.2°C) compared to pre-industrial levels, accelerating by 0.2 degrees per decade (Intergovernmental Panel on Climate Change, IPCC 2018). In 2019 CO_2 concentration levels in the atmosphere were about 409.8 ppm, and the boundary was therefore classified as 'in zone of uncertainty' (increasing risk) (European Environment Agency 2019).

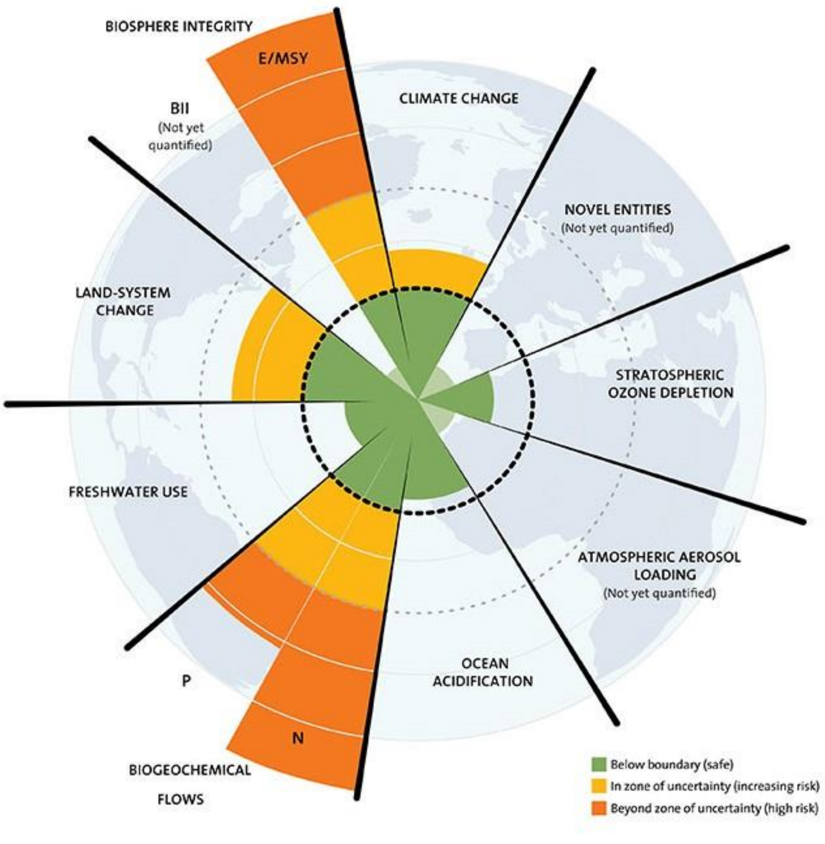

Figure 1: The nine Planetary Boundaries
Source: Steffen et al. (2015)

Despite its guidance for sustainable development the planetary boundary approach does not focus on the interdependencies between Earth Systems (Lade et al. 2020). Moreover, it does not consider welfare economics aspects that properly weigh the trade-offs between heterogeneous stabilisation goals. Nevertheless, preserving the stability of the biosphere will require the employment of a mix of policy instruments whose costs and benefits will have to be evaluated from an economic welfare perspective.

In other words, a cost-benefit analysis that focuses on economic efficiency aspects of combating climate change could be enhanced with a framework that identifies the constraints under which society can pursue development. Indeed according to Drupp (2018), setting limits in physical terms to economic growth and using utilitarian discounting approaches to optimise development within these thresholds might represent an enhancement to current cost-benefit analyses.

2.1 The Interrelationships between Climate Change and Biodiversity Loss

The evaluation of costs and benefits of climate change and biodiversity loss in traditional welfare economics tools cannot forgo a comprehension of the extent to which these systems interact from a physical and ecological perspective.

The rapid disruption of biodiversity and changes in climate are closely interlinked: in fact, they share underlying direct and indirect drivers, interact, and have intricated and cascading effects that impact people's quality of life and affect societal goals (Díaz et al. 2019).

An increase in temperature impacts species' by altering their geographical ranges, microhabitat use, distributional patterns and phenology (Parmesan 2006; Parmesan and Yohe 2003), impacting nutrient cycles, causing ocean acidification, and catalysing the invasion from alien species (Tol 2009).

The impacts of climate change on species are heterogeneous and vary across geography and taxonomies (Bowler et al. 2020; Blowes et al. 2019). In this framework, ecosystems at higher thermal limits, such as coral reefs, wetlands, and lakes are at particular risk (Pörtner et al. 2021), in fact, according to Bowler et al. (2020), global warming can be considered the primary driver of biodiversity loss in areas that humans have not settled.

In addition, even if climate change impacts are focused on taxonomies at narrow local scales, they can trigger a homogenisation of biological communities at greater scales, compromising the resilience of entire ecosystems and their functioning (Guerra et al. 2021;

Dornelas et al. 2014).

In contrast to the loss of biodiversity triggered by non-climatic human pressures (such as land-use change), the effects of global warming cause more long-run impacts (Bertrand et al. 2011). According to Sala (2000), climate change-related biodiversity loss is second only to land-use change and, it is projected that global warming will overtake land-use change as the leading cause of biodiversity loss for the vertebrate community by 2070 (Newbold 2018). Urban (2015) conducts a meta-analysis of 131 published studies to estimate how species' extinction risks depend on different factors such as temperature increase, geography, taxonomic groups, modelling techniques and species' dispersal assumptions. It predicts that 7.9% of species are going to become extinct due to climate change. A key finding of the study is that, as temperatures rise, extinction risk do not only increase, but accelerate.

Moreover, climate change and biodiversity loss are interrelated via feedback and mechanistic links: global warming intensifies the risks of biodiversity loss and, conversely, species and natural ecosystems are fundamental for stabilising greenhouse gases and supporting climate change adaptation (Pörtner et al. 2021).

In light of these interrelationships, climate change and biodiversity loss should be addressed with a synergistic approach (Pörtner et al. 2021). Indeed, climate mitigation has its benefits for species conservation, and, conversely, slowing down biodiversity loss can positively impact climate change. Neglecting the interrelationships between climate change and biodiversity loss and their causes may lead to suboptimal answers in confronting either problem.

3 The Value of Biodiversity

3.1 Biodiversity and Human Well-being in the Cascade Model Framework

Biodiversity contributes to human well-being in many ways: in the terminology of the *Dasgupta Review*, it represents an “enabling asset” of ecosystems, and its value is “embedded in the accounting prices of natural capital” (Dasgupta 2021, 43).

Indeed, according to Hoppe et al. (2018), biodiversity loss represents a threat to the capability of ecosystems to provide the goods and services that benefit society.

Costanza et al. (2017, 3) define ecosystem services as “the ecological characteristics, functions, or processes that directly or indirectly contribute to human well-being”. Therefore, in the framework of cost-benefit analyses, the notion of ecosystem services represents a valuable conceptual bridge to link biodiversity and human well-being.

In this framework, the Millennium Ecosystem Assessment (2005) can be considered as one of the most comprehensive global assessments of the interrelationships between biodiversity, ecosystem change and human well-being (Haines-Young and Potschin 2010). It finds three categories of ecosystem services: provisioning, cultural and regulating. Provisioning services include the supply of water, food, organic and genetic resources, whilst cultural services are described as the ones that provide aesthetic satisfaction and scientific information; lastly, regulation services include pest and pathogenic control, environmental hazards, and stabilisation of agricultural production.

The Intergovernmental Panel on Climate Change’s fourth assessment (IPCC 2007) depicts a comprehensive summary of the most relevant gaps in scientists’ knowledge to forecast the impacts of climate change on biodiversity and ecosystems services. One of the most critical ones, according to the report, is to establish the links between biodiversity and ecosystem functioning. Indeed this relationship has been the object of study by ecologists in the last three decades (Haines-Young and Potschin 2010). There is generally vast support in the literature to the theory that biodiversity enhances the natural processes that benefit society, even though their interactions are complex (Potschin and Haines-Young 2011).

Moreover, a central caveat in cost-benefit analyses evaluating biodiversity-related ecosystem services is that these amenities are not traded in the market and have no market price. In this regard, non-market valuation techniques are often used to infer their value, which, despite the criticism they have received, represent the only viable option at the present time (Bosello and De Cian 2014).

An analytical framework to analyse how changes in biodiversity affect the economic valuation of ecosystem services and their contributions to human well-being is represented by the cascade model, conceived by Haines-Young and Potschin (2010) and developed by the Economics of Ecosystems and Biodiversity (TEEB 2010). The idea of the model is to determine a ‘production chain’ linking biodiversity to the value it generates for human well-being (Haines-Young and Potschin 2010). Whilst this approach might oversimplify non-linear and complex natural phenomena (Costanza et al. 2017), it could be helpful to understand the contribution of every level of the cascade (Paul et al. 2020) and support integrated analyses linking biodiversity and human well-being.

Figure 2: Ecosystem Services Cascade
Adapted from Dasgupta (2021) and Haines-Young and Potschin (2010)

In this model representation, biodiversity directly affects ecosystem functioning and is a “driver of ecosystem properties” (Paul et al. 2020, 4). In the definition of de Groot, Wilson, and Boumans (2002), if there is a demand for amenities, ecosystem functions become ecosystem services that provide benefits to society.

3.2 Biodiversity and Ecosystem Productivity

The relationship between biodiversity and ecosystem functioning is a key step in the cascade model and has been object of study from scientists of different disciplines for decades (Paul et al. 2020).

Species and their ecosystems are interconnected in a web of non-linear interrelationships, with thresholds, limits, and discontinuities (Holling et al. 1995). Among others, the importance of biodiversity lies in its role in safeguarding ecosystem resilience by providing fundamental key functions under different environmental conditions (Perrings et al. 1995). According to Elmqvist et al. (2003), the heterogeneity of response to environmental change of species, known as response diversity, is fundamental for the stability and resilience of ecosystems. Analogously as industrial production requires the input of different goods for production, ecosystem functions need a multitude of species to create the ecosystem services that benefit societies. If some of the input components are missing, then the level of output is compromised. According to Dasgupta (2021), a biodiversity-rich ecosystem can be compared to a highly diversified portfolio in the finance sector: similarly as a diversification reduces the risks of financial losses, a high level of biodiversity ensures the stability of environments.

In general, the assumption that higher levels of biodiversity enhance ecosystem functioning and, consequently economic value prevails in the literature (Paul et al. 2020; Haines-Young and Potschin 2010). Positive support to this hypothesis has also been given by Naeem et al. (1995), Tilman (1995); Tilman, Wedin, and Knops (1996); Tilman, Lehman, and Thomson (1997); Lawton et al. (1998); Cardinale et al. (2007).

O'Connor et al. (2017) conduct a study to establish a general relationship between biodiversity and biomass production. According to the paper, when biomass is a good approximation for ecosystem services, an exponential function could be used to model ecosystem functioning dependent on biodiversity levels. Haines-Young and Potschin (2010) argue that the link between biodiversity and ecosystem functioning might take three different shapes. Figure 3 depicts the potential functional forms of ecosystem functioning in relationship to biodiversity, according to Haines-Young and Potschin (2010).

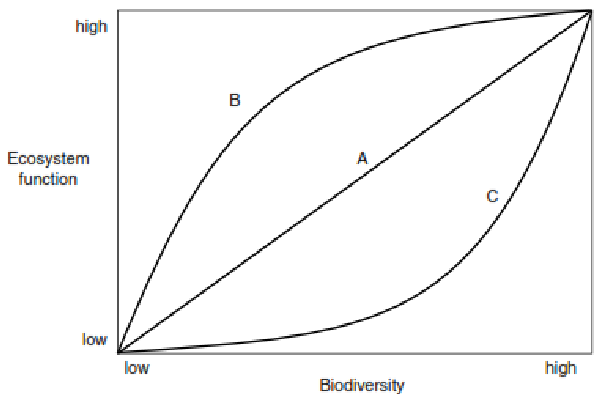

Figure 3: Hypothetical relationship between biodiversity and ecosystem functioning
Source: Haines-Young and Potschin (2010)

A type-A curve depicts a linear relationship, and in this case ecosystem functioning is highly sensitive to change in biodiversity, whilst, in type B, the relationship is curvilinear and saturates at higher levels of biodiversity, such that the change in ecosystem services are much more sensitive at lower levels of biodiversity. Additionally, it argues that the existence of a type-C curve could be supported to describe the abrupt change of ecosystem functioning due to the loss of keystone species.

3.3 The Economic Valuation of Biodiversity-related Ecosystem Services

The cascade model studies how variation in biodiversity levels affects people’s utility, a concept used by economists to measure the level of happiness and, therefore, the implicit subjective value of goods or services (Paul et al. 2020). According to the cascade model, biodiversity affects human well-being in both a direct and indirect way.

Biodiversity directly affects people’s utility via its bequest, existence, and altruistic values, often classified as non-use values (Millennium Ecosystem Assessment 2005). On the other hand, the indirect way biodiversity benefits human well-being is via the mediation of ecosystem services (Paul et al. 2020). In this thesis we focus mainly on the indirect relationship and emphasize the supporting function of biodiversity for the productivity of ecosystem services.

Given the multidimensionality of the value of biodiversity, relating the economic evaluation of existing species to the marginal productivity they deliver to ecosystems might represent a perspective that underestimates their non-use values (Dasgupta 2021)[1]. However, according to De Groot et al. (2012), in the context of cost-benefit analyses, estimating monetary values of biodiversity and ecosystem services can guide in the analysis of consumer preferences and improve decision-making. Nevertheless, the economic valuation of natural capital, like biodiversity and ecosystem services, represents an important challenge for environmental economists (Nijkamp et al. 2008).

Indeed the gains deriving from ecosystems to human well-being are scarcely represented by classic economic indicators. In fact, the loss in ecosystem services is never reflected in indicators such as Gross Domestic Product (GDP), which also fails to capture asset depreciation (Dasgupta 2021).

In theory, the economic value for a good or service is represented by the area below a demand curve, but this information for ecosystem services is not always available, as markets for biodiversity-related goods are non-existent (Paul et al. 2020).

To infer people’s Willingness to Pay (WTP) for biodiversity and ecosystem services, economists use non-market valuation techniques, which are divided into two great families: revealed preferences and stated preferences. The former type of technique aims to obtain people's WTP for ecosystem services by observing their decisions. The most commonly used methods of this family are hedonic pricing and travel cost method. On the other hand,

1 According to Dasgupta (2021), biodiversity is valuable for six different reasons: the interrelatedness of human life and nature, the dependence of human health, its recreative value, its contributions to ecosystem services, its existence value, and its sacredness value.

revealed preferences are survey-based methods and include contingent and group valuation (Perman 2011).

A few studies have tackled the challenge of estimating the economic value of ecosystem services. Costanza et al. (2014) estimated the global yearly value of ecosystem services between 125-145 $ trillions/yr., or 1.5-time of global GDP, and the value loss of ecosystem services in the range of 4.3–20.2 $ trillion/yr (3.4%-14%). The study conducted a meta-analysis on the WTP to ecosystem services in specific sites. To estimate the global value of ecosystem services it used a simple benefit-transfer method to estimate a global value. Several other studies have attempted attributing an economic value to ecosystem services: according to ten Brink (2011), the contribution of ecosystems services related to forestry, agriculture and fisheries constitute 15-20% of adjusted national GDP for countries such as Brazil, India, Indonesia. Baumgärtner et al. (2015) estimate that the world's ecosystem services loss 0.52% of their economic value per year between 1950-2010. According to Dasgupta (2010), the depreciation of some ecosystem services accounted for about 10% of GDP. Given these estimates, Drupp (2018) places the relative weight of ecosystem services consumption between 3-20% of the world's GDP, where the lower bound represents the contribution of food production to the world GDP.

Nevertheless, despite the increase in studies trying to attribute a monetary value to ecosystem services, the *Dasgupta Review* argues that justifying the preservation of the functioning of the biosphere because of its economic significance might be faulty. In fact, if the biosphere were annihilated, life would not exist, and therefore, there would be no one to enjoy the monetary compensation. From here stems the importance to discuss limited substitution possibilities between natural and manufactured capital, which we address in the following section.

4 Substitutability in Climate Change Economics

4.1 Limited Substitutability and the Role of Relative Prices

Limited substitution possibilities between scarce resource are a central topic in environmental economics (Drupp 2018). Nevertheless, SCC are often estimated in climate change models that assume economies will grow without ever impairing the ability of the biosphere to provide fundamental ecosystem services (Dasgupta 2021). In fact, most of the climate change IAMs assume high substitution possibilities between manufactured goods and natural goods (Neumayer 1999; Gerlagh and van der Zwaan 2002; Sterner and Persson 2008), which implies that climate change damages, regardless of the kind, can be compensated with an increase in consumption of manufactured amenities (Sterner and Perrson 2008).

This stance contrasts with the view of many ecologists and natural scientists who support a limited substitutability theory and mistrust the capabilities of technology in replacing natural capital (Gerlagh and van der Zwaan 2002). This strain of thought posits that conservation of natural resources, given their complementarity to manufactured amenities, is fundamental to preserve human well-being. The influential *Dasgupta Review* on the Economics of Biodiversity (Dasgupta 2021) emphasised the embeddedness of the economy in the environment and the low substitution possibilities between human-made goods and natural goods.

In the context of climate change cost-benefit analysis, it has been shown that the extent of substitutability between market and non-market goods plays a fundamental role in determining climate policy optimality (Drupp and Hänsel 2021). In the long run, limited substitutability between goods with heterogeneous growth rates, such as for market and non-market goods leads to an increase of relative prices of the amenities with lower growth rates, and thus to an augmented evaluations of climate change damages (Sterner and Persson 2008).

In the two-goods extension of the Ramsey neoclassical growth model by Drupp and Hänsel (2021) the change in relative price of the comprehensive non-market good over time is defined as relative price effect (RPE). The implicit price of the amenities is given by their marginal rate of substitution, which indicates how much the consumption of the goods would need to increase (decrease) to compensate a unit decrease (increase) of market goods to maintain utility constant (Drupp and Hänsel 2021). In this framework, the RPE measures the difference of the good-specific discount rates (Drupp and Hänsel 2021). Drupp and Hänsel

(2021) find that the RPE of non-market goods in their central calibration amounts to 4.1 percent in 2020 and to 1.9 percent in 2100.

In the study of relative price changes we conduct in this thesis, we build on the dual discounting approach and extend this framework to investigate whether, in addition to the relative price change between market and non-market goods, accounting for the relative price increases between two explicitly represented non-market goods, namely biodiversity-related ecosystem services and other non-market goods leads to more stringent climate policy.

In this framework, as the substitutability assumption is an important driver of relative price changes (Drupp and Hänsel 2021), investigating the substitution possibilities between the explicitly represented non-market goods, is of central importance.

In this context, the IPBES-IPCC co-sponsored Workshop on Biodiversity and Climate Change (Pörtner et al. 2021) highlighted the need to explore the interdependencies between climate change, biodiversity loss and other important contributors to human well-being such as health. In this context, the recent COVID-19 pandemic has shed light to the interrelatedness between global warming, biodiversity loss, and human health (Pörtner et al. 2021). In fact, the causes of the spread of the zoonotic disease have to be attributed to the invasion and destruction of ecosystems where climate change can be considered one of the main drivers (Lorentzen et al. 2020).

4.2 Substitutability between Biodiversity-related Ecosystem Services and Other Non-Market Goods

To investigate potential interrelationships between biodiversity-related ecosystem services and other non-market goods, as a first step, we need to clarify what non-market goods include in the model specifications we present in this thesis. Drupp and Hänsel (2021, 170) consider ‘non-market goods at a highly aggregated level, encompassing goods related to human health as well as environmental goods, ranging from clean water to aesthetic beauty’. In the study conducted in this thesis we disentangle the comprehensive non-market good variable into the consumption of biodiversity-related ecosystem services, representing all ‘environmental goods’ in the specification of Drupp and Hänsel (2021), and attribute to ‘other non-market goods’ all remaining amenities that are not priced in the market, such as health, culture, leisure, socially contingent consequences like migration and risk for conflicts.

Whilst there is a vast literature investigating the substitution possibilities between ecosystem services and manufactured goods in a dual good CES utility function framework (Drupp 2018), limited substitutability between biodiversity-related ecosystem services and other non-market goods in a nested CES framework represents a novel research frontier and methods for the empirical estimation of the substitutability parameter for this particular function specification are not available. Moreover, as both of these goods do have not a market price, estimating their elasticity of substitution represents an even more challenging task.

Due to the limited scope of this thesis, this section will not aim to develop an empirical methodology to determine plausible substitutability parameter values between ecosystem services and other non-market goods. However, it will conduct a literature review to present arguments for both a high and low substitutability assumption between these amenities. Arguing about potential substitutability relationships between biodiversity-related ecosystem services and other non-market amenities cannot forgo the investigation of their interdependencies. Despite the difficulty in obtaining empirical estimates of the substitutability parameter, the literature presents many lines of argument that could be used to support a relatively low substitutability assumption between biodiversity-ecosystem services and other fundamental non-market goods.

In this regard, a non-market good that appears to be interrelated with biodiversity-ecosystem services is health. In fact, biodiversity provides fundamental ecosystem services to societies by providing unique genetic material and supplying medicines: circa 4 billion people rely on natural drugs, and it is estimated that about 70% of cancer medicines have a natural origin or are synthetic but inspired by nature solutions (IPBES 2019). Additionally, limited substitutability between health and biodiversity-related ecosystem services could be argued in light of the COVID-19 pandemic, to which ca. 2.6 million deaths are associated (World Health Organisation 2021). The disease outbreak is thought to be related to the destruction of natural habitats, to which zoonoses are linked (Groom and Turk 2021; Pörtner et al. 2021). In this context, IPBES (2019) estimated zoonotic diseases (before the COVID-19 pandemic) to account for 17% of all infectious diseases and responsible for 700000 deaths per year; according to IPBES (2019), declines in biodiversity can negatively affect health by exacerbating inequalities in societies and hindering access to healthy diets.

More arguments favouring complementarity between ecosystem services and other non-market amenities could be presented in light of the cultural and spiritual values that biodiversity and ecosystems provide to society. Indeed religions, knowledge systems and social structures have always depended on ecosystems and their functioning, where biodiversity is a fundamental component (Millennium Ecosystem Assessment 2005). In this regard, cultural ecosystem services have benefitted people by providing, among others, artistic and spiritual fulfilment, recreation and aesthetic beauty (IPBES 2019).

On the other hand, arguing for a high degree of substitutability between biodiversity and related ecosystem services and other non-market goods would imply that losses in services from nature could be replaced with increased consumption levels of other non-market amenities like health or culture to maintain utility levels constant. In the case of the provision of medicines from nature, it could be argued that technological progress could increase the capability to replace natural drugs with synthetic ones or that future outbreaks of zoonotic diseases, originated by the invasion and destruction of ecosystems, could be stopped with the progress in medicine. Additionally, high substitution possibilities between non-market goods could be supported due to the fact that ecosystems, instead of providing benefits to human well-being, provide ‘disservices’ having deleterious effects on human health (Daw et al. 2016). Examples in this category might include health problems related to pollinating plants' allergies or vector-borne diseases (Gómez-Baggethun and Barton 2013). For cultural amenities, however, it is more difficult to argue for high substitution possibilities. In fact, according to Hirons et al. (2016), due to their subjective nature, cultural services from ecosystems cannot be generalised and once degraded, they cannot be replaced by any means.

More in general, as Dasgupta and Heal (1974) point out, even if a substitutability argument can be argued for a resource with large availabilities, with scarcity increases, its elasticity relationship with other goods will tend toward complementarity. In this sense, as the increase in scarcity of biodiversity-related ecosystem services accelerates, hypothetic high substitutability with other non-market amenities seems rather implausible. Moreover, according to Sterner and Persson (2008), if there is an aggregate variable representing a multitude of goods (in our example could be represented by a variety of more or less important biodiversity-related ecosystem services) with different elasticities of substitution, the overall substitutability of the variable will be dominated by the goods with lower elasticity, because of their increasing weight in the utility function.

5 Integrated Assessment Models of Climate Change

Contrasting climate change is a particularly challenging issue because it affects a wide arrange of disciplines and aspects of society; therefore, the development of effective policies requires integrating knowledge from different domains into single analytical frameworks (Nordhaus 2013).

IAMs, via the representation of economic, physical, and social dynamics, consent the analysis of the costs and benefits of climate change and the evaluation of targeted policy interventions (Weyant et al. 1996) and can be thus considered as “the basis for evidence-based policy and decision support” (Pecl et al. 2017, 28). Climate change IAMs are based on welfare economics models and consent to evaluate the optimal use of scarce resources, a topic at the heart of economics, and are therefore useful to evaluate trade-offs between different stabilization goals.

IAMs can be classified into two great families: policy optimisation and policy evaluation models (Weyant et al. 1996). The models of the former type have an objective function, which usually takes the form of an economic welfare function, to be maximised; the welfare function is obtained by aggregating a set of utility functions in the case of general equilibrium models or consumer and producer surplus in the case of partial equilibrium models. In general, these models are utilised to evaluate alternative paths and policies (Nordhaus 2013).

On the other hand, policy evaluation models are built on recursive or equilibrium models and generate pathways of important variables without conducting an optimisation (Nordhaus 2013). Typically, policy evaluation models are used to compare different climate policies (Nordhaus and Sztorc 2013).

One of the main characteristics of IAMs is that multiple research areas are consolidated in a unique framework, and rather than taking exogenous variables from other models or assumptions, they are able to work in an integrated way (Nordhaus and Sztorc 2013). Moreover, a convenient feature of IAMs is that they can be extended with alternative constructs; in fact, while making sure that all the other components maintain their validity with the variation in the assumption, these amendments can be handily incorporated into the model (Nordhaus 2019).

5.1 The Dynamic Integrated Climate-Economy model (DICE)

The Dynamic Integrated model of Climate and the Economy (DICE) is one of the most used IAMs by policymakers and is built on a general equilibrium policy optimization model that embraces the fields of economics, policy, and natural sciences of climate change (Nordhaus 2017).

DICE performs an intertemporal optimization to maximise the social welfare function representing the "discounted sum of the population-weighted utility of per capita consumption" (Nordhaus 2017, 1) in a standard Ramsey growth model.

The model differs from pre-existing classical macroeconomic ones because of the long timeframe setup; undeniably climate-change is a phenomenon that needs to be analysed in the long run (Nordhaus and Sztorc 2013).

Nordhaus extended the Ramsey model with three important features. Firstly, he modelled how greenhouse gases depend on the level of economic output via emissions. Secondly, he augmented the model by including the relationship between temperature and atmospheric carbon dioxide concentration via an increase in radiative forcing and lastly, he included the interrelationship between economic sector and climate change. In DICE, the economies choose the level of technology, emissions and capital and make consumption choices.

The social welfare function W is formally given by the following:

$$W = \sum_{t=1}^{t\,\max} U_t\left(C_t, L_t\right) \cdot R_t \tag{5.1.1}$$

Notation wise, W stands for welfare, U for Utility, C_t for consumption, L_t for population level and R_t for the discount factor. Population size L_t is exogenously given and the discount factor R_t determines the welfare weights on utility of future generations. Utility takes following functional form:

$$U_t\left(C_t\right) = L_t \cdot \left(\frac{C_t}{1-\eta}\right)^{1-\eta} \tag{5.1.2}$$

Therefore, DICE aims to optimise the stream of 'generalised consumption' C_t, which entails, in addition to market goods, non-market goods like ecosystem services and health (Nordhaus and Sztorc 2013).

In DICE there are two normative factors that influence the valuation of consumption in the intertemporal setting: the pure rate of social preference δ, and the intertemporal elasticity of the marginal utility of consumption η, or consumption elasticity. The intertemporal elasticity of the marginal utility of consumption determines the curvature of the utility function. The higher the value, the lower the marginal contributions of consumption to utility (Nordhaus 2013). On the other hand, the pure rate of social time preference δ determines the welfare weights on the utility of future generations (Nordhaus and Sztorc 2013).

The discount factor takes following explicit form:

$$R_t = (1+\delta)^{-t} \tag{5.1.3}$$

5.1.1 The Damage Function in DICE

"Projecting impacts is the most difficult task and has the greatest uncertainties of all the processes associated with global warming" (Nordhaus 2019, 1998). Damage functions in IAMs are generally represented as economic estimates of the impacts of temperature increases in terms of GDP (Stanton, Ackerman, and Kartha 2009). These estimates are important to balance between costly emissions reductions and climate damages. They are usually oriented toward the impact estimation for a doubling in pre-industrial carbon dioxide carbon concentration of GHG (Nordhaus and Sztorc 2013). In general, surveys estimate market and non-market impacts ranging from 0 to 10.2% of GDP for a temperature increase of 3°C (Howard and Sterner 2017).

In DICE, climate change damages are specified as a global aggregated function. They include the concepts of damages to market and non-market amenities, and they also feature an adjustment for catastrophic events (Nordhaus and Sztorc 2013). To account for omitted non-market damages such as ocean acidification and biodiversity loss, uncertainties and other difficult quantifiable impacts, Nordhaus (2017) increases by 25% the total damages' estimates via a judgemental adjustment (Nordhaus and Sztorc 2013).

Formally, economic damages from temperature increases are given by a quadratic function:

$$\Omega_t = \omega \cdot T_t^{\,2} \tag{5.1.1.1}$$

Ω_t represents the fraction of output loss at time t for a squared increase of global atmospheric temperature T_t, and ω represents the damage scaling parameter. The function is calibrated according to the monetised damages from the Tol (2009) article: for a temperature increase of 3°C, total damages amount to 2.12 % of GDP and for an increase to 6°C they amount to circa 8% (Nordhaus 2019).
In the model, the damage fraction Ω_t affects total output Y_t multiplicatively:

$$Y_t = (1-\Omega_t)(1-\Lambda_t)\mathrm{A}_t K_t^{\,\gamma} L_t^{\,1-\gamma} \tag{5.1.1.2}$$

Λ_t denotes spending in abatement, A_t total factor productivity, K_t and L_t capital and labour inputs, respectively. Non-market damages, being simply summed up to market damages in Ω_t, are perfectly substitutable with market damages (Drupp and Hänsel 2021) and therefore affect the level of total output. Moreover, despite the perfect substitutability between market and non-market damages, on the consumption side, market and non-market amenities feature an implicit Cobb-Douglas type of substitutability (Drupp and Hänsel 2021). This has to be attributed to the damage term $(1-\Omega_t)$ which affects output levels multiplicatively (Weitzman 2010) and to the relationship between the substitutability in production and consumption side (Drupp and Hänsel 2021).

5.2 The Estimation of Non-Market Damages

Estimating climate change damages to the non-market sector is of central importance in IAMs and has been a key inquiry since the early days of climate policy, representing one of the most challenging issues in climate-change economics (Drupp and Hänsel 2021).
Tol (2009) makes a taxonomy of the non-market impacts that are underestimated in climate change models; according to the paper, underrated non-market damages include ocean acidification, biodiversity loss, and health. According to Gerlagh and van der Zwaan (2002), the reasons of the underestimation can be attributed to the difficulty to estimate their non-

use and indirect use-values. Therefore, including non-market damages in climate change cost-benefit analyses results in a difficult task.

Before scaling up the total damage estimates to correct for uncertainties in damages to the non-market sector, former DICE versions relied on sectoral estimates (Nordhaus and Sztorc 2013). Nordhaus and Boyer (2000) determined the economic value of climate impacts on ‘ecosystems and human settlements’ by making rough estimates on the extent to which these systems are affected by an increase in temperature. They estimated total damages from a 2.5°C global temperature increase to be 1.74% of GDP and impacts to human ‘settlements and natural ecosystems’ to be 0.1% of GDP or 5.7% of total damages.

To obtain the climate change damage to ecosystem services estimate, the authors make a series of different subjective judgements. They estimate natural ecosystems to account between 10% and 25 % of national output and people’s WTP to prevent climate change damages for a temperature increase of 2.5 °C to be about 1% of the global output value. The damage estimate is then obtained by multiplying the value share of ecosystems in total output and people’s WTP.

Furthermore, according to Nordhaus and Moffat (2017, 19) “the estimates of the losses from ecosystems […] are omitted or unreliable […] and are likely to be underestimates of true damages”. In fact, according to Weitzman (2007), considering low probability catastrophic risks, the impacts of climate change on the ability to enjoy fundamental ecosystem services are more worrisome than the damage to conventional consumption goods. Indeed recent literature has estimated higher damage estimates to non-market amenities.

For example, for a temperature increase of 2.5°C, the *Stern Review* (Stern 2007) attributes 50% of total damages to the non-market sector, compared to the 25% assumed by Nordhaus. These figures are confirmed in Howard and Sylvan (2015) who survey expert economists to establish a consensus about climate change damages and risks. According to the report, the level of climate change damages that accrue to the non-market sector are 50.24% of total economic damages for a temperature increase of 3°C. Following these estimates, Drupp and Hänsel (2021) double market damages (1.63%) to include for the damage projections from the expert elicitation from Howard and Sylvan (2015), increasing therefore the estimate substantially. Yet Howard and Sylvan (2015) does not report the estimated damages for each sector, i.e. to ecosystem services or cultural amenities.

6 Integrating Climate Change-driven Biodiversity Loss in DICE

This section introduces the model amendments on DICE2016 (Nordhaus 2017) to account for the consumption of biodiversity-related ecosystem services and other non-market goods explicit based on the findings of the previous chapters.

6.1 Disentangling the Social Welfare Function

As a first step, following Drupp and Hänsel (2021), we disentangle the comprehensive consumption variable of Equation 4.2.3 via a CES utility function into the explicit consumption of market and non-market goods. Utility of the representative agent is given by the consumption of market goods C_t and from the consumption of non-market goods EQ_t in the following function:

$$U(C_t, EQ_t) = \frac{1}{1-\eta}\left(\alpha \cdot (EQ_t - \overline{EQ})^{\theta} + (1-\alpha) \cdot C_t^{\theta}\right)^{\frac{1-\eta}{\theta}} \tag{6.1.1}$$

$\overline{EQ}$ represents the minimum consumption of non-market goods to satisfy subsistence requirements (Drupp 2018). For $EQ_t < \overline{EQ}$ the agent's utility is a function of only non-market goods, reflecting the need to meet survival needs before consuming market goods. Further, θ represents the substitutability parameter between market and non-market goods and α is the share parameter for the weight of non-market goods in the representative agent's utility function. The substitutability parameter θ is related to the elasticity of substitution σ with following function:

$$\sigma = \frac{1}{1-\theta} \tag{6.1.2}$$

As a second step, to account for the explicit consumption of biodiversity-related ecosystem services and other non-market goods, we disentangle the comprehensive non-market good consumption term EQ_t via a nested CES utility function. Utility from the consumption of non-market amenities is given by the explicit consumption of biodiversity-related ecosystem services ES_t and other non-market goods O_t:

$$U(O_t, ES_t) = \left((1-\phi)\cdot O_t{}^{\pi} + \phi\cdot ES_t{}^{\pi}\right)^{\frac{1}{\pi}} \quad (6.1.3)$$

ϕ represents the share parameter of biodiversity-related ecosystem services in consumption of non-market goods, and π is the substitutability parameter between the non-market goods ES_t and O_t. After merging Equations 6.1.1 and 6.1.2, utility of the representative agent results in the following:

$$U(C_t, O_t, ES_t) = \frac{1}{1-\eta}\left(\alpha\cdot\left(\phi\cdot ES_t{}^{\pi} + \left((1-\phi)\cdot O_t{}^{\pi}\right)^{\frac{1}{\pi}} - \overline{EQ}\right)^{\theta} + (1-\alpha)\cdot C_t{}^{\theta}\right)^{\frac{1-\eta}{\theta}} \quad (6.1.4)$$

To ensure comparability with Drupp and Hänsel (2021), we set the initial value of $O_0 = ES_0 = C_0$. In this way, the parameter ϕ determines the weight of consumption of biodiversity-related ecosystem services for the representative household in the initial period. Both non-market goods degenerate with increases in global atmospheric average temperature increase T_t: the economic evaluation of other non-market goods shrinks with increasing temperatures and a damage scaling parameter ψ in line with Drupp and Hänsel (2021) and Sterner and Persson (2008):

$$O_t = \frac{O_0}{1+\psi T_t{}^2} \quad (6.1.5)$$

On the other hand, the economic evaluation of biodiversity-related ecosystem services deteriorates with increases in temperature-related biodiversity loss, as we will introduce in the next section.

6.2 Modelling the Interrelationships between Climate Change-driven Biodiversity Loss and Human Well-Being

In following section we propose a functional form to describe how the consumption of biodiversity-related ecosystem services degenerates due to global atmospheric average temperature increases. We model the biodiversity-dependent economic evaluation of ecosystem services following the schemes of the cascade model, where, as described in Section 3.2, it is assumed that the functioning of ecosystems, and their capability of providing services contributing to the welfare of society, is dependent on underlying biodiversity levels.

6.2.1 Integrating a Climate-Change driven Biodiversity Loss Function in DICE

As a first step, we integrate in DICE the relationship between average atmospheric temperature increase and biodiversity loss, measured in total species number with respect to year 1900 using a polynomial function of the second degree. The function takes following form:

$$B_t\,(T_t) = \beta_0 + \beta_1 \cdot T_t + \beta_2 \cdot T_t^{\,2} \qquad (6.2.1.1)$$

B_t reflects the percentage change in biodiversity for a given global atmospheric average temperature increase T_t with respect to pre-industrial levels. We use the parameter set from Kaushal and Navrud (2018), which calibrate the biodiversity loss function according to the climate change extinction predictions from the Urban (2015) study. Table 1 reports species' extinction predictions from Urban (2015) under 4 different climate change scenarios.

Global average temperature increase (pre-industrial)	0.8 °C	2 °C	4 °C	4.3 °C
Predicted Species' Extinctions	2.80%	5.20%	8.50%	16%
Scenario	Current	Target	RCP 6.0	RCP 8.5

Table 1: Climate change-driven extinction predictions according to Urban (2015)

The parameters shaping the climate change-driven biodiversity loss function we propose in this thesis are represented in Table 2.

Parameter	Value	Source
β_0	$2.8 \cdot 10^{-4}$	Pimm et al. (2015)
β_1	$1.73 \cdot 10^{-4}$	Kaushal and Navrud (2018)
β_2	$4.4 \cdot 10^{-3}$	Kaushal and Navrud (2018)

Table 2: Overview of biodiversity loss function parameters

The parameter β_0 reflects the natural background extinction rate with no climate change from the Pimm et al. (1995) study, whilst the β_1 and β_2 are the coefficient that determine the curvature of the function. The following graph depicts the relationship between temperature increase and species extinction risks.

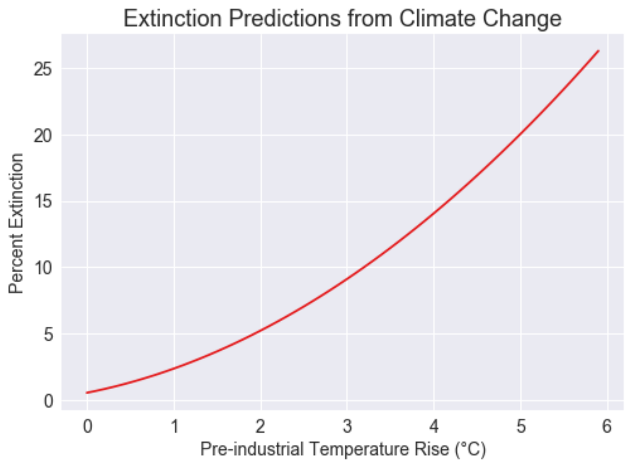

Figure 4: The relationship between atmospheric temperature increases and species' extinctions according to the Urban (2015) meta-analysis estimates.

The chart reflects the key finding from the Urban (2015): extinction risks from climate change accelerate with rises in global temperature.

6.2.2 Introducing the Biodiversity-related Ecosystem Services Damage Function according to the cascade model

Having introduced the temperature-related biodiversity loss function we now present how changes in biodiversity levels affect welfare levels in the schemes of the cascade model. To model how variations in biodiversity levels affects ecosystem functioning, we use an adapted functional from from O'Connor et al. (2017), and relate biodiversity changes to ecosystem functioning via a power function:

$$F(B_t) = 1 - B_t^{\gamma} \tag{6.2.2.1}$$

The more biodiversity loss B_t increases, the more ecosystem functioning is compromised. The exponent γ controls the curvature of ecosystem functioning in response to biodiversity loss. With $0 < B < 1$, for $\gamma < (>) 1$ there will be a more (less) than proportional reduction of ecosystem functioning due to biodiversity loss and for $\gamma = 1$ there will be a linear reduction.

For our analysis we will use a value of $\gamma = 1.58$, obtained by adapting the central value of the O'Connor et al. (2017) study to our functional form[2]. The choice of this value indicates a positive-concave relationship, thus a rather mild reduction in ecosystem functioning due to climate change-driven biodiversity loss, indicating a type-B relationship in the schemes from Haines-Young and Potschin (2010).

Additionally, for the analysis we make the simplifying assumption that species are a homogenous good contributing to the evaluation of ecosystem services equally, in line with previous work from Paul et al. (2020).

As a next step, we will assume that the degradation of ecosystem functioning due to increases in climate change-driven biodiversity loss will impact the economic evaluation of ecosystem services by including the initial valuation of ES_0 in Equation 6.2.2.1 multiplicatively. Finally, we add a damage scaling parameter ρ controlling how the economic evaluation of ecosystem services changes with increases in climate change-driven biodiversity loss.

The damage function of biodiversity-related ecosystem services results in the following

2 See Appendix for more details

equation:

$$ES_t = ES_0 \cdot \left(1 - \rho \cdot \left(\underbrace{\beta_0 + \beta_1 \cdot T_t + \beta_2 \cdot T_t^2}_{B_t}\right)^{\gamma}\right) \tag{6.2.2.2}$$

The evolution of biodiversity-related ecosystem services therefore depends on climate change-driven biodiversity loss B_t, the biodiversity-ecosystem functioning curvature coefficient γ, and the damage scaling parameter ρ.

7 The Integrated Assessment of Climate Change-driven Biodiversity Loss in DICE

7.1 The Impacts of Relative Price Changes on Climate Policy Optimality

In this section, we explore the implications of limited substitutability between biodiversity-related ecosystem services and other non-market goods for optimal climate policy.

We conduct the analysis by implementing the model changes in a python version of DICE2016 by Krichene (2019).

In the following, we describe in detail the calibration approach of the extended model we propose in this thesis. Given the lack of comprehensive estimates in the literature, we determine climate change damages to biodiversity-related ecosystem services and other non-maker goods by relying on a series of different assumptions and calculations, which we illustrate in detail in the following paragraphs.

As a first step, to determine the climate change damages estimate to biodiversity-related ecosystem services D^{ES}, following the methodology from Nordhaus and Boyer (2000), we assume that the climate change damage share accruing to ecosystem services amount to 5.7% of total damages and that this figure does not change for a temperature increase of 3°C. Therefore, we find that, for total damages amounting to 2.12% of GDP, climate change damages accruing to biodiversity-related ecosystem services D^{ES} are 0.12% of GDP. Next, following the approach from Drupp and Hänsel (2021), we disentangle the total damage estimate from the standard DICE model into market and non-market damages and, to align with the estimates from the Howard and Sylvan (2015) survey, we rescale the overall non-market damages estimate to double market damages. For a temperature increase of 3°C, market and non-market damages equally amount to 1.63% of global output totalling to 3.26% of GDP. Moreover, to account for the increased magnitude of non-market damages from Howard and Sylvan (2015), we rescale the biodiversity-related ecosystem services damage estimate D^{ES} with a simple proportion.

To achieve this task, we first calculate the share of overall non-market goods damages in total damages in the standard DICE. As described in Section 5.1, in more recent model versions, Nordhaus accounts for non-market damages by increasing the overall damage estimate by 25%. Hence, we calculate that, for total damages amounting to 2.12% of GDP, damages accruing to all non-market amenities are 0.53%/GDP.

As a result, given that damages to ecosystem services amount to 5.7% of total damages (0.12% of GDP) where total non-market damages equal 0.53% of GDP (25% of 2.12%), for non-market damages amounting to 1.63% of GDP, we calculate that damages to biodiversity-related ecosystem services D^{ES} add up to 0.36% of GDP or to 11% of total damages.

Moreover, the climate change damage estimate D^{ψ} to all other non-market amenities is given by the difference between overall non-market damages and damages to biodiversity-related ecosystem services. As a result, damages to other non-market goods amount to $D^{\psi} = 1.27\%$ of GDP, thus substantially higher than the damages accruing to biodiversity-related ecosystem services.

Finally, we calibrate the damage scaling parameters ρ and ψ according to the novel identified estimates by comparing two different model specifications. On the one hand, a model in which non-market damages, that include O_t and ES_t are perfectly substitutable with damages to market goods D^{Ω} and are included in consumption. On the other hand, a model in which climate change damages are attributed to market goods D^{κ}, other non-market goods D^{ψ} and ecosystem services D^{ES} specifically[3].

The natural starting point to explore how relative price changes affect climate policy optimality in the model featuring the consumption of biodiversity-related ecosystem services and other non-market goods explicitly, is to compare its output with the one of a model featuring only a comprehensive non-market goods consumption variable.

Therefore, for comparability, we draw the parameter set from Drupp and Hänsel (2021) central calibration, including the substitutability parameter between market and non-market goods θ , the share parameter of the weight of non-market goods in utility α and the subsistence requirement of non-market goods $\overline{EQ}$. Furthermore, we use the value of the rate of pure time preference δ and intertemporal elasticity of marginal utility of consumption η from the Drupp et al. (2018) experts' elicitation, also used in the Drupp and Hänsel (2021) study. For the share parameter of biodiversity-related ecosystem services in non-market goods consumption, we use a value of $\phi = 0.5$, thus in the lower range of the possible values indicated from the Drupp (2018) study reported in Section 3.1.2 of this thesis. With this parameter value, the share in consumption of other non-market goods will be equivalent to the one of biodiversity-related ecosystem services.

[3] See Appendix for more details.

For the analysis, we draw two substitutability parameters, encompassing a high and low substitutability scenario between the consumption of ecosystem services and other non-market goods. We use a parameter of $\pi = 1$ for the high (perfect) substitutability scenario, whilst for a low substitutability one, we choose a parameter value of $\pi = -2.5$ .Table 3 summarizes the parameter set for the central calibration.

Parameter	Description	Source	Value
α	Share of NMG in utility function	Drupp and Hänsel (2021)	0.1
$\overline{EQ}/EQ$	Subsistence consumption of non-market goods	“	10%
θ	Substitutability MG-NMG	“	-0.11
δ	Rate of pure time preference	Drupp et al. (2018)	1.10%
η	Intertemporal elasticity of marginal utility of consumption	Drupp et al. (2018)	1.35
γ	Biodiversity-ecosystem functioning exponent	O’Connor et al. (2017)	1.58
π	Substitutability parameter of biodiversity-related ecosystem services and other non-market goods	Own assumption	-2.5, 1
ϕ	Share of biodiversity-related ecosystem services in NMG	“	0.5
D^{ES}	Biodiversity-related ecosystem services damages estimate	“	0.36%
D^{ψ}	ONMG damages estimate	“	1.27%
D^{Ω}	Total damages estimate	Drupp and Hänsel (2021)	3.26%

Table 3: Parameter set for central calibration:
NMG: Non-Market Goods, MG: Market Goods, ONMG: Other Non-Market Goods

Additionally, to compare the extended model with the baseline model, which accounts for relative price changes only partially, we include in the analysis a DICE2016 (Nordhaus 2017) standard model calibration.
Further, to contrast the impacts of relative price changes on optimal climate policy with the well-understood influence of the discounting parameter δ, we include in the analysis a DICE2016 (Nordhaus 2017) model featuring a lower rate of pure time preference of δ=0.01% as suggested by Stern (2007), opposing the parameter value δ=1.5% used in the original model specification. We will henceforward name the models “Nordhaus” and “Stern” respectively.

The following table gives an overview on the rate of pure time preference and intertemporal elasticity of marginal utility of consumption used in the different model specifications.

Parameter	Description	Nordhaus	Stern	Drupp and Hänsel	Extended Model
δ	Rate of pure time preference	1.5%[1]	0.01%[2]	1.1%[3]	1.1%[3]
η	Intertemporal elasticity of marginal utility of consumption	1.45%[1]	1.45%[1]	1.35%[3]	1.35%[3]

Table 4: Overview of intergenerational and social discounting parameters for the different model specifications. Sources: 1 Nordhaus (2017), 2 Stern (2007), 3 Drupp et al. (2018).

Furthermore, to neutralize the effects of different damage estimates in the analysis, we scale up the overall damage estimate in the Nordhaus and Stern models[4].

[4] To calibrate the baseline model according to higher damage estimates, we find the new damage scaling parameter $\omega = 0.00362$ from Equation 5.1.1.1, such that for a temperature increase of 3°C, total damages Ω amount to 3.26% of GDP.

7.1.1 Exploring the Role of Substitutability for Optimal Climate Policy

Moving forward, we run the model and explore the impact of the substitutability parameter π on optimal climate policy.

Figure 5 depicts optimal industrial CO_2 emissions and optimal atmospheric temperature increases compared to pre-industrial levels in the previously introduced model specifications.

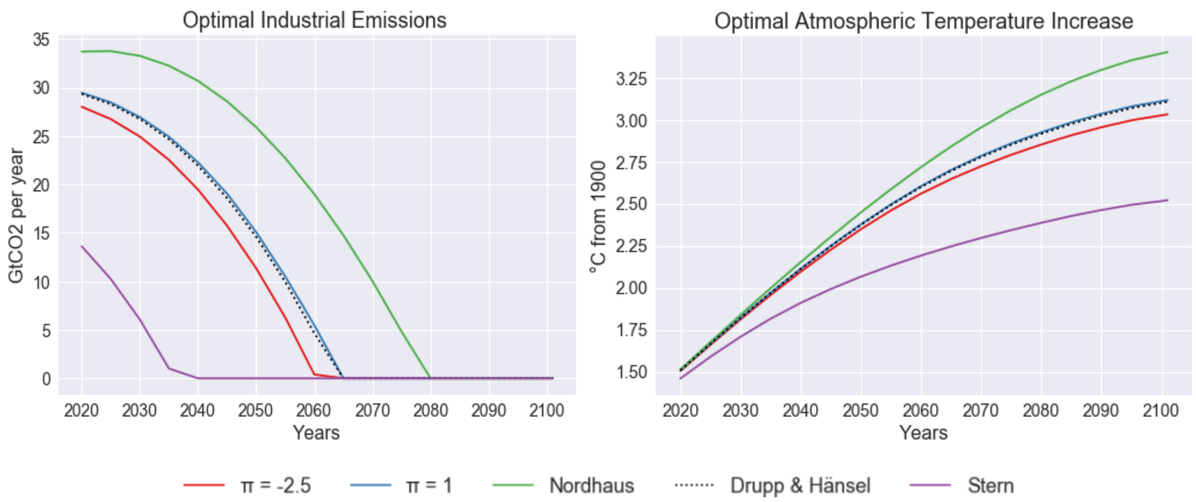

Figure 5: Optimal industrial emissions and global average atmospheric temperatures increase according to five different model specifications.

The dotted black lines in the Figures represents the model output from Drupp and Hänsel (2021), nearly overlapped by the trajectory from the extended model with a perfect substitutability parameter $\pi = 1$, marked with a solid blue line. The solid red lines represent optimal temperature increases and industrial emissions for a model featuring a low substitutability assumption between biodiversity-related ecosystem services and other non-market goods. Lastly, the higher and lower curves are represented by the Nordhaus and Stern models, respectively. The discrepancy between these two models has to be attributed entirely to the rate of pure time preference, the only parameter differing between these two model specifications.

Comparing the optimal emissions and temperatures trajectories from the Drupp and Hänsel (2021) and the extended model featuring a perfect substitutability assumption with the output from the Nordhaus model calibration, we can notice significant lower optimal emissions and temperature pathways. Alongside a higher rate of pure time preference and intertemporal elasticity of marginal utility of consumption, higher mitigation efforts have to be attributed to the explicit accounting of non-market goods' relative price changes, as analysed in depth in Drupp and Hänsel (2021). Indeed for the analysis we use the substitutability parameter

$\theta = -0.11$ from the Drupp and Hänsel (2021) central calibration, which features a slightly lower substitutability degree than the implicit Cobb-Douglas type substitutability featured in the standard Nordhaus model (Drupp and Hänsel 2021).

Focussing on the extended model trajectories, we can notice a decrease in the optimal level of emissions and temperatures depending on the substitutability assumption between the explicitly represented non-market goods. Assuming low substitutability among non-market goods translates into a full economy decarbonisation 5 years and in advance compared to a model with perfect substitutability (2060 vs. 2065) and lower peak temperatures (0.09 °C). Therefore, accounting for the heterogeneity in growth rates and assuming limited substitutability between biodiversity-related ecosystem services and other non-market goods leads to more stringent optimal climate policy outcomes, as compared to a model featuring perfect substitutability between the explicitly represented non-market goods or a model featuring a heterogenous non-market goods’ variable.

7.1.2 Estimating the Social Costs of Carbon

Before scrutinizing more in detail the mechanisms leading to a more stringent climate policy, we assess the impacts of the substitutability assumption on the SCC estimates. Figure 6 depicts the evolution of the SCC for an 80 years' time-horizon for the model specifications introduced earlier in this section[5].

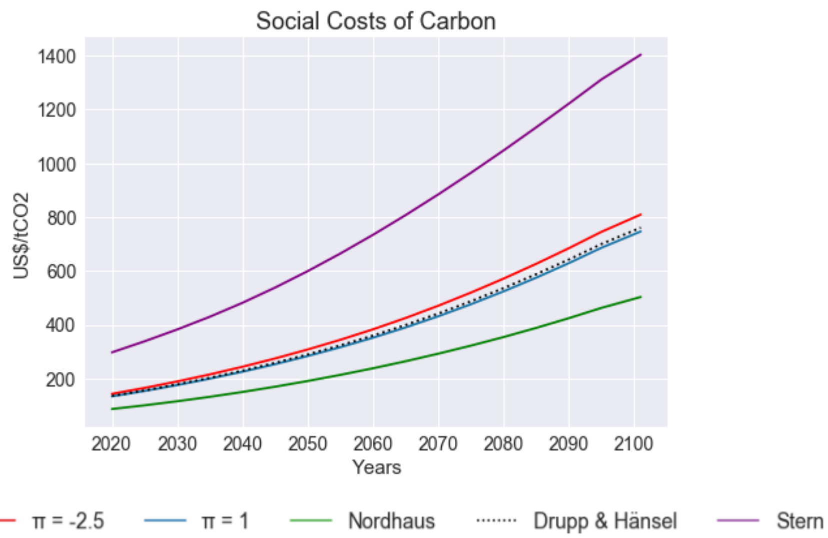

Figure 6: Evolution of the Social Costs of Carbon estimate in an 80 years' timeframe according to five different model specifications and calibrations .

For the year 2020 (2100), the SCC estimates for the model featuring the explicit consumption of biodiversity-related ecosystem services and other non-market goods are 134.67 (751.11) $/tCO_2$ and 147.28 (838.61) $/tCO_2$, with perfect substitutability and low substitutability, respectively, which correspond to a relative difference of 9.6 % (11.6%), thus diverging in time. Using the Drupp and Hänsel (2021) model output, the estimate amounts to 135.51$/tCO_2$ and 760.79 $/tCO_2$, 2020 and 2100, respectively, thus very close to the perfect substitutability model output.

Therefore, the figure suggests that, with a low substitutability between other non-market goods and biodiversity-related ecosystem services policymakers should impose higher carbon prices, compared to a case where substitution possibilities are perfect.

5 The SCC are calculated using the NEOS Knitro solver for the AMPL optimization language.

7.1.3 Assessing the Impact of Relative Price Changes on Optimal Climate Policy

In this section we estimate the relative price changes of the explicitly represented non-market goods.

In the extended model framework we have introduced in this thesis, the RPE of the explicitly represented non-market goods indicates how much the consumption of one good would need to increase (decrease) to compensate a marginal decrease (increase) of the other non-market good to maintain utility from the consumption of non-market goods constant.

Given that the climate change damage estimate of other non-market goods is higher compared to the one of biodiversity-related ecosystem services, their growth rates deteriorates at a faster rate. Therefore, we will expect a positive RPE of other non-market goods.

Formally, the RPE of other non-market goods O_t is given by the difference in growth rates of ES_t and O_t, and depends on the substitutability parameter π:

$$RPE_t = (1-\pi)\cdot\left(g_{ES_t} - g_{O_t}\right) \qquad (7.4.1)$$

With heterogeneous growth rates, where $g_{O_t} < g_{ES_t}$ and a perfect substitutability parameter $\pi = 1$, the RPE will be null, whilst for $\pi < 1$, we will observe a positive RPE_t of other non-market goods. This demonstrates the importance of the substitutability parameter in this framework. Figure 7 illustrates the evolution of the RPE between other non-market goods and biodiversity-related ecosystem services in a 80 years' period.

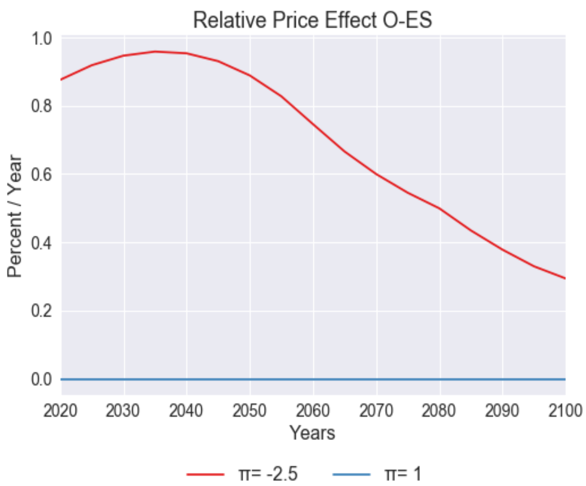

Figure 7: RPE of other non-market goods O_t with two different substitutability values.

Whilst the RPE is null in the case of perfect substitutability, by assuming complementarity between the two goods (solid red line), we can observe a RPE in the order of 0.9% per year in 2020, peaking at 0.99% by year 2035, and decreasing to 0.4% per year by 2100. The decrease of the RPE has to be attributed to increasingly higher mitigation efforts which slow down the divergence in growth rates of the goods, and therefore their relative scarcities. This analytical framework suggests that, to correct the imbalances in relative price changes, policymakers should impose three different good-specific discount rates: the RPE of overall non-market goods from Drupp and Hänsel (2021), which we disentangled in our analysis, dictates the difference in good-specific discount rates between market goods and biodiversity-related goods, whilst the RPE of other non-market goods indicates the discrepancy between the discount rate of other non-market goods and biodiversity-related ecosystem services.

7.2 **Evaluating Optimal Climate Policy within the Planetary Boundaries Framework**

In this section we evaluate the compliance of optimal climate policy trajectories with the biophysical limits of the climate change and biosphere integrity boundaries.

7.2.1 Assessing the Impacts of Optimal Climate Policy on the Climate Change Boundary

As a first step, we evaluate how the projected atmospheric CO_2 concentration pathways resulting from optimal emission trajectories comply with the climate change boundary. The control variable of the boundary is expressed in ppm CO_2 in the atmosphere.
DICE models the carbon cycle with a three-equations system, where carbon concentrations in the atmosphere $M_{AT\,t}$ are expressed in gigatons of carbon (GtC) with respect to year 1975 and given by the following equation:

$$M_{AT\,t} = E_t + \upsilon_{11} \cdot M_{AT(t-1)} + \upsilon_{21} \cdot M_{UP(t-1)} \qquad (7.2.1.1)$$

E_t represent $GtCO_2$ emissions, υ_{ij} are the flow parameters of carbon between reservoirs, and M_{UP} represents carbon concentration levels in the upper ocean. To convert the GtC measure in the control variable of the planetary boundary, we multiply $M_{AT\,t}$ by the equivalent value in ppm of CO_2 contained in one gigaton of carbon, which equals 0.469 (parts per million).
Figure 7 depicts estimated CO_2 concentration pathways according to optimal emission pathways and their compliance with the physical limits of the climate change boundary in a timeframe of 80 years.

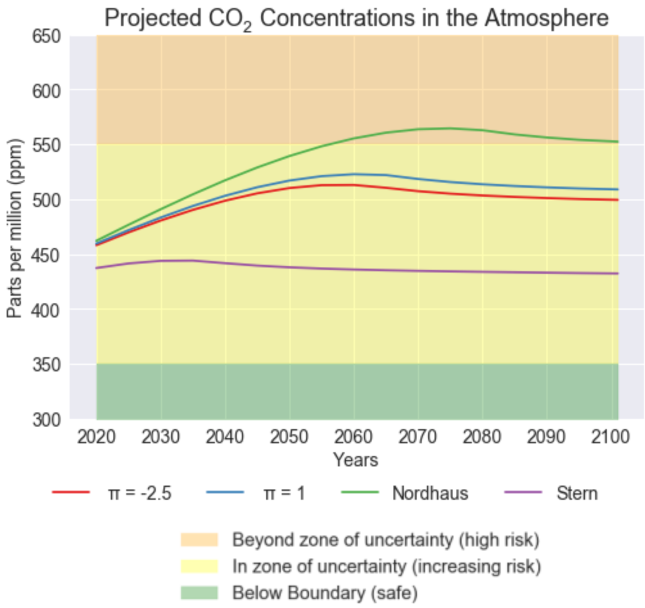

Figure 8: Evaluating compliance of optimal emission trajectories with the climate change planetary boundary.

The figure reveals the two key thresholds of the planetary boundary: 350 and 550 ppm. The former delimiting the 'zone of uncertainty', the latter the 'zone beyond uncertainty' where society enters the high-risk zone.

In the Stern model calibration CO_2 concentrations are almost linear and stabilize at 444 ppm until the end of the century. On the other hand, CO_2 concentrations resulting from the relative prices' model and Nordhaus calibration are hump shaped. In the extended model with low (perfect) substitutability between non-market goods, CO_2 concentrations peak at 512.97 (522.78) ppm around mid-century and stabilize to circa 500 (510) ppm towards the end of the century. In the Nordhaus calibration, CO_2 concentrations reach their maximum levels around 2075, peaking at 564,58 ppm breaking the 'zone of uncertainty' boundary and leading society 'Beyond the zone of uncertainty'. Moreover, Figure 7 shows that none of the CO_2 concentration pathways shows sufficient convergence toward the safe zone if the boundary.

7.2.2 Assessing the Impacts of Optimal Climate Policy on the Biosphere Integrity Boundary

As a second step, we evaluate to which extent the biodiversity loss resulting from optimal global warming exacerbates the pressures on the biosphere integrity boundary.

As described in Section 2, the biosphere integrity boundary is quantified via a dual approach: the control variables are extinctions per million species per year (E/MSY) and Biosphere Integrity Index, currently under development. Given that the BII is not quantifiable at the present moment, we will limit the analysis to measuring the loss in genetic diversity due to global warming. Moreover, as in our analysis we have focussed on the relationship between climate change and species extinctions and have neglected other important drivers of biodiversity loss, such as land-use change or ocean acidification, the assessment on the biosphere integrity boundary will be only limited.

The biodiversity loss function described by Equation 6.2.2.1 yields the percentage of total species loss B_t and indicates the cumulative climate change-driven extinction percentage for a given temperature increase T_t. The following figure depicts the projected climate change-driven biodiversity loss according to optimal temperature increases:

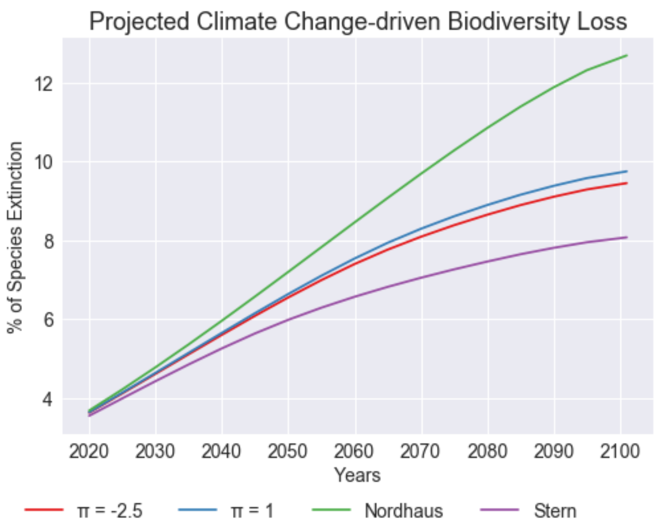

Figure 9: Species extinction predictions according to optimal global atmospheric temperature increases.

The graph suggests that 3.6% of species might have already faced extinction due to global average atmospheric temperature increases with respect to 1900 levels. Additionally, it projects that in 2100, according to the Nordhaus and Stern optimal temperature increases, 13% and 8.2% of species will risk extinction. Moreover, it displays a slight difference in the extinction rates for the extended models: in the model calibration featuring a complementarity assumption between non-market goods, the climate change-driven extinction rates in 2100 are 9.45%, 0.3% lower compared with a model featuring perfect substitutability.

Next, we assess the impacts on the biosphere integrity boundary. To obtain the control variable of the boundary, measured in extinction rates per million species per year, we need to transform the total extinction rate B_t. The control variable measures extinction rates occurring per year if species were one million.

Given that the output in DICE is given a 5-years interval, to obtain E/MSY, we find the average variation in extinction rates for each period and multiply this value by one million. Formally, the control variable is given by the following:

$$E / MSY_t = \frac{B_t - B_{t-1}}{t_{step}} \cdot 10^6 \qquad (7.2.2.1)$$

Figure 10 depicts the evolution of the biosphere integrity control variable over a period of 80 years:

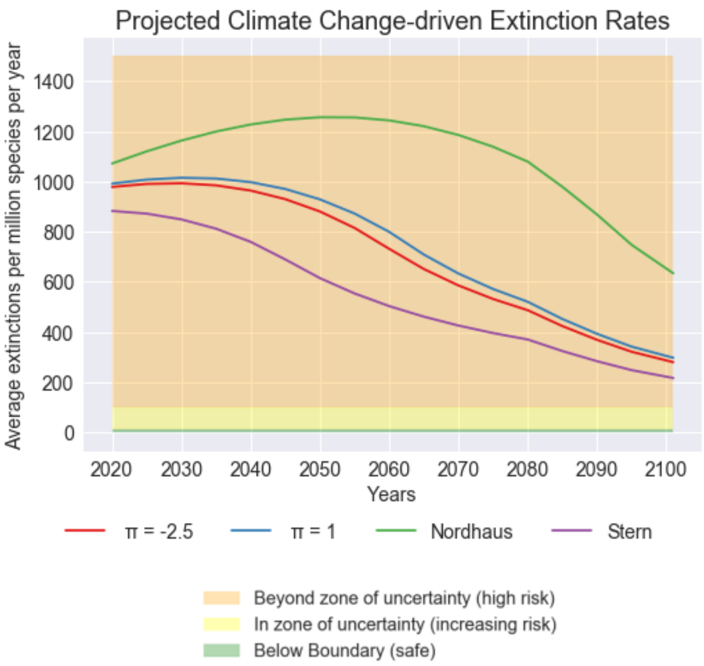

Figure 10: Assessing the impacts of climate change-driven biodiversity loss on the biosphere integrity boundary.

The figure proves the critical status of the biosphere integrity boundary and furthermore highlights the non-compliance of optimal climate policy trajectories with the planetary boundary.

Moreover, Figure 10 indicates decreasing climate change-driven extinction rates for the relative prices and the Stern models. Starting from the value of 1000 E/MSY, the trajectories of the relative prices model reach values of 258.7 and 243.6 E/MSY by the end of the century with complementarity and perfect substitutability assumption, respectively. In the Stern model calibration extinction rates decrease to 189 E/MSY by the end of the century. In contrast, with optimal temperature increases resulting from the Nordhaus calibration, extinction rates trajectories are hump-shaped. Extinction rates dramatically increase to 1256.9 AE/MSY in the year 2055 before starting to decline until 2100.

In summary, the graph confirms the criticality status of this planetary boundary and provides a disturbing picture of the projected biodiversity loss in the current century.

Furthermore, it is noteworthy to highlight that our analysis of the impacts of optimal climate policy is restricted on of change-driven biodiversity and neglects all other drivers of species extinctions. Therefore, the figures suggested by the graph should be interpreted with care: adding all other biodiversity loss drivers in the analysis would increase the resulting pressures on the boundary.

8 Summary of Results

In this section we summarize the key results of this work.

In Section 2 we introduce climate change and biodiversity loss in the planetary boundaries framework and explore their interrelationships on a physical basis. In Section 3 we tackle the task of attributing an economic value to biodiversity. For this purpose, we conduct a literature review to explore the interdependencies between biodiversity and human well-being and emphasise the value of species in light of their supporting function to ecosystems which provide benefits to society. Moreover, we review studies that attempt to attribute an economic value to biodiversity-related ecosystem services. In Section 4 we introduce substitutability as a fundamental tool in welfare economics and its importance in the framework of relative price changes. Additionally, having presented the role centrality of the substitutability between market and non-market goods in the study of relative price increases, we introduce arguments to support high and low substitutability between biodiversity-related ecosystem services and other non-market goods. Next, in Section 5, having presented IAMs as a central tool for climate change cost-benefit analyses and highlighted their main characteristics, we introduce the DICE model with its core elements. Furthermore, after having analysed how non-market damages are included in the model, we conduct a literature review to present updated non-market damage estimates. Next, in Section 6, we present the model extensions building on Drupp and Hänsel (2021) and propose a damage function for biodiversity-related ecosystem services. Based on the findings of the initial sections of the thesis, we propose a function in which climate-change biodiversity loss affects the level of ecosystem functioning and therefore the level of ecosystem services that affect utility levels positively.

In Section 7, we evaluate the impacts of the explicit inclusion of biodiversity-related ecosystem services and other non-market goods on climate policy and test the sensitivity of the results to a high and a low substitutability assumption between these amenities. Our analysis finds that the difference in optimal climate policy trajectories between a model featuring perfect substitutability among the non-market goods and a model featuring these amenities only comprehensively is negligible. In contrast, we find that assuming complementarity between non-market goods has considerable effects on climate policy stringency. Indeed, assuming weak substitutability between biodiversity-related ecosystem services and other non-market goods translates in a net zero-emissions economy five years in advance, in lower peak temperatures (0.09 °C) and SCC 9.6 (11.6) percent higher in 2020 (2100) compared to a scenario where these goods are perfectly substitutable.

We subsequently attribute the cause of a stricter climate policy to relative price changes of the explicitly represented non-market goods. In fact, with the complementarity assumption, we find a positive relative price effect of other non-market goods. To correct for the imbalance of relative price changes, we propose a three-goods discounting approach. Reflecting the estimated relative price changes, we suggest a discount rate for other non-market goods 0.9 percent (0.4 percent) per year lower than biodiversity-related ecosystem services in 2020 (2100). Moreover, following the results from Drupp and Hänsel (2021), we attribute to biodiversity-related ecosystem services a discount rate 4.1 percent (1.9 percent) lower in 2020 (2100) compared to market goods.

In a second step of the analysis, we evaluate the sustainability of optimal climate policy pathways by projecting their impacts on the climate change and biosphere integrity boundaries. Concerning the climate change boundary, our analysis reveals that optimal emissions' trajectories for the relative prices model featuring low (perfect) substitutability and Stern calibration are projected to maintain atmospheric CO_2 concentration levels below the 'high risk' zone, stabilizing in the zone of 'increasing risk' at circa 500 (510) ppm and 440 ppm, respectively. On the other hand, in the Nordhaus model, CO_2 concentration levels are projected to peak at 564,58 ppm and break the 'zone of uncertainty' threshold. Moreover we find that none of the model trajectories shows a sufficient convergence toward the 350 ppm boundary.

Secondly, we evaluate how optimal temperature increases affect species' extinction rates and quantify the impacts for the biosphere integrity boundary. Using a climate change-driven biodiversity loss function calibrated according to the estimates from Urban (2015), we estimate temperature-related extinction rate expressed as cumulative species losses. Next, we convert this measure into extinctions per million species per year.

Our analysis reveals that optimal temperature increases in the Nordhaus calibration are projected to increase extinction rates to 1256.9 E/MSY in 2055 and lead to 13% cumulative species losses in 2100. In contrast, starting from the value of 1000 E/MSY, the extinction rates of the relative prices model decline to values of 258.7 and 243.6 E/MSY by the end of the century leading to 9.45% and 9.75% total extinction rates, with complementarity and perfect substitutability assumption, respectively. In the Stern model calibration extinction rates decrease to 189 E/MSY by the end of the century and total species extinctions amount to 8.07%. In light of the estimated extinction rates, well above boundary extremes in all model calibrations, our analysis confirms the criticality of the boundary status.

9 Discussion

In this section, we address the methodological issues found in the development of this thesis and discuss how the assumptions made in the course of the work limit our analysis. We also take the opportunity to identify challenges for future research work.

The integration of climate change-driven biodiversity loss in cost-benefit analyses presents several methodological and conceptual challenges. In this framework, the difficulty in measuring biodiversity represent a major caveat. Moreover, understanding the physical impacts of climate change specifically on species is a complex task; according to Urban (2015), the results of studies analysing causal effects between temperature increase and biodiversity loss are highly sensitive to assumptions about species dispersal scenarios and often neglect important factors such as interrelationships among species and evolutionary processes.

Accordingly, future research work should address the role of uncertainty in determining the impacts of climate change on species extinction and make greater efforts in quantifying biodiversity.

Furthermore, having accounted only for biodiversity loss caused by climate change, the impact assessment of optimal climate policy on the biosphere integrity boundary was partial. For a more comprehensive assessment, further research should analyse how other drivers of biodiversity loss, such as land-use change, ocean acidification, and alien species invasion, may add pressures to the boundary.

Moreover, in our analysis we have indirectly placed an economic value on biodiversity in light of its role in supporting ecosystem functioning. In this perspective, according to IPCC (2007), a major limitation in the comprehension of the interrelationships between biodiversity loss and human well-being is a confined understanding of the interactions between species and ecosystems. In this regard, future research should investigate the role of nonlinearities, feedback loops, and tipping points in the interrelationships between species and ecosystem functioning and analyse how different assumptions on the relationship between biodiversity and ecosystem functioning may impact climate policy optimality.

Another restriction in this context is that placing an economic value on species presents a challenge from both an economic and moral perspective. In fact, values of ecosystem services estimated via non-market valuation techniques such as contingent valuation are often biased (Bosello and De Cian 2014). Additionally, from a moral standpoint, the assessment of species based on their use-values represents a purely anthropocentric perspective that may neglect important aspects of biodiversity conservation, such as its existence and bequest values (Dasgupta 2021). Therefore, more comprehensive cost benefit analyses should include different perspectives on the value of biodiversity and propose alternative methods to improve the quality of the economic valuation of non-market amenities.

In addition, in the relative price study conducted in this thesis, the substitutability parameter between biodiversity-related ecosystem services and other non-market goods had central importance. The lack of empirical data affected the analysis insofar, as experimental values were used without any specific foundation. In this regard, to better quantify the impacts of non-market goods relative price changes on optimal climate policies more precisely, future work should propose novel methodologies to determine plausible substitutability parameter values.

Similarly, a critical role in determining the magnitude of changes in relative prices is played by the non-market damages estimates (Drupp and Hänsel 2021). Yet, the difficulty in estimating this parameter specifically for biodiversity-related ecosystem services and other non-market goods adds uncertainty to the estimation of relative price changes. Indeed, in order to assign an economic value to the explicitly represented non-market amenities, our analysis relied on a cascade of assumptions. From this perspective, future research should aim to propose new approaches to estimate climate change damages to ecosystem services related to biodiversity and to other non-market goods.

Moreover, future work should revisit the calibration approach of non-market damages proposed in this thesis and test the sensitivity of the results based, for example, on alternative non-market damage estimates. In this way, it could be for example investigated to which extent the amplification of non-market damages affects relative price changes and therefore climate policy optimality, to subsequently analyse to which degree the novel identified optimal pathways comply with the physical constraints of the planetary boundaries. In this context, Hänsel et al. (2020) demonstrated that the attainment to stricter climate policy goals, such as limiting atmospheric temperature increases to 1.5°C can become optimal when properly updating DICE with higher damage estimates and extending the model with alternative constructs. In this sense, via the usage of updated non-market estimates and alternative calibration approaches, novel analyses could investigate, under which assumptions efficiency aspects of climate policy can satisfy the physical constraints of the planetary boundaries.

10 Conclusions

The aim of this thesis was to investigate the role of biodiversity loss for the integrated assessment of climate change and to evaluate the sustainability of optimal climate policy trajectories within the planetary boundaries framework.
In the extended model we have introduced in this thesis, we analysed the dynamics of relative price changes and studied the implications of limited substitutability between biodiversity-related ecosystem services and other non-market goods for optimal climate policy.
Additionally, we have provided an analytical framework to project the impacts of the novel identified optimal climate policies on the climate change and biosphere integrity boundaries. Our analysis reveals that, if it can be argued for limited substitutability between biodiversity-related ecosystem services and other non-market goods, stricter optimal climate policy should be observed as compared to a case where these goods are perfectly substitutable or are accounted in the social welfare only in an aggregated form. In fact, assuming perfect substitutability and neglecting complementarities between biodiversity-related ecosystem services and other non-market goods could lead to an underestimation of the SCC by 9.6 percent and 11.6 percent in 2020 and 2100, respectively.
Additionally, in the presence of limited substitutability, by evaluating numerically their RPE, our framework proposes to discount other non-market goods at a 0.9 (0.4) percent lower rate per year in 2020 (2100) compared to biodiversity-related ecosystem services. Furthermore, building on the results from Drupp and Hänsel (2021), our model suggests discounting biodiversity-related ecosystem services goods at a 0.9 (0.4) percent lower discount rate per year in 2020 (2100) compared to market goods. Our results therefore imply that, when the complementarity hypothesis between the explicitly represented non-market amenities can be backed up, public cost-benefit analyses should use three different discount rates.
From this perspective, we once more emphasize the need of conducting more research efforts to understand the interrelationships between biodiversity-related ecosystem services and other non-market goods and therefore to find more information on the substitutability parameter. Moreover, in this regard, in light of the arguments presented in the literature review, arguing for perfect substitution possibilities between biodiversity-related ecosystem services and other non-market goods seems unplausible.
Finally, in analysing the sustainability of the novel identified optimal climate policy trajectories within the planetary boundaries framework we conclude that none of the newly identified pathways satisfies the physical constraints of the climate change and biosphere

integrity boundaries. Therefore, with the limiting key assumptions and calibration approaches of the models presented in this thesis, the achievement of the planetary boundary's stabilization goals is suboptimal from an economic perspective. Nevertheless, our analysis provides substantial arguments on the need to evaluate relative price changes between non-market amenities, offering to policymakers valid insights to support and strive for more stringent climate policies that help maintaining humanity in the "safe operating space".

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Appendix

A.1 Calibration of Non-Market Goods Damages

As a first step, we obtain the ‘all other non-market goods’ damage scaling parameter ψ by comparing two different model specifications: on the one hand, a model in which non-market damages, that include O_t and ES_t are perfectly substitutable with damages to market goods D^{Ω} and are included in consumption. On the other hand, a model in which climate change damages are attributed to market goods D^{κ}, other non-market goods D^{ψ} and ecosystem services D^{ES} specifically:

$$W_0\left(ES_0, O_0, (1-D^{\Omega})\cdot C_0, L_0\right)=W_0\cdot\left(\left(1-D^{\psi}\right)\cdot O_0, (1-D^{ES})\cdot ES_0, (1-D^{\kappa})\cdot C_0, L_0\right) \tag{A.1.1}$$

Which can be rewritten in the following:

$$\alpha\cdot\left(\left(\phi\cdot ES_0^{\ \pi}+(1-\phi)\cdot O_0^{\ \pi}\right)^{\frac{1}{\pi}}-\overline{EQ}\right)^{\theta}+\left(1-\alpha\right)\cdot\left(\left(1-TD\right)\cdot C_0\right)^{\theta}=\alpha\cdot\left(\left(\phi\cdot\left(\left(1-D^{ES}\right)\cdot ES_0\right)^{\pi}+\left(1-\phi\right)\cdot\left(\frac{O_0}{1+\psi T^2}\right)^{\pi}\right)^{\frac{1}{\pi}}-\overline{EQ}\right)^{\theta}+\left(1-\alpha\right)\cdot\left(\left(1-D^{\kappa}\right)\cdot C_0\right)^{\theta} \tag{A.1.2}$$

Next, we find ψ as a residual:

$$\psi = \left(\frac{1}{T^{2}}\right)\cdot\left\{\left[\left(\frac{1}{1-\phi}\right)\cdot\left(\left(\left(\frac{\alpha\cdot\left(\left(\phi\cdot ES_{0}^{\pi}+(1-\phi)\cdot O_{0}^{\pi}\right)^{\frac{1}{\pi}}-\overline{EQ}\right)^{\theta}+(1-\alpha)\left((1-TD)\cdot C_{0}\right)^{\theta}-(1-\alpha)\cdot\left((1-D^{\kappa})\cdot C_{0}\right)^{\theta}}{\alpha}\right)^{\frac{1}{\theta}}+\overline{EQ}\right)^{\pi}-\phi\cdot\left((1-D^{ES})\cdot ES_{0}\right)^{\pi}\right)^{-\frac{1}{\pi}}\cdot O_{0}\right]-1\right\} \quad \textbf{(A.1.3)}$$

As a second step, we find ρ by rewriting Equation (A.1.2) in the following:

$$\alpha\cdot\left(\left((1-\phi)\cdot O_{0}^{\pi}+\phi\cdot ES_{0}^{\pi}\right)^{\frac{1}{\pi}}-\overline{EQ}\right)^{\theta}+(1-\alpha)\cdot\left((1-D^{\Omega})\cdot C_{0}\right)^{\theta}=\alpha\cdot\left(\left[(1-\phi)\cdot\left(\frac{O_{0}}{1+\psi T^{2}}\right)^{\pi}+\phi\cdot\left(ES_{0}\cdot(1-\rho\cdot B^{\gamma}_{(T=3^{\circ})})\right)^{\pi}\right]^{\frac{1}{\pi}}-\overline{EQ}\right)^{\theta}+(1-\alpha)\cdot((1-D^{\kappa})\cdot C_{0})^{\theta} \quad \textbf{(A.1.4)}$$

Finally, we calculate biodiversity levels for a temperature increase of 3°C and find the ecosystem services damage scaling parameter ρ by solving by ψ from Equation (A.1.3):

$$\rho = -\left(\frac{1}{ES_0 \cdot B_{T=3}{}^{\gamma}}\right) \cdot \left\{ \left[\left(\frac{1}{\phi}\right) \cdot \left(\left(\left(\frac{\alpha \cdot \left(\left(\phi \cdot ES_0{}^{\pi} + (1-\phi) \cdot O_0{}^{\pi}\right)^{\frac{1}{\pi}} - \overline{EQ} \right)^{\theta} + (1-\alpha)\left(\left(1-TD\right) \cdot C_0\right)^{\theta} - (1-\alpha) \cdot \left(\left(1-D^{\kappa}\right) \cdot C_0\right)^{\theta}}{\alpha} \right)^{\frac{1}{\theta}} + \overline{EQ} \right)^{\pi} - (1-\phi) \cdot \left(\frac{O_0}{1+\psi T^2}\right)^{\pi} \right)^{\frac{1}{\pi}} \cdot O_0 \right] - 1 \right\} \quad \textbf{(A.1.5)}$$

A.2 Determining the Biodiversity-Ecosystem Functioning Curvature Coefficient

O'Connor et al. (2017) estimates the biodiversity-ecosystem functioning curvature coefficients to account for the variation in ecosystem functioning due to decrease in biodiversity levels B^x (x = 0.26, 95% CI: 0.16, 0.37).
The equivalent coefficients that accounts for variation in ecosystem functioning due to increases in biodiversity loss are obtained with following transformation:

$$1 - B_{loss}{}^{\gamma} = B^{x} \quad \textbf{(A.2.1)}$$

Solving for γ we can find the coefficient for biodiversity loss.

$$1 - B_{loss}{}^{\gamma} = B^{0.26} \triangleright \gamma = 1.58 \quad \textbf{(A.2.2)}$$