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Different Shades of Green: Unveiling Opportunities and Challenges of Green Growth Solutions in Developing Countries

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To Eulalia and Elio.

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Thesis abstract

From the highest peak on the Himalayas to the sandy shores of the Caribbean islands, the impact of climate change is now visible everywhere. The unpredictability of this phenomenon is forcing governments around the globe to curb emissions and limit global warming to a critical range of 1.5°C to 2°C. To tackle this challenge, the concepts of 'green growth' and 'green economy' policies have emerged as potential solutions. These strategies offer a glimmer of hope by providing a way to address climate change while fostering economic growth. However, after over ten years of debating the many virtues the green growth offers, one question remains: How effective is it?

This thesis addresses this concern by exploring green growth relevance, adequacy, and impact in promoting sustainable and environmentally friendly growth in developing countries. Three case studies are developed to investigate the opportunities and limitations it presents. In Cambodia, the research evaluates the economic and environmental impact of the national carbon-neutrality strategy. In small island states, the analysis reveals the role of nature-based solutions in sustaining growth. In India, and focusing on New Delhi, this study looks at the challenges and solutions posed by megalopolis's increasing urban population and waste mismanagement practices.

This thesis finds that carbon neutrality is an elusive goal, particularly for developing countries. Albeit the efforts to reach it are virtuous in many respects, they fail to deliver the Paris Agreement targets. Meanwhile, green solutions are particularly helpful and cost-effective in sustaining and protecting the livelihoods of the coastal population. The conservation of these ecosystems is further desirable from an ecological and economic standpoint as it vastly exceeds the returns from grey infrastructure solutions. Finally, a circular approach to waste management delivers substantial health and environmental benefits while providing considerable economic benefits.

The thesis concludes by providing some policy implications. It suggests that simultaneously achieving Paris targets while sustaining growth is challenging in developing countries. More should be done to sustain the technological transition and policy enforcement in these countries. Nevertheless, policy enforcement in itself may not be enough. To be actionable and credible, green growth policies must be supported by behavioural change campaigns and a more balanced approach to the way societies produce and consume.

Riassunto della tesi

Dalla vetta più alta dell'Himalaya alle coste sabbiose delle isole caraibiche, l'impatto del cambiamento climatico è ormai visibile ovunque. L'imprevedibilità di questo fenomeno sta costringendo i governi di tutto il mondo a ridurre le emissioni e limitare il riscaldamento globale a un intervallo critico compreso tra 1,5°C e 2°C. Per affrontare questa sfida, i concetti di "crescita verde" e di "economia verde" sono emersi come possibili soluzioni. Queste strategie offrono un barlume di speranza fornendo un modo per affrontare il cambiamento climatico promuovendo al contempo la crescita economica. Tuttavia, dopo oltre dieci anni di dibattiti sulle molte virtù offerte dalla crescita verde, rimane una domanda: quanto è efficace?

Questa tesi cerca di rispondere a questa domanda esplorando la rilevanza, l'adequatezza e l'impatto della crescita verde nella promozione di una crescita sostenibile nei paesi in via di sviluppo. Vengono sviluppati tre casi di studio per analizzare le opportunità e i limiti che questa presenta. In Cambogia, la ricerca valuta l'impatto economico ed ambientale della strategia nazionale di neutralità carbonica. Nei piccoli stati insulari in via di sviluppo, si studia il ruolo delle soluzioni basate sulla natura nel sostenere la crescita. In India, e concentrandosi su Nuova Delhi, si esaminano le sfide poste dall'aumento della popolazione urbana e le soluzioni alla cattiva gestione dei rifiuti.

Questa tesi rileva che la neutralità carbonica è un obiettivo sfuggente, in particolare per i paesi in via di sviluppo. Sebbene gli sforzi siano virtuosi sotto molti aspetti, quest'ultimi non riescono a garantire il raggiungimento degli obiettivi dell'Accordo di Parigi. Nel frattempo, le soluzioni verdi sono particolarmente utili ed economiche per sostenere e proteggere i mezzi di sussistenza della popolazione costiera. La conservazione di questi ecosistemi è auspicabile sia da un punto di vista ecologico ed economico, in quanto supera di gran lunga i rendimenti delle soluzioni infrastrutturali grigie. Infine, un approccio circolare alla gestione dei rifiuti offre notevoli benefici per la salute e l'ambiente, fornendo al contempo notevoli benefici economici.

La tesi si conclude fornendo alcune implicazioni politiche e sottolineando le difficoltà nei paesi in via di sviluppo di far collimare gli obiettivi di crescita economica con agli obiettivi ambientali di Parigi. In sintesi, si dovrebbe fare di più per sostenere la transizione tecnologica e l'applicazione delle politiche in questi paesi. Tuttavia, l'applicazione delle politiche di per sé potrebbe non essere sufficiente. Per essere attuabili e credibili, le politiche di crescita verde devono essere sostenute da campagne di cambiamento comportamentale e da un approccio più equilibrato al modo in cui le società producono e consumano.

List of papers

Paper no. 1

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Paper no. 3

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Abbreviations

BT	Benefit transfer
CBA	Cost-benefit analysis
CPI	Consumer price index
CV	Contingent valuation
EKC	Environmental Kuznets curve
ES	Ecosystem service
ESVD	Ecosystem services valuation database
GDP	Gross domestic product
Ha	Hectare
LCA	Life cycle assessment
LTS4CN	Long-term strategy for carbon neutrality
MSW	Municipal solid waste
NbS	Nature-based solution
NDC	Nationally Determined Contributions
PH	Porter's hypothesis
PNG	Papua New Guinea
PPP	Purchasing power parity
SAMEA	Social accounting matrix with environmental accounts
SD	Standard deviation
SIDS	Small island developing states
TEV	Total economic value
WTP	Willingness to pay

1 Introduction

1.1 Backgrounds

Within the larger debate concerning economic growth and environmental sustainability, this thesis investigates the coherence, potential and trade-offs between green growth and climate mitigation in developing countries. The aim is to verify whether some of the green growth policies and tools currently applied in emerging countries are consistent with the carbon neutrality targets outlined in the Paris Agreement. In particular, the thesis seeks to identify how policy-makers and civil societies could address increasing social inequality, environmental degradation and climate change effects – while sustaining economic development.

Climate change, global pandemics, financial crises, and market instabilities have brought to the forefront the need for institutional reforms, regulations, and ethical considerations regarding the way the global economy works (Breckenfelder *et al.*, 2023; Leach *et al.*, 2021). Neglecting to do so could lead to water shortages, resource supply challenges, pollution, climate change, and biodiversity loss, all of which could threaten our very survival on this planet (IPCC, 2023).

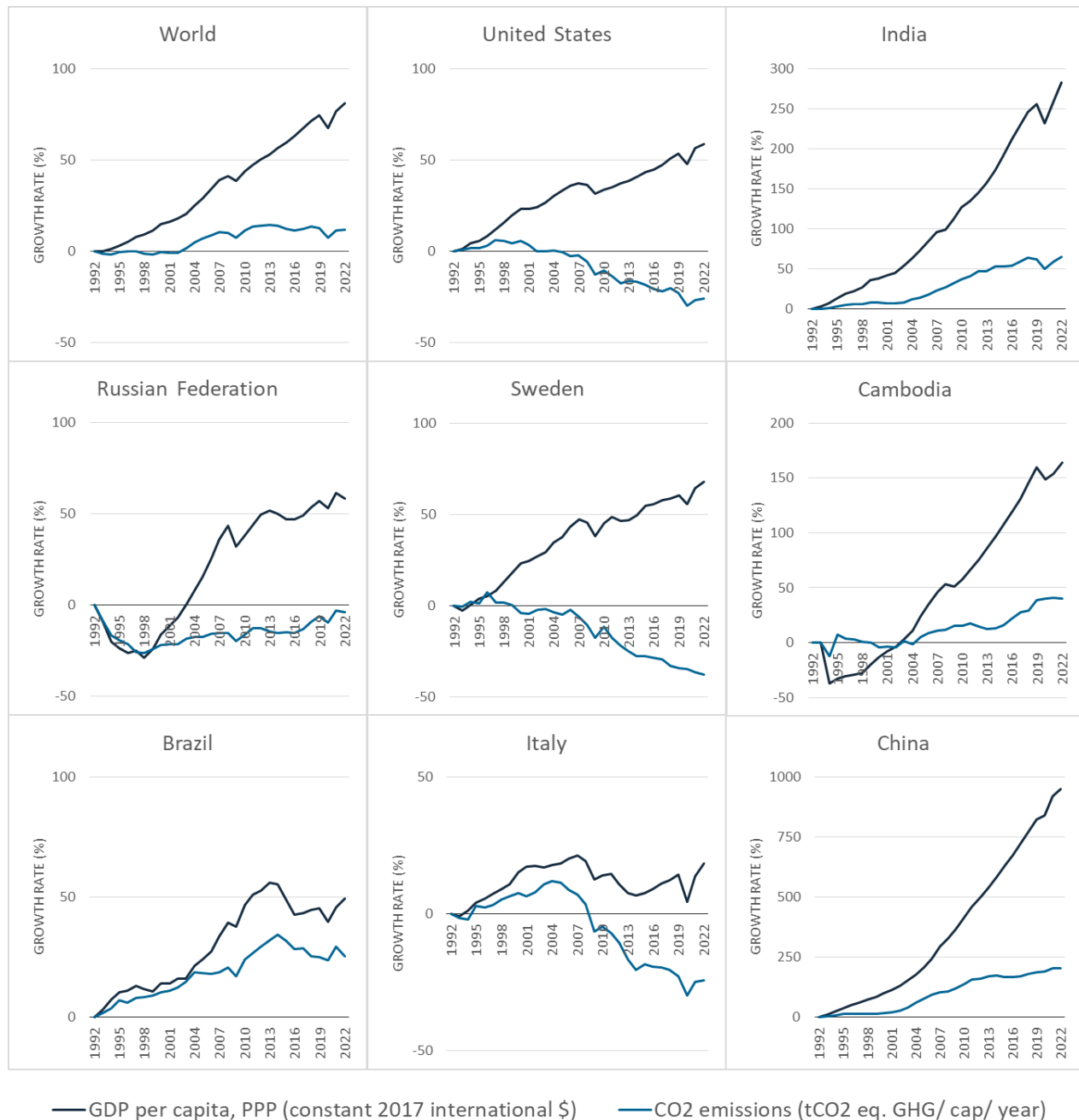
To guarantee an equitable life for future generations, we must develop innovative approaches to production and consumption and redefine our development concept. First, we should rethink how we measure welfare and possibly move away from GDP as an indicator of progress (Hoekstra, 2019). Secondly, we should reframe the economic growth debate around respecting ecological limits and planetary boundaries (Raworth, 2017a, 2017b; Sobkowiak, Senn and Vollmer, 2023). Failing to do so would exacerbate an already precarious situation. At present, more than half of the planetary boundaries are already exceeded (Richardson *et al.*, 2023), and climate change impacts riddle worldwide.

Green growth is seen as the possible solution to mitigate climate change and ensure sustainable economic growth by reaching carbon neutrality. However, much of the belief in achieving Paris targets through green growth depends on the capacity to decouple economic growth from greenhouse gas emissions. The latter requires reducing waste and energy consumption, shifting to renewable energy sources, reducing fossil fuel reliance, managing resources efficiently, and expanding the adoption of green technologies, products, and services.

However, there is growing concern about the progress made so far and increasing scepticism that the current roll-out pace of green growth solutions is adequate to meet Paris targets before hitting climate overshooting (King, Savin and Drews, 2023). A quick look at decoupling trends over the last 30 years shows improvements in advanced economies, some marginal results at the global

level and remarkable negative trends in developing countries (Figure 1). A recent analysis also reveals that we can only reach equitable and sustainable development within planetary boundaries by drastically increasing carbon productivity by 5% yearly (Stoknes and Rockström, 2018).¹ Currently, the only countries that have these results are Denmark, Finland, and Sweden. Opposite, the largest emitting countries and groups, such as the USA, India, China, the EU28 and the OECD members, fall far short of the required threshold (*Ibid.*).

Figure 1: Change in per capita GDP and CO2 emissions



Source: Own elaboration from data from World Bank (2024b) and EU JRC (2023)

¹ Carbon productivity (i.e. $\Delta \text{GDP} / \Delta \text{emission}$) is the inverse of carbon intensity.

Some studies already questioned the reliability of green economy as a development approach for emerging economies (Houssam *et al.*, 2023). Especially so given that green growth assumes a level of structural transformation and industrial innovation more appropriate for developed nations rather than developing countries (Barbier, 2016). The prevailing heterogeneous growth patterns across countries and world regions prove this. The development and adoption of energy and environment-related technologies is still mainly led by European countries. Meanwhile, American and Asian countries are partly catching-up, but their contribution to the global transition is getting smaller over time (Lin and Wang, 2019).

1.2 Thesis problem statement

Against the information provided in section 1.1, the problem statement of this thesis is the following:

“The evidence on the harmful impact of climate change on society and the environment is now overwhelming. Thus, governments of developed and developing countries are pressed to reduce GHG emissions and consumption footprint to maintain global warming within 1.5° - 2° Celsius. Green growth and green economy policies are seen as viable means to mitigate the impact of climate change while maintaining economic growth. This thesis investigates the relevance, adequacy and effectiveness of the green growth approach in the quest for sustainable and environmental-friendly growth in developing countries”.

1.3 Research objectives and questions

This research aims to verify whether some of the green growth policies and tools currently implemented in emerging countries are consistent with the carbon neutrality targets set forth in the Paris Agreement. In particular, the thesis seeks to identify how policy-makers and civil society could address increasing social inequality, environmental degradation and climate change effects while sustaining economic development.

To address these concerns, the research is organised around two objectives that will be met through an in-depth analysis of three case studies. Each case study tests some of the underlying hypotheses of the green growth agenda and the Paris Agreement in different country contexts. The hypotheses being tested concern the prevailing arguments brought forward by green growth supporters, namely that:

- Technological progress will allow for absolute decoupling of economic growth and carbon emissions and prompt a smooth transition towards carbon neutrality
- Trade-offs across economic growth, environmental conservation and social inclusiveness can be overcome through green economy policies, technologies and actions, ultimately leading to simultaneous increases in growth, income and environmental protection.
- afforestation and reforestation adequately sustain the transition to carbon neutrality
- Nature-based solutions are cost-effective solutions to climate change mitigation and withhold a great potential in sustaining and greening the economy in small island development state
- the blue economy provides new economic opportunities from the available coastal and marine resources.

The effects of climate change on societies and the environment are now evident. Therefore, governments in developed and developing countries are being called upon to reduce their emissions to keep global warming within 1.5°-2° Celsius. The principles of Green Growth (GG) are seen as viable means to achieve the goals of the Paris Agreement, mitigate the impact of climate change, and maintain economic growth.

Therefore, this thesis investigates green growth's relevance, appropriateness and effectiveness in ensuring environmentally sound growth in developing countries. The research is organised around two objectives:

1. Expanding knowledge on the potential and limitations of GG in supporting the transition of emerging countries to low-carbon economies
2. Identifying the environmental impact and possible solutions to the over-exploitation of natural resources caused by economic and population growth.

The first research objective (item 1) is justified for the evidence gaps and conflicting results shown in section 1.1, and it can be met by answering the following questions:

- a. Do green policies promote growth by safeguarding the environment and reducing emissions in low-income countries?
- b. What is the potential of nature-based solutions in supporting growth in developing countries?

To answer these questions, the research uses two case studies. The first case evaluates the credibility of Cambodia's long-term strategy for carbon neutrality (LTS4CN) and the adequacy of the

Nationally Determined Contributions (NDCs) to meet the 2030 climate targets. The analysis uses a social accounting matrix with environmental accounts (SAMEA) to estimate the economic and environmental impacts of the LTS4CN.

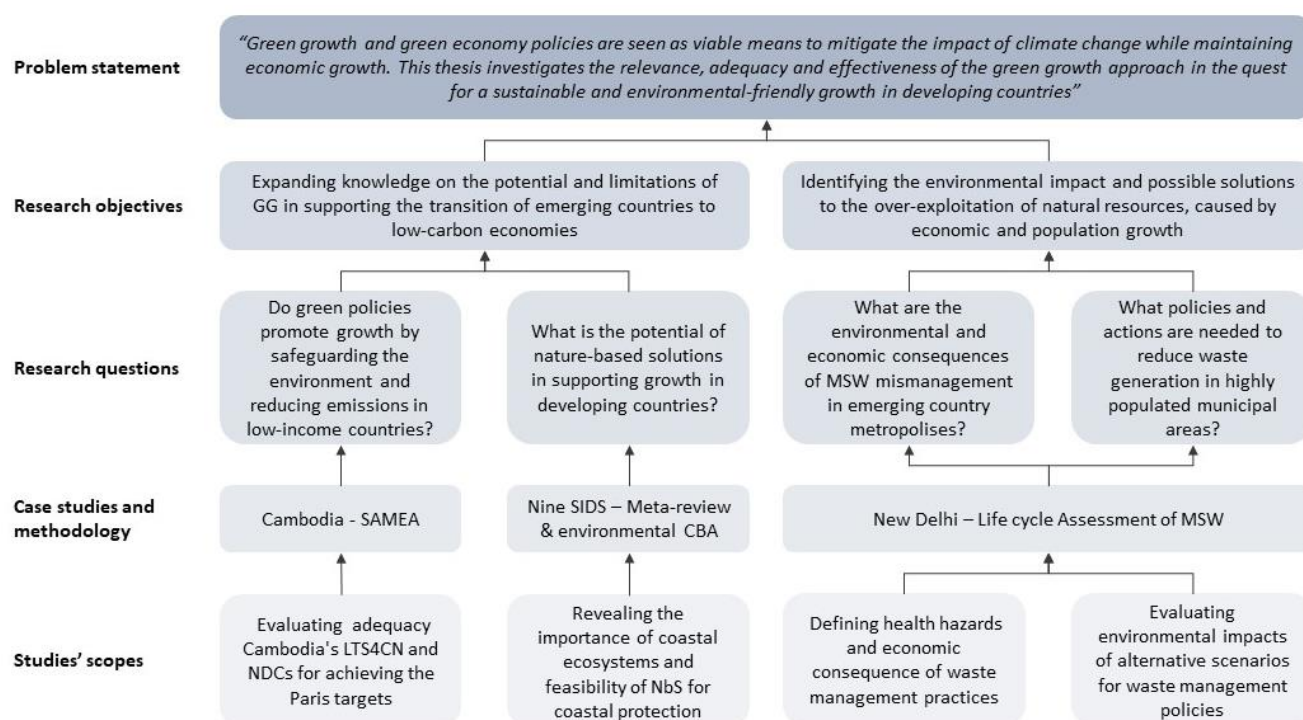
The second case study analyses the importance of coastal ecosystems (mangroves, coral reefs, and seagrass beds) in ensuring economic well-being, coastal protection, and environmental security in nine small island developing states (SIDS). The study uses a meta-review of existing literature and an environmental cost-benefit analysis to verify the feasibility of green vs. grey coastal protection solutions.

The second research objective (item 2) can be achieved by answering the following questions:

- c. What are the environmental and economic consequences of municipal solid waste mismanagement in emerging country metropolises?
- d. What policies and actions are needed to reduce waste generation in highly populated municipal areas?

The case study on New Delhi answers these questions by analysing the prominence of circular economy approaches applied to municipal solid waste in highly populated areas. The research uses a life cycle assessment of municipal solid waste, thereby defining environmental impacts and formulating alternative scenarios for waste management policies. Figure 2 provides a summary of the research approach used.

Figure 2: Thesis research approach



1.4 Research scope

The research scope is to fill the knowledge gap around the effectiveness of green growth in tackling global warming and promoting a sustainable transition to carbon neutrality. The focus is on verifying the approach's relevance within the context of Southeast Asia and Small Islands developing economies. Ultimately, this research also aims to inform policy-makers and development practitioners about the research issues and objectives defined above (see chapters 1.1 – 1.3).

This thesis evaluates the results that could be obtained by implementing climate change mitigation strategies. Hence, the methodological approach of this research is that of quantitative policy analysis. The methods being used (i.e. SAMEA, CBA and LCA) cover both the environmental and the economic domains and are used to verify climate change mitigation and adaptation policies and actions.

Meanwhile, this research will not look into the relevance and effectiveness of green growth in developed countries and will not evaluate the relevance and effectiveness of the policy formulation process. Similarly, not all green growth policies could be analysed, and some notable solutions (e.g. renewable energy) were excluded.

1.5 Thesis outline

Following the introduction provided in chapter 1, the thesis specifies the conceptual and theoretical framework relating to the research. Chapter 2 provides an overview of the main elements, concepts, and critics that have characterised the debate on green growth and the green economy since its inception.

Chapter 3 explains the research approach and the data analysis procedures. Additional details on the estimate procedures are not included here but are provided along with the case study details.

Chapter 4 outlines the case studies and provides a detailed analysis of the results. Each case study is presented as a stand-alone research paper, including an introductory part, a methodological section and a results analysis and discussion part.

Chapter 5 concludes with an analysis of the thesis's relevance, main implications, and limitations. Additionally, the chapter provides food for thought on policy implications and future research areas open up by the evidence gaps left behind by this research.

2 Conceptual and Theoretical framework

This chapter offers a review of the key theoretical and conceptual elements concerning the larger green transition debate. Chapter 2.1 retraces the origins of green growth and provides additional elements concerning the connection between green growth and sustainable development. Chapter 2.2 explores the main definitions and interpretations that have been given to green growth and green economy concepts over the years. This chapter also covers technical elements concerning decoupling, Porter's hypothesis and the Environmental Kuznets curve. It concludes by discussing the main arguments in favour and against green growth, including elements from the post-growth debate.

2.1 Green growth origins and background

2.1.1 Advancements in economic thinking: from growth limits to sustainable development

The idea of green growth originates in the wider discussion of growth limits. This discussion can be traced back to the end of the 18th century, when Thomas Malthus (1798) warned against the potential detrimental effects of prolonged population growth and limited food availability in his famous work, "An Essay on the Principle of Population." Malthus argued that a world in which population growth exceeded food production and resource availability was destined for disaster and famine. Following, Jevons (1865), Marshall (1890) and Hotelling (1931) further analysed the

constraints and failures resulting from the competitive use of limited resources, laying the foundations of natural resource economics.

These concerns remained relevant until the early 20th century, when they began to subside thanks to the Green Revolution brought forward by Norman Borlaug in 1944, with the application of chemicals, fertilisers, and new agronomic techniques. Meanwhile, economists began to consider that technological advancements could allow for an almost perfect substitution of capital. Economic growth would not be hampered by capital scarcity and could be expanded thanks to technological advancement and progress. As a result, consumers could substitute any capital (economic, environmental, and social) whenever its scarcity increased its price.

The neoclassical growth models that emerged were the result of contributions made by Ramsey (1928), Solow (1956, 1957), Swan (1956), Koopmans (1965) and Denison (1967). According to the standard neoclassical growth model, the steady-state growth rate of the economy beyond any technological progress is zero. This means that traditional macroeconomic policies, such as government investment, can affect per capita income but not impact the economy's long-term growth rate. Technological advancements in manufacturing are not openly explained in the model and are considered a "black box." However, continuous exogenous technological improvements can offset the adverse effects of the diminishing marginal productivity of capital, leading to long-term growth. Despite this, neoclassical models continue to perceive the environment as a limiting factor to growth, while the population growth rate – which is defined exogenously – determines the level of real per capita income.

From there, and following the fast population growth and increasing pollution around the globe, it emerged the branch of the economics theory investigating the sustainability and natural limits of growth. Several contributions analysed the effect of economic activity on environmental quality and resource depletion (Boulding, 1966; Commoner, 1971; Daly, 1973; Ehrlich, 1968; Georgescu-Roegen, 1971; Meadows, Randers and Behrens III, 1972). One of the main conclusions from these early studies was that unlimited growth is impossible on a planet with a finite amount of resources and a growing population.

In the 1970s, Inglehart observed that economic development in Western societies was pushing for a societal value change, with younger generations favouring collective immaterial values (i.e. autonomy, self-expression, freedom and healthy environment) to the individual materialist needs (Inglehart, 1971, 1977). Concurrently, Dasgupta and Heal (1974), Nordhaus (1974), Solow

(1974) and Stiglitz (1974) integrated the environment into the production function and investigated the optimal use of scarce resources in endogenous growth models.

Moving forward, the prevailing argument in the early 1990s favoured a vision where technological progress and market mechanisms could lessen the effect of natural resource scarcity on long-term economic growth. Natural capital was no longer perceived as a limiting factor – rather as a productive capital – which, when preserved and regulated through environmental policies, would lead to long-term growth (Bovenberg and Smulders, 1996). Similarly, Grossman and Krueger (1991; 1995) found that economic growth could be compatible with environmental quality and that as society becomes more economically developed, it starts placing greater value on non-economic values, such as social and environmental equity. The authors showed an inverted U-shaped relationship between per capita income and environmental quality. Based on it, environmental degradation initially increases with economic growth before declining again when the economy evolves from a product- to a service-based economy. The relationship is known as the Environmental Kuznets Curve (EKC), named after the Kuznets curve, which shows a similar relationship between economic inequality and development.

As a result, balancing economic growth and environmental conservation became the main focus of international discussions. The concept of Sustainable Development originated from the 1972 UN Conference on the Human Environment. However, it gained significant popularity only after the release of the Brundtland Report (1987) and the 1992 UN Conference on Environment and Development (UNCED). The Brundtland report was the first to emphasise the need for a comprehensive approach integrating economic development, environmental protection, social justice, and inclusion.

There are numerous definitions describing the idea of sustainable development. However, most of these definitions converge in defining it as a development approach that satisfies the current generation's needs while securing those of future generations. The concept of intergenerational equity is crucial to this idea. The needs of impoverished communities worldwide are emphasised as a priority. Additionally, this development approach acknowledges the restrictions that technology and social organisations impose in expanding the environment's ability to meet current and future requirements (UN, 1987).

More recent interpretations assign sustainable development with the primary goal of fostering a vibrant economy that provides full employment, high-quality education, and health

protection while respecting cultural diversity and guaranteeing social and environmental protection in a peaceful and secure world (OECD, 2012).

2.1.2 The Green Growth advent

In March 2005, representatives of 52 Asian and Pacific governments gathered at the Fifth Ministerial Conference on Environment and Development in Seoul. The primary focus of the conference was to identify synergies between environmental sustainability and economic growth. Participants also discussed strategies for moving away from unsustainable development patterns and towards environmentally sustainable economic growth. The notion of '*green growth*' emerged as a way to achieve sustainable development and the Millennium Development Goals, particularly those related to poverty reduction and environmental sustainability. Henceforth, policies should be geared to safeguard the natural environment and create the right incentives for realising an economy that is low carbon, resource efficient, and socially inclusive (UNESCAP, 2005). Following the conference, delegates approved the Seoul Initiative on Environmentally Sustainable Economic Growth (Green Growth) and a regional sustainable development implementation plan.

The pursuit of green growth gained momentum after the 2007-2008 global financial crisis. In 2009, OECD member countries signed the Green Growth Declaration, whereby the green growth approach was conceived as an opportunity to reformulate the production-consumption paradigm that prevailed till then and restore long-term growth and financial stability (OECD, 2009).

The OECD was then entrusted with developing the comprehensive framework and overarching strategy to implement the socio-economic, environmental, and technological elements underpinning the required green growth approach (OECD, 2009). As a result, the promotion of a green economy became more prevalent, supporting the realisation of green growth and green jobs and prioritising the efficient use of natural resources.

In 2010, 46 Asian and Pacific countries signed the 'Ministerial Declaration on Environment and Development in Asia and the Pacific'. The declaration – proposed by the United Nations Economic and Social Council for Asia and the Pacific (UNESCAP) – aimed to encourage and support international cooperation among developing countries to apply green growth tools and implement programs, actions, and policies that promote environmentally sustainable economies (UNESCAP, 2010b). Additionally, participating countries committed to coordinated regional activities included in the Astana Green Bridge Initiative (UNESCAP, 2010a).

The Global Green Growth Institute (GGGI) was established in response to the increasing concern for the environment. Later on, in 2012, the World Bank, UNEP, OECD and the GGGI jointly

launched the Green Growth Knowledge Platform (GGKP). The GGKP is an international platform designed to support and promote green growth and the green economy. Its primary objective is identifying and addressing significant knowledge gaps in green growth theory and practice. Furthermore, it helps countries to design and implement strategies that will help them move forward and towards a green economy.

The concept of green growth originally developed was pivotal to these strategies, as it aimed to ensure that natural resources continued to provide the resources and environmental services necessary for human well-being. In essence, green growth entails promoting economic growth and development while safeguarding natural heritage to provide the environmental resources and services that our well-being and that of future generations depend on (OECD, 2011, 2013, 2015).

Market-based policy measures were seen as effective means to achieve these objectives, with environmental taxes and trading schemes being used to green up the economy (Barbier, Markandya and Pearce, 2013; Pearce, Markandya and Barbier, 1989). Developing new, clean, and environmentally friendly technology was also seen as critical to reducing environmental impacts, with green products, markets, and jobs being suggested as ways to achieve further economic development without increasing environmental harm (Barbier, Markandya and Pearce, 2013; Hickel and Kallis, 2020).

2.2 Green Growth and Green Economy Theory

2.2.1 Funding principles of green growth and green economy

2.2.1.1 Green growth - definitions and interpretations

The dedicated literature provides multiple definitions of green growth and several interpretations of what it entails (for an overview, see UNDESA 2012a). While each definition differs, there are recurring elements describing the main concept behind green growth.

One of these elements concerns the role that environmental protection plays in fostering growth. For instance, the OECD defines green growth as an approach that brings about *‘economic growth and development while ensuring that natural assets continue to provide the resources and environmental services on which our well-being relies’* (OECD, 2011, p. 114). Similarly, UNESCAP perceives green growth as *‘a path to achieving economic growth and well-being while using fewer resources and [...] emissions’* (UNESCAP, 2012, 17). The World Bank (2012) describes it as *‘a growth that is efficient in its use of natural resources, clean in that it minimizes pollution and environmental impacts’* (World Bank, 2012, p. 2).

Other definitions emphasise the technological aspect. For instance, it highlights that it secures '*new growth engines through research and development of green technology*' (Government of the Republic of Korea, 2010 Art. 2).

Other elements relate instead to the aptness of green growth to promote inclusive and equal growth in developing countries. The ultimate objective is to have an '*Inclusive green growth...[that]... operationalize sustainable development by reconciling developing countries' urgent need for rapid growth and poverty alleviation*' (World Bank, 2012, p. 2) while also tackling inequality, resource distribution and exclusion (Bass *et al.*, 2016).

As indicated in section 2.1.2, green growth partly emerged as a response to the worldwide financial crisis of 2008. At that time, international organisations and national governments were seizing the opportunity to tackle economic recession and climate change simultaneously. They intended to achieve this objective through policies, tools and market mechanisms that favoured investments in economic activities bearing minimal environmental impact (OECD, 2013, 2015).

By using the definitions above, we can understand green growth as an approach that aims to maintain natural resources while promoting long-term economic growth that is fair and inclusive. Additionally, this approach encourages the development of innovative industries and low-carbon technologies to minimise environmental damage (Hussain *et al.*, 2021). Hence, green growth concepts, policies and actions aim to decouple economic growth from environmental impacts, promote technological progress and social equity, and address market failures.

2.2.1.2 Green economy - definitions and interpretations

One of the first mentions of a green economy was made in 1989 when Pearce, Markandya and Barbier (1989) anticipated that the economy needed to become greener to achieve sustainable development. However, only after the 2008-2009 financial crisis did the green economy idea gain global political and economic attention.

Despite this, there is no universally accepted definition, and the term is subject to various interpretations (UNDESA, 2012a). Scholars have investigated the blurry boundaries between green economy and green growth, noting that these two concepts are often used as synonyms, while they are not (Merino-Saum *et al.*, 2020). Green economy is a multidimensional concept that underscores the interdependence between the environment and the economy. As such, it has a higher potential than green growth to transform society, as it combines social issues and economic growth while considering the environmental finitude (Merino-Saum *et al.*, 2020). Some scholars posit that the green economy reflects a concept that varies depending on the context (Death, 2015) and relates

to an “umbrella” concept encompassing multiple implications for growth and natural resource use (Loiseau *et al.*, 2016). Others reconciled it with the circular economy and the bioeconomy ideals (D'Amato *et al.*, 2017) or with the instrumental role it plays in realising sustainable development and achieving the climate mitigation targets of the Paris Agreement (Georgeson, Maslin and Poessinouw, 2017; Loiseau *et al.*, 2016).

A frequently cited definition is that by UNEP, which describes the green economy as a growth approach that *‘results in improved human well-being and social equity, while significantly reducing environmental risks and ecological scarcity’* (UNEP, 2011, p. 2). The definition provided by UNEP implicitly refers to the instrumental capacity of ecosystem services and the natural environment to sustain human welfare.

Following this interpretation, a green economy can be seen as one driven by low-carbon technology, resource efficiency, environmental protection, and social equity. Further, the green economy suggests that ecosystem services and ecological processes can support society and the economy without hampering the sustainability of these ecosystems (D'Amato and Korhonen, 2021). This, and the associated poverty and social elements, make it particularly relevant for developing countries, especially after considering that one key structural feature of emerging economies is their dependence on natural resources and ecosystem services (Barbier, 2016; D'Amato and Korhonen, 2021).

Hence, the constituting elements of the green economy include green infrastructures, nature-based solutions and the accounting of ecosystem services (Chirieleison and Rizzi, 2023; D'Amato *et al.*, 2019). However, the green economy extends its reach by including ideas belonging to the Circular Economy concepts (for definitions, see Box 1), the reuse-reduce-recycle approach for product life cycle (D'Amato *et al.*, 2017).

To conclude, the green economy can be interpreted as a balanced approach merging all three dimensions of sustainable development, where additional emphasis is put on the social dimension. In addition, there is greater recognition that the green economy has to work within the planetary boundaries and ecological limits to ensure sustainable and resilient development against growing climate hazards and uncertainty (UNDESA, 2012b).

Box 1: Additional definitions of blue economy, nature based solutions and circular economy

Blue Economy

The concept of Blue economy largely aligns to that of green economy. The concept tries to mainstream the tools and policies of the green economy in the management of coastal resources, to ensure the protection of marine biodiversity and ecosystems. The latter, once again, combined with economic growth and livelihood protection for coastal populations (Eikeset *et al.*, 2018). The aspect of economic use and value of ocean resources is particularly relevant from SIDS. In fact, local populations heavily rely on ocean ecosystem services and resources to fetch market (e.g. tourism, aquaculture, marine biotechnology, renewable energy from winds and tide), and non-market benefits (i.e. protection from floods, storms, coastal erosion protection or carbon sequestration potential and from wind and tides (Acosta *et al.*, 2021; OECD, 2021a).

Nature-based solutions (NbS)

According to the OECD (2020) '*NbS are measures that protect, sustainably manage or restore nature, with the goal of maintaining or enhancing ecosystem services to address a variety of social, environmental and economic challenges*'. Since they seek to reduce pollution, safeguards the environment, and increase the stock of natural capital, NbS aligns perfectly with the principles of strong sustainability. These solutions are applicable in both urban, rural and marine environments and they represent useful instruments to implement the green economy strategy (Loiseau *et al.*, 2016). Solutions vary in nature, scope, and size depending on the context where they are applied. However, one common feature is that they provide simultaneous benefits; including carbon storage, raw materials, flood control and biodiversity protection.

Circular Economy

The circular economy concept aims to create sustainable production-consumption systems that minimize inputs and outputs. Within production processes, this is achieved by increasing material and energy efficiency, reducing unused waste and scrap material, and prolonging product usage throughout its life cycle (Geisendorf and Pietrulla, 2018). Circular economy solutions involve rethinking product/service design to enable efficiency gains, reducing the material and energy required for production, providing long-term maintenance and repair, fostering sharing, reuse, refurbishing, and remanufacturing, repurposing, recycling, and categorizing waste into inorganic and biological components. Additionally, renewable energy sources should be employed (Ellen MacArthur Foundation, 2015).

2.2.2 Relating green growth and green economy elements to sustainable development

The exact meaning of sustainable development is contested and can assume several interpretations (Akao, 2023; Williams and Millington, 2004). However, one of the most cited readings of its concept is provided in the Brundtland report (UN, 1987). There, *sustainable* development is conceived as a responsible process that uses resources wisely, makes strategic investments, advances technology, and implements institutional reforms to meet humanity's current and future needs (UN, 1987). The report recognised the challenges and limitations posed by existing technology and even suggested using renewable energy as one possible solution. In summary, sustainable development was considered the only viable solution to protect the environment and promote intergenerational equity. However, while the report identified current and future challenges, its solutions lacked practicality and specificity, failing to provide a clear roadmap for change. In addition, the Brundtland report (UN, 1987) emphasised the need for a balanced and equitable approach to growth that implicitly tries to achieve sustainability across all three dimensions of sustainable development: environmental, social, and economic.²

Starting from this premise, several researchers investigated the ontological positions of green growth, green economy and sustainable development (Kasztelan, 2017) and verified whether pursuing green transition would concurrently allow for sustainable development (Houssam *et al.*, 2023; Weick, 2016). However, discerning the potential for green growth and green economy to ensure sustainable growth requires prior defining the type of sustainable development being envisaged.

The concept of sustainable development is divided into 'weak' and 'strong', depending on whether natural capital and human-made capital are considered interchangeable (Hartwick, 1977; Solow, 1974). The weak sustainability concept posits that environmental capital is fully replaceable by manufactured capital and that society will be better off as long as the aggregate capital stock grows. The additional revenues generated by increasing capital returns could then be used to compensate future generations for the environmental damage created (Hartwick, 1978; Neumayer, 2013; Pearce and Atkinson, 1993; Pearce, Barbier and Markandya, 1991; Stern, 1997).

Differently, the strong sustainability concept sees natural capital as an irreplaceable element to human well-being for the life-supporting services it provides (Costanza and Daly, 1992). In that respect, manufactured capital cannot fully substitute natural capital for at least three reasons.

² Purvis, Mao and Robinson (2019) provide an interesting review on the conceptual origins of the tripartite (social, environmental and economic) sustainability concept.

Firstly, since human-made capital is made of natural capital, the former cannot constitute a perfect substitute for the latter (Ekins *et al.*, 2003). Secondly, because of the critical function and irreversible features of natural capital. While manufactured capital can be reproduced, and its destruction can be reversed, natural capital is irreparable when overexploited (De Groot *et al.*, 2003). Species extinction is one clear example of the irreversible trait of natural capital. In addition, natural ecosystems provide a wide range of life-supporting services (e.g. pollination) that exceed those that could be provided by manufactured capital. The lack of knowledge about the exact functioning of natural systems makes it difficult to predict the effects of destroying natural capital on human well-being and to find a technological substitute for it (De Groot *et al.*, 2000). Third, natural capital cannot be compensated by increased artificial capital stock in the future. Thus, preserving natural capital denotes an intergenerational equity issue and constitutes a moral and justice obligation of the current generation towards the future generation (Dedeurwaerdere, 2015).

In many respects, green growth and green economy principles are similar to those guiding the sustainable development agenda. Several authors conceived these approaches as means to operationalise sustainable development (UNEP, 2012) since they are more prescriptive in identifying possible routes to change (OECD, 2011; World Bank, 2012) or to decrease pressure on the environment (Loiseau *et al.*, 2016). Others see green growth as the only way to effectively reduce the burden of environmental degradation in the long term by mitigating the climate change effect. In his review, Nicholas Stern, N. H. (2007) concludes that mitigation benefits will be significantly higher than the cost of inaction and that ignoring climate change can lead to a substantial reduction in economic growth. More recently, Stern and Stiglitz (2023) argued that the increasing return to scale pushed by recent technological advances makes the green transition more convenient than ever. Thus, greening up the economy is desirable from both an environmental and economic perspective, as it will increase growth and well-being simultaneously. In addition, a takeaway conclusion from this study is that the supposed trade-off between economic growth and environmental health appears to subside. Nevertheless, not everyone agrees to see the green economy approach as a means to achieve sustainable development (D'Alessandro *et al.*, 2020; Kothari, Demaria and Acosta, 2014; Söderholm, 2020).

Certainly, green growth and a green economy cannot be considered direct substitutes for sustainable development but rather valuable complements to it. These approaches have a targeted focus, with a practical policy agenda to achieve a balanced economic and environmental relationship. The reconciliation between economic growth and environmental sustainability can be

obtained if this process is driven by nature-based solutions and ecological adaptation mechanisms and complemented by techno-knowledge solutions (D'Amato *et al.*, 2017). Thus, the emphasis is on creating the right conditions for innovation, investment, and competition and generating new economic growth while promoting resilient ecosystems.

However, whether they would also lead to sustainable development largely depends on the detailed actions and public policies set forth (Stern and Stiglitz, 2023). As indicated by Loiseau *et al.* (2016), green strategies vary significantly depending on the targeted economic sectors and the country's economic status. Some solutions align more closely with environmental economics (e.g. clean energy production through renewable energy sources) and fall within the weak sustainability paradigm.³ Other solutions (e.g. nature-based solutions or green infrastructure) are more in line with ecological economics and favour the strong sustainability narrative (D'Amato *et al.*, 2017; Loiseau *et al.*, 2016).⁴

Thus, a full reconciliation between sustainable development, green growth and green economy appears possible as far as technological and policy progress brings about a structural change in the economic growth paradigm. Kasztelan (2017) provides a reconciled vision of green growth, green economy, and sustainable development as an: *'economic growth which contributes to rational utilisation of natural capital, prevents and reduces pollution, and creates chances to improve the overall social welfare by building a green economy, and finally makes it possible to enter on the path towards sustainable development'*. (Kasztelan, 2017, p. 497)

The recent progress in reducing carbon emissions and increasing renewable energy sources (IEA, 2023) leads us to think that the green transition could ultimately trigger sustainable development. Meanwhile, Barua and Khataniar (2016) concluded that emerging Asian economies follow a weak-sustainability path, while high-income countries in the region are gradually shifting towards a strong-sustainability path – although they have not reached it yet. Understanding whether the ongoing green transition would enable net-zero emissions by 2050 largely depends upon the decoupling pace between economic growth and emissions.

³ Environmental economics studies environmental externalities, market failures and the optimal use of natural resources to find solutions to the natural environment decline. Scholars in this field largely agree that government can correct market failures and ensure the optimal use of scarce natural resources through incentive mechanism Haab and Whitehead (2017). This theory assumes that human-made capital and natural capital are perfectly substitutable, meaning that gains in one element can offset a loss in the other Pearce and Atkinson (1993).

⁴ Ecological economics is a theory that studies the relationship between natural ecosystems and economic systems. This theory is based on the strong sustainability concept, suggesting that natural and human-made capital are complementary but are not interchangeable. This means that gains in human-made capital cannot offset the loss of natural capital Røpke (2005).

2.2.3 Corollary elements of the green transition: Decoupling, Environmental Kuznets Curve and Porter Hypothesis

2.2.3.1 Decoupling

As mentioned in section 2.2.2, disentangling economic growth from environmental pressure is a prerequisite to reaching sustainable development. Whether countries could reach a weak or strong sustainability development following green transition principles highly depends on their capacity to innovate production systems and decouple economic growth from carbon emissions.

With reference to climate change, decoupling refers to the relationship between economic growth – normally measured with GDP – and carbon emission – normally measured as tons of CO₂ equivalents. Decoupling can be classified as ‘absolute’ (strong) or ‘relative’ (weak) based on the growth rates of the climate and economic variables. According to Tapio (2005), strong decoupling occurs when the GDP grows and emissions decrease, while weak decoupling occurs when GDP grows faster than emissions. Alternative growth rate combinations lead to either coupling or negative decoupling (more details are provided in chapter 4.1.2.8 and Table 6).

Decoupling has been extensively studied conceptually (Deutch, 2017; Turner and Katris, 2019; Wiedenhofer *et al.*, 2020), empirically (Haberl *et al.*, 2020), and at global (Hubacek *et al.*, 2021) national (Han, Liu and Liu, 2022) and subnational level (Cohen *et al.*, 2019; Luo *et al.*, 2017; Wang, Zhao and Li, 2019).

Evidence on decoupling is mixed and highly dependent – among other elements – upon countries’ income levels, policy stringency, and external factors. Recent studies have found that rich countries have shown improvements in decoupling in recent years (Haberl *et al.*, 2020; Le Quéré *et al.*, 2019; Vogel and Hickel, 2023). However, the absolute decoupling being detected is partly related to external concurrent factors that led to economic contraction and lower GDP growth rather than genuine systemic change (Le Quéré *et al.*, 2019). Others argue that emission reduction in high-income countries is partly due to import increase (Indriawati *et al.*, 2023) and that trade openness is actually detrimental to green growth in developing countries (Tawiah, Zakari and Adedoyin, 2021). This emission-trade relationship – known as ‘carbon leakage’ – occurs when high-income countries reduce their carbon balance not due to more stringent environmental regulations but because of increasing imports from emerging economies. In practice, the carbon emission issue is burdened on the exporter. Thus, the evidence suggests that absolute decoupling is an exception to the rule rather than the norm (Haberl *et al.*, 2020). In fact, this has been observed only briefly for certain countries, economic sectors, and locations with very low mitigation rates (Parrique *et al.*, 2019).

Instead, relative decoupling still prevails across high-income countries (*ibid.*). Moreover, the emission reduction and decoupling achieved in developed countries fall far short of the Paris Agreement targets (Vogel and Hickel, 2023). If these countries were to reduce emissions according to the 2013-2019 average trend, carbon neutrality would neither be reached by 2030 – as established – nor by 2050. To meet these targets, decoupling rates must increase by a factor of 12 by 2030, on average. Similar results from recent studies indicate that the huge increase in decoupling rates compliant with the Paris targets makes green growth and carbon neutrality very unlikely to occur even in highly advanced economies (Hickel and Kallis, 2020; Lenaerts, Tagliapietra and Wolff, 2022; Vogel and Hickel, 2023).

Certainly, several studies recognise that environmental and climate policies have partly contributed to disseminating low-carbon technology across sectors (Budina *et al.*, 2023; Hasna *et al.*, 2023), inducing an overall improvement in climate change mitigation (Nascimento and Höhne, 2023; van Soest *et al.*, 2021). Nevertheless, such policies have thus far contributed only to relative decoupling (Haberl *et al.*, 2020). In a recent study, the OECD (2021b) confirms the current technological innovation may have helped fetch the '*low-hanging fruits of energy savings*' (*Ibid.* p.9) but that higher targets of emission reductions would require substantial technology changes and resource reallocations.

From the above, absolute decoupling appears to lack empirical support but can still be considered technically possible (Hickel and Kallis, 2020). However, the world economy is unlikely to reach it soon enough, at a global scale, while staying within the planet's boundaries (Kallis *et al.*, 2018; Parrique *et al.*, 2019).

Against this background, several studies recommend more effective and stringent policies – based on and driven by absolute decoupling – to meet the Paris targets (D'Alessandro *et al.*, 2020; Haberl *et al.*, 2020; Le Quéré *et al.*, 2019). Exceeding these targets may shove Earth's ecosystem over several tipping points, leading to irreversible and severe changes (OECD, 2023, p. 3).

2.2.3.2 The Porter's hypothesis and the environmental Kuznets curve

Much of the credibility of the green transition discourse relies on the cogency of two additional elements: the Porter's hypothesis (PH) and the environmental Kuznets curve (EKC).

The PH posits that strict environmental regulations can stimulate the innovation and implementation of cleaner technologies, leading to more efficient production processes and products. Innovation would then bring about environmental improvements, while the cost savings – resulting from efficiency gains – would be enough to offset the costs of complying with new

regulations and the innovation cost (Porter, 1991; Porter and van der Linde, 1995). In other words, stricter environmental regulations and economic growth are fully compatible, and the former sustains the latter. The PH is further classified into three categories, depending on the results obtained through the more stringent environmental policy. A strong PH version suggests an increase in productivity, a weak version shows an increase in innovation, and a narrow version concerns improvements in both areas (Jaffe and Palmer, 1997).

The EKC theory suggests an inverted U-shaped relationship between income and greenhouse gas (GHG) emissions. At the start, an increase in income drives emissions to grow. However, beyond certain GDP levels, emissions begin to decrease as countries become more energy-efficient, technologically advanced, and more inclined and able to reduce GHG emissions (Grossman and Krueger, 1991; 1995). In summary, the EKC theory suggests that economic growth can be a solution to environmental issues rather than a threat.

Verifying the existence of the PH and the EKC has proved tricky, as the empirical literature yields discording results. In addition, research on PH in the Global South is particularly scant, leaving a knowledge gap in identifying how developing countries could catch up with more advanced economies in reducing emissions (Kyle, 2023). Meanwhile, two meta-analyses found heterogeneous results but finally confirmed the validity of the PH hypothesis (Cohen and Tubb, 2018; Li, Li and Gan, 2022). In European countries, the OECD (2021b) has found partial support for the strong version of the PH (i.e. on productivity) in the short-run, but it also validated the presence of the pollution haven hypothesis.⁵ Opposite, Benatti *et al.* (2023) reject the PH, indicating that more stringent environmental policies negatively affect the productivity growth of highly polluting countries – at least in the short run.

The empirical literature on the EKC was found to include significant flaws in the statistical methods and assumptions being used (Stern, 2004). Recent advances have investigated the presence of EKC in both developed (Awaworyi Churchill *et al.*, 2018; Ben Jebli, Ben Youssef and Ozturk, 2016) and developing countries (Alam *et al.*, 2016).

In Asia, the evidence behind the existence of EKC is mixed. For instance, regional studies have found empirical support for the presence of EKC in several Asian countries (Apergis and Ozturk, 2015; Wijaya Tjoek and Wu, 2018). Opposite, country case studies in Cambodia have disproved the

⁵ The Pollution Haven Hypothesis (McGuire (1982) posits that firms will relocate production in countries where environmental policies are less stringent to maintain their competitiveness. The relocation of production plants would also shift the emissions to other regions leading to a 'carbon-leakage'.

validity of the EKC (Ozturk and Al-mulali, 2015), while in India, the EKC has been both verified (Rani and Kumar, 2019) and rejected (Alam *et al.*, 2016; Yang and Zhao, 2014).

Large global analyses also provide mixed results. In their study on 213 countries, Aşici (2013) found that the EKC relationship does not hold and that income growth increases pressure on the environment. Results vary depending on the country's development stage, with larger effects measured from low- and middle-income countries. In addition, trade openness negatively affects environmental protection, supporting the idea that countries adopt looser environmental policies to maintain their competitiveness in the international market.⁶ Foreign direct investments (FDI) are another critical factor in this relationship (Li *et al.*, 2022). A more recent meta-analysis of 101 studies strongly supports the existence of the EKC relationship, particularly in the long run (Saqib and Benhmad, 2021). Other mixed results show the presence of inverted U-shaped, N-shaped and inverted N-shaped relationships across countries (Allard *et al.*, 2018; Awaworyi Churchill *et al.*, 2018).

2.2.4 Critical views on the green transition and the post-growth argument

Over the years, green transition principles have gathered major political support and several criticisms. A recent study indicates that environmental economists, applied scientists, and engineers are optimistic in believing that green growth is achievable thanks to technological progress. However, there is growing scepticism among European scholars concerning this technological optimism (King, Savin and Drews, 2023). Some of these critics disparage the idea that market transformation and technological progress could adequately safeguard environmental resources and ensure a sustainable development path (Huesemann, 2003), particularly so within a global economy that has already exceeded the planetary boundaries (Butler, 2017; Raworth, 2017b; Rockström *et al.*, 2009; Steffen *et al.*, 2015).

One of the central criticisms of the green transition concerns the idea that technological innovation can increase energy efficiency and reduce carbon intensity across industries sufficiently enough to cut emissions. As we have seen in section 2.2.3.1, the general optimism about the capacity of technology to advance the transition to carbon neutrality is not backstopped by compelling evidence of this happening anywhere. Less so in developing countries, where governance and implementation issues, partly due to inadequate human, financial and technical resources, hinder governments' capacity to sustain the transition to low-carbon economies (Barbier,

⁶ This is known as the race-to-the-bottom hypothesis (Daly (1993)).

2016; Capozza, I. and Samson, R., 2019). Even when succeeding in undergoing structural reforms and migrating from an agricultural- to a service-based economy, developing countries are unable to deliver green growth (Baniya, Giurco and Kelly, 2021).

Another argument is that technological progress in itself may not be enough. This happens, for instance, when technological solutions to one environmental issue create new problems or shift the pressure elsewhere (e.g. electric vehicles vs minerals; biofuels vs land use change; nuclear power energy vs nuclear waste disposal) (Parrique *et al.*, 2019).

Otherwise, because of a 'rebound effect', according to which efficiency gains can lead to a higher and cheaper supply of technology and energy, which, in turn, decreases market prices. From there, the more favourable market conditions may increase aggregate demand and increase production and consumption emissions (Gunderson and Yun, 2017; Lenaerts, Tagliapietra and Wolff, 2022; Parrique *et al.*, 2019).⁷ Subsequently, the money saved through the efficiency gains provided by renewable energy and improved technology may totally or partly be offset by increased consumption and resource use. The degree of the offset depends on whether the incremental savings are spent on energy-efficient or polluting activities (Parrique *et al.*, 2019).

Efficiency gains can be maximised only if accompanied by reduced economic activity and increased pursuit of sufficiency (Parrique *et al.*, 2019).⁸ Policies promoting sufficiency may include restrictions on using fossil fuels-based technologies and redistribution of wealth, work and natural resources (Haberl *et al.*, 2020). Ultimately, it seems reasonable to presume that the necessary reductions to reach carbon neutrality will be met faster in a world without growth than with growth (Hickel and Kallis, 2020; Parrique *et al.*, 2019).

Other critics argue that green growth fails to address the main source behind environmental overexploitation, and it represents the rebranding of a traditional approach rather than a radical systemic change (Brand, 2012; Kallis, 2011). Especially in the global south, there is a need to divert from the highly inequitable and ecologically damaging growth patterns, which justify big infrastructural projects by disputing the environmental debate. Instead, greener economies can only be obtained via more radical changes to how we produce, consume and trade rather than by obsessively defending the growth *status quo* (Death, 2015).

⁷ This phenomenon is also known as the 'Jevons Paradox' and the two terms are often used as synonyms although they differ. The Jevons paradox refers to efficiency gains and resource use on the production side while the rebound effects affects the consumption side (York and McGee (2016).

⁸ Sufficiency intended as moderation and frugality in resource consumption that does not exceed what is strictly required to satisfy one's needs.

From there, it seems that the green growth agenda is politically motivated rather than driven by scientific evidence. Hickel and Kallis (2020) argue that the global community is forcing green growth through because it is politically unacceptable to question economic growth and also because no nation would voluntarily limit growth. Hence, green growth is pushed by political convenience rather than scientific evidence. Instead, *'we should assess the facts and then draw conclusions, rather than start with palatable conclusions and ignore inconvenient facts'* (Hickel and Kallis, 2020, p. 483).

Against this backdrop, the critics of green growth gave rise to the post-growth theory founded on two alternative development approaches, namely degrowth and agrowth (King, Savin and Drews, 2023). The degrowth agenda pivots around the need to downsize consumption and production footprints instead of sustaining economic growth in hopes of future environmental efficiency gains (Georgescu-Roegen, 1975; Latouche, 2009). The global community should pursue social and environmental justice rather than economic growth (King, Savin and Drews, 2023). According to degrowth supporters, holding up economic growth is the only way to curb emissions. Particularly so, as the attempts to reduce the reliance on fossil fuels, promote the use of renewable energy sources and implement carbon-credit schemes are not helping to mitigate climate change effects (D'Alessandro *et al.*, 2020; Lux, Fromont and Le Vo, 2023; van den Bergh and Savin, 2021).

Moreover, the perspective provided by the green growth approach is that of limiting the impact of economic growth on the environment but not that of addressing its root cause. As shown by several authors, the efforts made thus far are insufficient to reach carbon neutrality targets in the near future (Haberl *et al.*, 2020; Hickel and Kallis, 2020; Vogel and Hickel, 2023). Ultimately, meeting these targets requires a much more radical and transformative revolution of our daily practices and social institutions (Paterson, 2021).

Another alternative to the green growth approach is the 'agrowth' concept, which emphasises the need to abandon the growth concept altogether (van den Bergh, 2011; van den Bergh and Kallis, 2012). According to King, Savin and Drews (2023), the agrowth position focuses more on societal and environmental well-being and is agnostic towards GDP growth. For that, its concept stands mid-way between the degrowth and green growth principles.

2.2.5 Counterarguments in favour of a green transition

Although the de-growth and agrowth concepts have attracted increasing attention in recent years, they failed to gain traction and political support in part due to a lack of a clear strategy on how to restructure the global economy and reshape the way society works. The lack of major

political backing largely relates to the established interests and power relations prevailing in developed societies (Hickel and Kallis, 2020; Kallis *et al.*, 2018).

Concurrently, it is hard to imagine how developing countries – some of whom are starting to enjoy improvements in well-being and living standards and desperately need to escape poverty – would accept undergoing major reforms following a degrowth plan which lacks convincing details of how the poorest can achieve improved life quality (Lenaerts, Tagliapietra and Wolff, 2022; Lipsey, 2019; Vazquez-Brust and Plaza-Úbeda, 2021). Several critics also point to the redistributive mechanism envisaged to balance social equity between and within countries. Such measures imply income convergence between rich and poor countries to an income level that is nonetheless hard to define (Lenaerts, Tagliapietra and Wolff, 2022).

Another counterargument concerns the discovery of substantial rebound effects in energy savings strategies. Sorrell, Gatersleben and Druckman (2020) posit that sufficiency actions have little influence on the final energy use and that rebound effects prevail and erode a large part of the energy and emission savings.

At the same time, economic growth, innovations and sustainable technological transfer are found to affect positively green growth, especially in developed countries (Fernandes *et al.*, 2021; Hussain *et al.*, 2021; Tawiah, Zakari and Adedoyin, 2021). Thus, it seems possible to achieve economic growth and sustainable green development by promoting the adoption and dissemination of green technology. Also, while economic growth is a major source of resource overexploitation and higher carbon emissions (Ben Amara and Qiao, 2023), it is also a major driver of the technological revolution that we need to improve our well-being and reduce our consumption footprint. It is hard to imagine how we could backstop progress without growth.

Beyond the technological quest, even if we were to halt economic growth tomorrow, we would still need to implement green growth measures to eliminate highly polluting technologies, so no growth could exist without green growth (Lipsey, 2019). On top of it, an economic halt could initiate a downward spiral affecting debt sustainability, social welfare and the financing of public health systems in several developing and developed economies (Lenaerts, Tagliapietra and Wolff, 2022).

Finally, according to Stern and Stiglitz (2023), the no-growth movement is diverting attention from the central issue: finding a sustainable solution to the environmental destruction caused by our current production and consumption patterns. This issue can be solved by changing how we produce and consume and shifting our consumption habits towards more environmentally friendly

goods and services. Fortunately, some of the required technologies to do so are already available, and this trend is gaining momentum.

To summarise, we can notice that while the degrowth agenda is certainly valid and desirable from various standpoints, it still lacks a pragmatic vision of how to implement it. It seems that green growth remains the dominant solution for environmental sustainability, while degrowth remains a marginal approach (Fernandes *et al.*, 2021, p. 2031). Following Lipsey (2019), the idea of pushing the green growth agenda forth until it fails to meet its targets seems reasonable. If emissions are not declining fast enough, then additional, more restrictive measures could be taken, including a halt to growth. Yet, it remains to be seen how we could define the exact time when a change in strategy should be taken. After all, we must avoid that later in time becomes too late.

3 Research objectives and design

This chapter briefly overviews the research approach used in developing this thesis. Chapter 3.1 offers a glimpse at the research approach, describing the overarching method adopted in choosing and developing the case studies. Chapter 3.2 briefly describes the analytical method and calculation procedures used in the data analysis. Instead, more details on the methodological aspects are provided under each case study, including explicit referencing to data sources and validation.

3.1 Research approach

This thesis delves into its objective through the analysis of three case studies. While each study examines different development options in different country contexts, they all relate to green growth policies and practices in developing countries of the Asia and Pacific region. The results of each case study are used to answer research questions and achieve the thesis objective (see chapter 1.3). It is important to note that these research findings cannot be generalised to address the primary concern around green growth in developing countries. Nevertheless, they bring results and contribute to the ongoing climate change and carbon neutrality discussion by highlighting the progress and issues faced by emerging economies in tackling the climate emergency. To ensure that the research is aligned with these concepts and findings, the case studies were conducted by examining pertinent literature, and their conclusions were discussed in the context of previous research in the respective field.

This research has been established on a solid foundation, which ensures that it is grounded in the current knowledge and understanding of the field. This has been ensured by carefully examining the existing research body in the field. The ultimate objective is to contribute to the current understanding of the topic and advance field knowledge meaningfully.

3.2 Data collection and analysis methods

The three case studies are analysed employing different quantitative methods. Each technique has been carefully chosen according to the scope of the analysis. Similarly, the data type helped define which was the most appropriate that could help address the research questions. The data collection method also varied depending on the analysis method being chosen. The following paragraphs offer an overview of the data collection and analysis methods of each case study. More details are presented in the case study method sections (see chapters 4.1.2, 4.2.4 and 4.3.3)

Social accounting matrix with environmental accounts

The analysis of the first case study employed a social accounting matrix that was extended with environmental accounts (SAMEA) and life-cycle impact indicators. The method was selected because it allowed us to monitor the economic and environmental consequences of Cambodia's carbon neutrality strategy (LTS4CN). The policy modelling approach used in the first paper is particularly innovative as it simultaneously captures the impact on the biosphere and the economic system. The model monitors changes in over 15 pollutants, five natural resources and five environmental themes. In addition, this is the first time that the SAMEA has been applied in Cambodia, and it is used to measure decoupling progress.

Econometrics

Ordinary least squares and Tobit regression methods were used in the first paper to determine household income elasticities by quintile for food, non-food and total expenditure using data from the Cambodia living standards measurement survey (NIS, 2021). The elasticity values were then used to adjust the household consumption behaviour in the SAMEA model.

Decoupling

Decoupling indicators are calculated according to the methodology proposed by OECD (2002) and Tapio (2005). In particular, we calculated sectors decoupling factors and elasticities to verify decoupling trends over time. Calculating decoupling elasticities allowed us to determine the

strength and direction of the decoupling as well as the prevailing decoupling scenario (strong vs weak).

Environmental cost-benefit analysis

The second paper employs a CBA to evaluate the ecosystem services of the marine and coastal ecosystems. The values of each ecosystem – and the costs of restoring them – were calculated thanks to a meta-review of studies and by applying a benefit transfer method. The analysis includes benefits from direct, indirect and optional sources. Unit values of ecosystem benefits from one study site were transferred to the policy site after adjusting for differences in inflation and living standards across countries. Values were adjusted for changes in the Consumer Price Index (CPI) that occurred between the estimation period and the latest year for which CPI values are available. The analysis validates the value of each ecosystem in delivering.

Monte Carlo simulation

The results of the second paper were tested for uncertainty and variability through the Monte Carlo method. After removing outliers from the cost sample, probability distributions were fitted to cost variables. Cost variables were related to log-normal distribution or triangular distribution functions. Instead, revenues were fitted with normal or log-normal distribution functions. Profitability results were tested against 10,000 randomised trials and presented for various discount rate scenarios (i.e. 2, 5 and 10 percent).

Life-Cycle Assessment

The third paper employs a Life Cycle Assessment (LCA) to evaluate the environmental impact of Municipal Solid Waste practices in New Delhi - India. The method is used to run six scenarios. The first two business-as-usual (BAU) scenarios forecast waste generation in the city after accounting for population growth projections for 2030 and 2050. Two alternative scenarios consider a change in management practices, where compostable MSW is diverted from landfills and waste to energy plants towards compost plants by varying proportions. Finally, the other two scenarios combine the previous four scenarios and quantify impact reductions by 2030 and 2050 resulting from a change in MSW management between landfills, waste-to-energy plants, and composting.

4 Case studies

4.1 Paper 1: Carbon Neutrality and Green Growth in Cambodia: Quantitative Policy Modelling of Economic and Environmental Impacts

Abstract: Addressing climate change and biodiversity loss highlights the need to incorporate environmental considerations into policy and economic decision-making. Our research focuses on Cambodia, where we aim to enhance policy planning by accounting for greenhouse gas emissions and natural resource depletion associated with economic growth. We propose an integrated analytical framework to evaluate the country's carbon reduction strategy in the forestry sector. To achieve this, we computed the 'Social Accounting Matrix including Environmental Accounts' (SAMEA) for Cambodia, consisting of 94 accounts that track monetary transactions and physical changes in emissions, waste, and natural resources. We integrated the physical and financial flows into life cycle impact assessment (LCIA) indicators and environmental themes to track the effect on human health. We demonstrate that green growth policies sustain economic growth but fail to meet 2050 Paris targets and the emission reduction path envisioned in the forestry sector. We posit that bolder actions are needed to reduce emissions consistently with the long-term carbon neutrality strategy. This research also bridges the gap in environmental statistics and supports informed decision-making for a more sustainable future by demonstrating the feasibility of using quantitative tools in data-limited contexts to support sustainable development and policy-making.

4.1.1 Introduction

Policy and project evaluations are normally carried out using analytical methods whose examination spectrum is restricted within the economic domain. The desirability and aptness of a change in the *status quo* are often assessed by looking exclusively at income generation potentials or financial indicators. Nevertheless, the increasing evidence on climate change and biodiversity loss has contributed to raising awareness of the need to integrate economic decision-making with environmental accounting. Nowadays, national statistics are more closely integrating environmental and economic accounts to help policy-makers monitor interactions between the environment and the economy (United Nations, 1993). Strategic planning and decision-making processes are improved by making allowances for the accounting of gas emissions and the depletion of natural resources, which are linked to economic growth. Meanwhile, using such statistics in the evaluation process allows for gathering full evidence of economic choices – and their consequences – accounting for the private and social costs of the use of natural resources and the degradation of the environment.

In our study, we present an integrated analytical framework combining Cambodia's economic and environmental accounts. This framework represents a hybrid between the 'national Accounting matrix including environmental accounts' (NAMEA) developed by Haan, Keuning and Bosch (1993) and the 'social accounting matrix' (SAM) developed by Pyatt and Round (1985). In the technical literature, this hybrid takes various names, among which: a) 'social accounting matrix including environmental accounts' (SAMEA), and b) 'environmentally extended social accounting matrix' (ESAM). Both names indicate an accounting framework where national economic accounts (expressed in monetary units) and environmental accounts (expressed in physical units) are integrated. Our study will use the name 'SAMEA' to describe our tool. The ground-breaking aspect of our research lies in the unprecedented combination of the SAMEA with the Life Cycle Impact Assessment (LCIA). By integrating these two quantitative tools, we comprehensively analyse the economic, environmental and human impacts associated with carbon neutrality and green growth policies in Cambodia. To our knowledge, this integration has not been explored in previous studies.

The objective of this study is three-fold. Primarily, we use the SAMEA to analyse the economic and environmental impacts associated with carbon neutrality and green growth policies in Cambodia. Second, we showcase that integrated quantitative analysis can effectively support decision-making even in countries with limited data availability, thereby contributing to sustainable development efforts. Finally, environmental statistics on Cambodia are limited, and there is no

published work on using the SAMEA in the country, less so an integrated SAMEA-LCIA analysis; hence, we aim to fill this two-pronged gap through this study.

Following this introductory section, the rest of this chapter is organised as follows. Section 4.1.2 describes the methodological framework of our SAMEA, starting with a comprehensive description of the SAM structure, data sources, and estimation procedure. Section 4.1.3 provides the results of the quantitative analyses and policy simulations. Finally, section 4.1.4 gives some concluding remarks on this study.

4.1.2 Material and Methods

4.1.2.1 The Social Accounting Matrix

The SAM methodology was first developed by Sir Richard Stone (1948, 1955, 1962) and later refined by the seminal works of Pyatt and Thorbecke (1976), Pyatt and Roe (1977) and Pyatt and Round (1985). This methodology is considered an extension of the traditional Input-Output (I/O) model proposed by Leontief (1936) and consists of a matrix that reports the monetary exchanges that occurred within an economic system during a period, usually a year. The matrix allows us to investigate and analyse the transactions and relations occurring within and between economic sectors during the production, distribution, utilisation and income accumulation processes.

The connection across subsystems of the SAM allows tracking monetary flows occurring between sectors and measuring the structural changes within the economy (i.e. injections and multiplying effects in the system) resulting from a policy change or a project intervention.

The information is compiled in a double-entry table (i.e. the matrix), describing the structure of the economic system through key blocks (i.e. actors, productive factors and activities), assumed as the origin and destination of transaction flows. Each key block is disaggregated further into accounts headed to the institutional sectors (e.g. type of households, specific commodities and productive sectors) depending on data availability. In a typical SAM structure, columns represent the outflows of the different economic agents, that is, the expenditure of any aggregate with respect to the others, while rows represent the inflows, namely the income formation. A simplified scheme of the SAM is presented in Figure 3. The intersections between rows and columns define the set of macroeconomic variables.

Figure 3: A simplified SAM scheme

	Productive Factors	Household	Government	Production sectors and Commodities	Savings and investment	Rest of the World	Total inflows
Productive Factors		Domestic Employment	Government Employment	Value-added		Payments from abroad	Total factor income
Household	Labour incomes and profits	Inter-household transfers	Social transfers			Foreign remittances	Total household income
Government	Taxes on labour and profits	Direct taxes		Indirect taxes	Taxes from capital account	Foreign grants and loans	Total Government income
Production sectors and Commodities	Domestic supply	Private consumption	Recurrent spending	I/O Matrix (intermediate demand)	Investment and stock	Export payments	Total demand and activity income
Saving and Investment		Private savings	Fiscal surplus			Current account balance	Total savings
Rest of the World	Factors payments abroad	Household transfer	Government transfers	Imports payment			Total imports
Total outflows	Total factors spending	Total Household expenditure	Total Government Expenditure	Total Gross output	Total investment spending	Total Export	

Source: Ferrarese and Mazzoli (2018)

4.1.2.2 Social accounting matrix including environmental accounts

To analyse policy impacts on both the economy and the environment, the SAM framework has been expanded to include separate accounts for natural resources and pollutants. The approach is similar to that originally implemented by the Netherlands statistical bureau (Boo *et al.*, 1993; Haan and Keuning, 1996; Haan, Keuning and Bosch, 1993) in creating a national accounting matrix including environmental accounts (NAMEA). However, integrating environmental accounts into a SAM allows us to combine social accounts (e.g. households) with environmental impact resulting from economic activity (e.g. households' consumption).

This environmentally-extended SAM (SAMEA) enables quantifying pollution and the consequences of containment actions. The additional accounts connect the economic system to natural resources and enable an integrated reading of the economic and environmental phenomena.

The SAMEA estimated in this work has a structure based on the criteria formulated by the System of Integrated Environmental and Economic Accounts (United Nations *et al.*, 2014) and the

System of National Accounts (SNA 2008) (United Nations *et al.*, 2009). However, the entries in the environmental accounts are accounted in physical units – and not in monetary terms – to avoid critical aspects of environmental goods and services pricing (Christmas, 2017; Shabman and Stephenson, 2000). For this reason, we could not fully reconcile these accounts with the SNA structure to balance the totals, and they were treated separately as satellite accounts.

The environmental section of the SAMEA contains two flow matrices expressed in physical units. By rows, the first matrix shows the flows of natural resources that the production system uses as inputs (e.g. water resource abstractions) or those resources reabsorbed in the system (e.g. waste that is collected and processed). By columns, the second flow matrix shows the flow reabsorbed by nature (e.g. water discharge) after its use for production and consumption purposes, as well as the related pollutants emissions. A simplified scheme of the SAMEA is presented in Table 4.

The accounts that populate the environmental dimension are a) the main pollutants responsible for the GHG emissions; b) waste produced or absorbed in the economic system because of the production activity; c) natural resources depleted or discovered (e.g. biomass, fossil fuels), and d) environmental themes as well as life cycle impact assessment indicators (e.g. GHG effect, ozone depletion).

4.1.2.3 The GHG emission accounts

In columns, this account presents the origin of pollutants by consumers and producers and transboundary pollution from the rest of the world (RoW). The three main sources of emissions are production, consumption and international imports. The export of emissions appears in the intersection between GHG emissions and the RoW account. By rows, it shows the absorption of pollutants into the economic processes. The list of pollutants included in our study is provided in Table 1.

Table 1: List of pollutants tracked in the environmental accounts

Aggregated Substance	
Carbon dioxide (CO ₂)	Methane (CH ₄)
Nitrous Oxide (N ₂ O)	F-gases (HFC)
Disaggregated substances	
Air pollutants	
Ammonia (NH ₃)	Black Carbon (BC)
Carbon monoxide (CO)	Nitrous oxides (NO and NO ₂), expressed as NO ₂ (NO _x)
Non-methane volatile organic compound (NMVOC)	Organic Carbon (OC)
Particulate Matter 10 (PM ₁₀)	Particulate Matter 2.5 (PM _{2.5})
Sulphur dioxide (SO ₂)	
Toxic pollutants	
Divalent Mercury (Hg _D)	Gaseous elemental mercury (Hg _G)
Particulate associated Mercury (Hg _P)	Total Mercury (Hg)

Source: Own elaboration

4.1.2.4 Waste account

Waste, produced or absorbed as a result of the production and consumption activities, is presented in a separate account. By column, the account shows waste emission, while by row, it shows waste absorption into the economic processes (i.e. recycling).

4.1.2.5 Natural resources account

A dedicated account presents the loss and regeneration of natural resources, such as biomass, minerals, fossil fuels, water and forests. The columns show the addition to stocks – linked to the discovery or regeneration of natural capital – while the rows present the use of natural resources by economic activities.

4.1.2.6 Environmental themes and impact indicators

These accounts refer to environmental impact indicators at global and national scales. Global-level indicators include the GHG effect caused by emissions and ozone formation, indicating how Cambodia contributes to global warming.

The Kyoto Protocol (1997) established that six gases – CO₂, CH₄, N₂O, sulphur hexafluoride (SF₆), hydrofluorocarbon (HFCs) and perfluorocarbons (PFCs) – contribute to global warming. In our dataset, gases with fluorine compounds (i.e. SF₆, HFCs and PFCs) were lumped together as fluorinated gases (F-gases). Thus, our calculation of the global warming effect considered CO₂, CH₄,

and N₂O and F-gases. Also, since we lacked disaggregated emission data of F-gases, we could not estimate the ozone depletion potential.

The contribution of each gas to global warming is based on the global warming potential (GWP), over 100 years, that each gas withholds. The estimated gas emission is multiplied by the corresponding weighting factor (i.e. GWP) and expressed in CO₂ equivalence. The GWP factors used in this analysis are those established by the sixth assessment report (AR6) of the IPCC (2021) and are summarised in Table 2.

Table 2: IPCC AR6 global warming potentials

Greenhouse gas	GWP over 100 years
CO ₂	1
CH ₄ (fossil origin)	29.8
CH ₄ (non-fossil origin)	27.2
N ₂ O	273

Source: adapted from IPCC, 2021

Gas emissions contribute to increased temperatures, hydrological changes, higher frequency of extreme events and an overall change in climate. This modifies natural ecosystem structure, function, and phenology and increases plant and animal species losses. Similarly, climate change affects human health through an increase in mortality due to chronic and infectious diseases and the morbidity of cardiovascular and respiratory illness (IPCC, 2022a). Against this background, we estimated GHG emission impact on human health and ecosystems, disaggregating the impact on terrestrial, marine and freshwater ecosystems, where possible.

Ozone is formed in a photochemical reaction between volatile organic compounds and nitrogen oxides (i.e. NO₂, NO_x, NO). Higher exposure to ozone can lead to cardiovascular and respiratory diseases, impacting human health and increasing mortality (Zhang, Wei and Fang, 2019). Similarly, the excess presence of ozone affects plant and soil generative processes and can threaten key ecosystem functions (Agathokleous *et al.*, 2020). Thus, we estimated the impact on human health and natural ecosystem quality by calculating the potential for ozone formation.

The acidification effect is determined by aggregating the emission of acidifying gases (i.e. NO_x, SO₂ and NH₃). Changes in soil acidity level affect soil properties and nutrient availability and modify biota distribution, richness and presence (Yadav *et al.*, 2020). Meanwhile, increasing CO₂ emissions are absorbed into ocean water and contribute to water acidification and changes to the marine ecosystem (IPCC, 2022b). Other CO₂ precursors (i.e. CO, CH₄) and acidifying substances (i.e.

NO_x and SO₂) also contribute to lowering pH levels by releasing hydrogen ions in water (Bach *et al.*, 2016).

Exposure to increasing levels of fine particulate matter harms human health, causing more than four million premature deaths globally every year (Shiraiwa *et al.*, 2017). We estimated human health damage by accounting for fine-dust formation due to PM_{2.5} emissions.

The waste indicators estimate the total amount of polluting waste generated by all economic activities during the referenced period. Finally, the indicator on the use or loss of natural capital reports depletion of biomass, mineral ores, fossil, water and forest resources.

Table 3 summarises the substances contributing to the measurement of each indicator. Emission volumes are expressed in megatonnes (Mt, 10⁶ tonnes) equivalences of a defined gas. Damage to human health is reported in disability-adjusted life years (DALY), indicating the number of years lost due to disability or premature death from illness. Damage to the environment and biodiversity is shown either as a potentially disappeared fraction (PDF) or a potentially affected fraction (PAF). The PDF reports the proportion of species that may be lost during a period (a year in our study) due to external pressures – pollutant emissions in this case. The PAF refers to the risk of ecotoxicology due to exposure to toxic pollutants, and it is measured as the fraction of species that are exposed beyond an acceptable threshold. The supplementary material section provides the values of endpoint characterisation factors used to convert emission data into impact indicators.

Table 3: Summary table on substance contribution to environmental themes and impacts

Environmental theme	Impact	Substances	Unit
GHG effect	Global warming	CO2, CH4, N2O, F-gases	Mt CO2 eq.
	Damage to human health	CO2, CH4, N2O, F-gases	DALY
	Damage to ecosystems	CO2, CH4, N2O, F-gases	PDF
	Freshwater biodiversity loss	CO2, CH4, N2O, F-gases, NOx, Hg_G, Hg_D, Hg_P, Hg	PDF year kg ⁻¹
	Damage to the terrestrial and freshwater ecosystems	CO2, CH4, N2O, F-gases	PAF year ⁻¹
	Damage to marine ecosystems	CO2, CH4, N2O, F-gases	PAF
	Damage to global ecosystems	CO2, CH4, N2O, F-gases	PAF
	Biodiversity loss	Based on sector value added	PDF year ⁻¹
Ozone formation	Damage to human health	NOx, NMVOC	Mt NOx eq.
	Damage to ecosystems	NOx, NMVOC	Mt NOx eq.
Acidification	Terrestrial acidification	NOx, SO2, NH3	Mt SO2 eq.
	Ocean acidification	CO2, CH4, CO, NOx, SO2	Mt CO2 eq.
Fine dust emission	Damage to human health	PM2.5	Mt PM2.5 eq.
Waste	Resource use	Waste	Tonnes
Use/loss of natural resources		Biomass, minerals, fossil, forest	Tonnes
		Water	Million m3

Source: Own elaboration

Table 4: A simplified scheme of the SAMEA

	Productive Factors	Household	Government	Production sectors and Commodities	Savings and investment	Rest of the word (ROW)	GHG emissions	Waste	Natural resources	Environmental themes
Productive Factors	SAM								Discovery and renewal of resources	
Household							Emission from consumption	Waste from consumption		
Government							Emission from production	Waste from production		
Production sectors and Commodities										
Saving and Investment							Transboundary emission from ROW			
Rest of the World (ROW)										
GHG emissions		Absorption from consumption		Absorption from production		Transboundary emission to ROW				Accumulation of pollutants
Waste				Waste absorbed						
Natural resources	Extraction and depletion						Emissions from land-use change			Net reduction / depletion
Environmental themes	GHG inventory				Environmental indicators					

Source: Own elaboration

4.1.2.7 Data sources and estimation of parameters

The 2022 SAMEA for Cambodia was built by compiling data from several sources. The primary base for the estimation was the Cambodia Input-Output table of 2018, sourced from the OECD statistical database. The I/O table was updated with the most recent statistics from the World Bank (WB) Country Economy Data, Cambodia National Statistics Office and Asian Development Banks Statistics (ADB). In particular, we used economic data from the World Bank database on GDP and value added by the economic sector and from the ADB for a disaggregation of the economic sector. Historical data from households' consumption and population collected by the National Institute of Statistics (NIS) were also used. We applied the RAS method to balance the SAMEA (Bacharach, 1971). Table 5 summarises data and information sources in our model.

Table 5: Details and sources of the Cambodia SAMEA data

Data	Variables	Source
<i>SAM construction and accounts</i>		
Input-Output framework	Supply and Use tables or the Symmetrical table, fully reconciled with the main macroeconomic accounts based on the Cambodia OECD I/O table	(OECD, 2022)
National Accounts (aggregated and sorted by institutional unit)	These integrated accounts show, by institutional sectors, the flow of payments to and from institutions as double-entry accounts, from production to the primary and secondary income distribution. The ADB key indicator database and the WB world development indicators database were used to populate the accounts.	(ADB, 2021; World Bank, 2023)
Consumption patterns, household expenditure and income.	Income distribution, expenditure patterns, and income elasticities from Cambodian households are disaggregated by living setting (urban vs rural) and income quintiles.	(NIS, 2015, 2020, 2021)
<i>Environmental accounts</i>		
GHG emission	GHG emissions include CO ₂ (fossil only), CH ₄ , N ₂ O and F-gases. They are aggregated using Global Warming Potential values from IPCC AR5 (GWP-100 AR5) and were updated to AR6-100 factors. Emissions are provided for the following	Emissions Database for Global Atmospheric Research EDGARv8.0 (Crippa <i>et al.</i> , 2023)

	economic sectors: Power Industry, Buildings, Transport, Agriculture, Fuel exploitation, Processes, and Waste.	
Other gases, aerosols and toxic pollutants	Data on emissions of ozone precursor gases (CO, NOx, NMVOC), acidifying gases (NH3, NOx, SO2), fine Particulate Matter (PM10 and PM2.5) and Carbonaceous speciation (BC, OC)	EDGARv6.1 air pollutant dataset (European Commission, 2022)
Material	Data related to material flow extraction domestic use, import and export, including biomass (i.e. crops, crop residues, grazed biomass and fodder crops, wood, wild catch), metal ores (i.e. ferrous and non-ferrous), non-metallic minerals (i.e. construction dominant and industrial or agricultural dominant) and fossil fuel (i.e. coal, oil shares and tar sands, natural gas, petroleum).	Global Material Flows Database of the United Nations Environment Programme and the International Resource Panel (UNEP IRP, 2022)
Water	Data on water use, availability and extraction	FAO (2022a)
Forestry	Data on forest extension, loss, and composition, as well as CO2eq gross emission and removal	Global Forest Watch database (GFW, 2022), Harris <i>et al.</i> (2021) and FAO (2022b)
Waste	Data on waste generation, collection and recycling	Kaza <i>et al.</i> (2018)

Currently, the Cambodia SAMEA for 2022 includes 94 accounts divided into:

- Ten agriculture activity sectors, including eight sub-sectors by type of commodity produced (i.e. grains-legumes, rice, fruits, mangoes, tubers, cassava, sugarcane, other agriculture)
- Twenty-two industry sectors
- Nineteen service sectors
- One Value added sector grouping productive factors (i.e. labour and capital)

- Eleven institutional accounts represented by ten categories of representative households differentiated as rural vs urban households and further disaggregated by per capita income quintiles, and the Government account
- Seventeen polluting substances
- Five natural resources
- Five environmental themes
- Four additional accounts concerning taxes, capital formation (i.e. savings and investments), rest of the world and totals.

4.1.2.8 Modelling impact with the SAMEA

In section 2.1, we indicated that the SAM provides a snapshot of all the exchanges between sectors during a period. Each sector is represented in a row and a column of the matrix, with the rows presenting the inflows to the sector and the column the outflow from the sector. In the SAMEA, transactions between economic sectors are quantified in monetary terms, while those in the environmental sectors are reported in physical quantities. As such, the SAMEA is an accounting tool that can quantify the flows in the economy.

However, the model can be modified to perform impact analysis of project or policy changes. Estimating impact through a SAMEA requires first establishing the endogenous and exogenous accounts in our model. Values of endogenous accounts are determined within and by the model, while the values of exogenous accounts are an external shock from outside the model.

As per standard practice in using SAM-based models, we considered the government, capital formation and the rest of the world to be exogenous. Instead, the endogenous accounts consist of productive activities, services, institutions and environmental accounts.

Thus, we can separate the endogenous and exogenous accounts by partitioning the transaction matrix into submatrices of different sizes. We can define the transaction matrix G as:

$$G = \begin{bmatrix} E & X \\ L & D \end{bmatrix} \quad \text{Eq. 1}$$

where: E contains all the endogenous transactions, X shows external injections in the economy (i.e. exogenous expenditures), L represents the endogenous expenses towards exogenous accounts (i.e. leakages), and D reports the transactions between exogenous accounts.

Following the theoretical approach shown by Pyatt and Round (1979), we can transform the flows in the matrix into accounting coefficients by dividing each amount within the matrix by its respective column sum. Since we are interested in determining the model capacity to multiply external stimuli, we would only work with the endogenous accounts in E and L .

After naming F the combined submatrix of E and L , and by defying $t' = \sum_{i=1}^k (e_{ij} + l_{ij})$ as the column sum of these endogenous accounts, we obtain the matrix of average expenditure propensities by dividing each element of F by the corresponding element of vector t' as:

$$A = F t'^{-1} \quad \text{Eq. 2}$$

When the economy is in equilibrium, total demand equals total supply. Any change in the final demand activates the production process and stimulates the demand for intermediated inputs in all sectors. Thus, the change in the economy and the environment can be defined as

$$\Delta Y_i = \sum_{j=1}^n a_{ij} \Delta Y_j + \Delta X_i \quad i = 1, \dots, n. \quad \text{Eq. 3}$$

Where a_{ij} are the average expenditure propensities calculated in eq.2, ΔY_j is the intermediate and ΔX_i the final demand, whose combination defines the variation in total output ΔY_j . In matrix form and solving by Y , we define how exogenous changes in X are absorbed and further transmitted in the economy as:

$$Y = AY + X \Rightarrow (I - A)Y = X \Rightarrow Y = (I - A)^{-1}X \text{ or } Y = M_a X \quad \text{Eq. 4}$$

where Y is the final increase in total demand due to exogenous injections X compounded by the SAMEA multipliers M_a – calculated as the inverse Leontief matrix $(I - A)^{-1}$. Matrix I is the identity array. Hence, the multiplier matrix is the transmission mechanism that propagates the external shock to endogenous accounts, determining the new output and emission levels in the economy.

We further revised the multipliers in M_a to account for various modelling assumptions. Firstly, we replaced the average expenditure propensities in A and created a new matrix C with the marginal expenditure propensities (Lewis and Thorbecke, 1992; Pyatt and Round, 1979). We assume

that household expenditures differ at the margin from the average expenditure when income changes. Thus, we estimated the household income elasticities by quintile for food, non-food and total expenditure using data from the Cambodia living standards measurement survey (NIS, 2021). This way, we estimated a new multiplier matrix $\mathbf{M}_f$ accounting for expenditure elasticities. Regression results are provided in the supplementary material section.

Finally, we accounted for the unrealistic assumption that productive sectors have an unlimited production capacity that fully meets the increase in demand. We control for the fully elastic supply of some sectors by identifying supply-constrained and –unconstrained sectors and by creating a mixed multiplier matrix $\mathbf{M}_m$ as shown by Lewis and Thorbecke (1992). Following the notation adopted by Rocchi, Romano and Hamza (2013), we can partition matrix $\mathbf{C}$ as:

$$\mathbf{C} = \begin{pmatrix} \mathbf{C}_{nc} & \mathbf{Q} \\ \mathbf{T} & \mathbf{C}_c \end{pmatrix} \quad \text{Eq. 5}$$

Where $\mathbf{C}_{nc}$ and $\mathbf{C}_c$ contain the marginal expenditure propensities among the unconstrained and constrained sectors, respectively. Note that $\mathbf{C}_{nc}$ also include the propensities of factors and institutions. Submatrix $\mathbf{T}$ contains the marginal expenditure propensities of factors, institutions and unconstrained sectors on constrained sectors' output. Inversely, $\mathbf{Q}$ reports the marginal expenditure propensities of factors, institutions and constrained sectors on unconstrained sectors' output.

Given that:

$$\begin{pmatrix} \mathbf{I} - \mathbf{C}_{nc} & \mathbf{0} \\ \mathbf{T} & \mathbf{I} \end{pmatrix} = (\mathbf{I} - \mathbf{M}_m) \quad \text{Eq. 6}$$

where $\mathbf{0}$ is the null matrix, the mixed multipliers $\mathbf{M}_m$ matrix can be estimated as:

$$\mathbf{M}_m = \begin{pmatrix} \mathbf{I} - \mathbf{C}_{nc} & \mathbf{0} \\ \mathbf{T} & -\mathbf{I} \end{pmatrix}^{-1} \begin{pmatrix} \mathbf{I} & \mathbf{Q} \\ \mathbf{0} & -(\mathbf{I} - \mathbf{C}_c) \end{pmatrix} \quad \text{Eq. 7}$$

Resorting to eq. 4, the final impact of an exogenous increase in demand can be measured as:

$$\begin{pmatrix} \mathbf{Y}_{nc} \\ \mathbf{X}_c \end{pmatrix} = \mathbf{M}_m \begin{pmatrix} \mathbf{X}_{nc} \\ \mathbf{Y}_c \end{pmatrix} \quad \text{Eq. 8}$$

where the total output of unconstraint sectors and total incomes of endogenous production factors and institutions in $\mathbf{Y}_{nc}$ – alongside with the net export in constrained sectors $\mathbf{X}_c$ – are determined by the exogenous unconstrained-sectors demand $\mathbf{X}_{nc}$ and constrained sectors' output

Y_c , and the mixed multiplier matrix M_m . In other words, the exogenous stimulus will spur production in unconstrained sectors and increase imports in constrained sectors.

We disaggregated these multipliers into forward and backward effects to reveal sectors' relevance as goods and services suppliers to upstream sectors (i.e. forward) or as demand centres for downstream sectors (i.e. backwards). The forward and backward multipliers are calculated as row and column sums of the matrix M_m , respectively.

An interesting evaluation in the context of developing countries relates to the simulation of structural changes to the economy in response to policy changes. Such interventions cannot be simply studied as the effects of increasing households' disposable income since policy changes would likely affect the structural change in the economy and imply changes to SAMEA's coefficients and multipliers. For this reason, the simulation of policies capable of increasing climate mitigation and adaptation and altering sectors' productivity can be simulated through a variation of the coefficient matrix in equation 4 and equation 8. Thus, in simulating policies, equation 4 can be modified as:

$$\Delta Y = (I - A^*)^{-1}[(\Delta A)X + \Delta Y] \text{ or } Y = M_m^* X \quad \text{Eq. 9}$$

where A and A^* are the SAM matrixes with and without the change, ΔY is the vector of exogenous changes in income or capital expenditures, and M_m^* is the modified mixed multiplier matrix reflecting changes in the productivity and emission parameters brought by the policy.

Finally, we calculated emission decoupling indicators according to the methodology proposed by OECD (2002) and Tapio (2005). In particular, we calculated sectors decoupling factors and elasticities to verify decoupling trends over time. We define the decoupling factor of sector s as:

$$Df_s = 1 - \frac{E_{t_n}/GDP_{t_n}}{E_{t_0}/GDP_{t_0}} \quad \text{Eq. 10}$$

Where t_0 is the analysis start period, and t_n its end period. The second term on the right hand side of eq. 10 defines the decoupling ratio over time. Subtracting it to 1 allows us to estimate a decoupling factor that takes negative values or zero when decoupling has not occurred and a maximum value of 1 in case of absolute decoupling (OECD, 2002).

Calculating decoupling elasticities allows us to determine the strength and direction of the decoupling. We define emission elasticities of sector s as:

$$D\varepsilon_s = \frac{\Delta E_s/E_s}{\Delta GDP_s/GDP_s} \quad \text{Eq. 11}$$

with $\Delta E_s/E_s$ and $\Delta GDP_s/GDP_s$ are the emission and GDP growth rates of sectors over time, respectively. According to Tapio (2005), strong decoupling occurs when GDP grows and emissions decrease, in which case $D\varepsilon_s < 0$. Weak decoupling occurs when GDP grows faster than emissions. Possible alternative decoupling scenarios based on elasticity results are presented in Table 6.

Table 6: Alternative coupling/decoupling scenarios

Scenarios	$D\varepsilon_s$	ΔE_s	ΔGDP_s	Notes
Strong decoupling	< 0	< 0	> 0	
Weak decoupling	0 - 0.8	> 0	> 0	GDP grows faster than emissions
Expansive coupling	0.8 - 1.2	> 0	> 0	GDP and emissions grow at an akin rate
Expansive negative decoupling	> 1.2	> 0	> 0	Emissions grow faster than GDP
Strong negative decoupling	< 0	> 0	< 0	
Weak negative decoupling	0 - 0.8	< 0	< 0	GDP declines faster than emissions
Recessive coupling	0.8 - 1.2	< 0	< 0	GDP and emissions decline at an akin rate
Recessive decoupling	> 1.2	< 0	< 0	Emissions decline faster than GDP

Source: Own elaboration from Tapio (2005) and Lundquist (2021)

4.1.2.9 Modelling Cambodia's policies for a net zero economy

In this study, we tested some of the agriculture and forestry policies outlined in the updated Cambodia's nationally determined contribution (NCSD, 2020) and reflected in Cambodia's long-term strategy for carbon neutrality - LTS4CN (NCSD, 2021). The LTS4CN suggests that Cambodia could achieve carbon neutrality in 2050 with forests providing a total carbon sink of 50 MtCO₂eq to curb emissions from the energy (28 MtCO₂eq), agriculture (19 MtCO₂eq), waste (1.6 MtCO₂eq) and industrial sectors (1.2 MtCO₂eq).

Compared to the business-as-usual (BAU) scenario, where carbon emissions are expected to increase to up to 156 MtCO₂eq, Cambodia would cumulatively abate about 1,400 MtCO₂eq by

2050. Mitigation actions in agriculture are expected to reduce emissions by 15.6 MtCO₂eq, mainly by reducing methane-intensive rice and livestock production and promoting composting and producing biogas in livestock management, organic fertiliser, and deep fertiliser technology. The LTS4CN is cutting emissions by 45% with respect to the BAU scenario. Most of the emission reduction comes from rice cultivation, which is expected to be reduced by 14.6 MtCO₂eq in 2050. The near-term target for the reduction of rice emissions in 2030 is set at 9.72 MtCO₂eq, 56% less than the amount reported in 2020 (17.2 MtCO₂eq) and 54% less the total agriculture emission of the BAU scenario (21.46 MtCO₂eq).⁹

Forests are expected to provide a significant carbon sink, allowing other sectors to transition incrementally towards carbon neutrality. The forestry and other land use (FOLU) sector is expected to be carbon neutral from 2031. The LTS4CN assumes a decline in the deforestation rate by more than 50 percent in 2030, while reforestation and afforestation efforts will expand the carbon sink capacity as forests continue to grow. Deforestation is assumed to stop by 2045. Afforestation and reforestation are assumed to increase constantly to restore 1.1 million hectares of native forest restoration and reforest 1.6 million hectares. These combined actions will result in a carbon sink of 50 MtCO₂e in 2050. Table 7 summarises emission targets and expected reduction rates for specific sectors.

Table 7: LTS4CN emission scenarios for 2030 and 2050

Sector	Emissions in 2020 (MtCO ₂ eq.)	Emissions by 2030 (MtCO ₂ eq.)		Reduction from BAU (%)	Emissions by 2050 (MtCO ₂ eq.)		Reduction from BAU (%)
		BAU	LTS4CN		BAU	LTS4CN	
Whole country	81.77	97.52	52.37	46.2	156	0.3	99.8
Agriculture	19.55	23.77	11.69	50.8	35	19.4	44.5
Rice cultivation	17.25	21.46	9.72	54.7	32.12	17.52	54.5
FOLU	41.64	34	2.60	92.3	21.2	-50.2	336
Deforestation	51.26	41.85	20	52.2	26.1	0	100

Source: Own elaboration from data included in NCSD (2020) and NCSD (2021)

⁹ Since the LTS4CN did not provided emission reduction targets for 2030, we extrapolated this data through plot digitalization of graphs included in NCSD (2021) and trend linearization where appropriate.

4.1.2.10 Policy modelling

The agriculture and FOLU policies contemplated in the LTS4CN are tested in our SAMEA to verify whether the efforts outlined in the long-term strategy suffice to reach the expected emission targets in 2030. Each policy is simulated by either adjusting the multipliers matrix or revising the emission coefficients (eq. 9). We considered the budget allocated and the sector of interest for each budget line. Based on the latter, we simulated the contribution to economic growth (direct and indirect impacts) that proper implementation of the policy would bring about, as well as the environmental results (GHG and LCIA) and decoupling capacity of each policy.

The budget reflected in Table 8 is used in the SAMEA to estimate the economic and environmental impact resulting from the implementation of these policies. This budget constitutes the economic stimulus required in the short run to achieve productivity gains, emission reductions and sector development. To simulate the economic and environmental impact of these initiatives, the policy budget has been re-classified consistently with the SAMEA structure. The harmonisation procedure allows us to construct an investment matrix that considers expenditures occurring over the eight-year implementation period (see supplementary material for detailed estimates).

The LTS4CN and the NDC programmatic plan (NCSD, 2020, 2021) do not specify the contribution of each policy to GHG emission reductions. However, the LTS4CN contemplated three emission reduction scenarios for FOLU, establishing emission reduction targets by 2030 (Table 9). Thus, we used the assumptions of these scenarios to model and validate the consistency and capacity to reach carbon neutrality targets by 2030.

Table 8: LTS4CN policies and budgets in the agriculture and forestry sectors

Policy no.	Action	Sector	Scope/target	Cost (US\$ mil.)
1	Adaptation	Agriculture	Climate Change mitigation as the project considering low-carbon agricultural technology and improved new management approach in agroecosystem	10

2	Adaptation	Agriculture	Promoting climate-friendly agribusiness rice value chain and mitigating the impact of rice farming on the environment.	37
4	Adaptation	Agriculture	Promoting climate-friendly agribusiness value chain and reducing soil degradation and greenhouse gas emissions through CSA practices	12.5
5	Adaptation	Agriculture	Increasing resilience to climate change and natural disasters and improving resilience facilities, tools and technologies.	69.6
8	Adaptation	Agriculture	Promotion of the environment and conservation of natural resources	1.5
16	Mitigation	Forestry/energy	Strengthen climate-resilient energy through increased clean charcoal production	10
22	Mitigation	Forestry	Reducing GHG emissions by avoiding the extraction of the forest from Angkor Conservation Park and Kulen Conservation Park for housing, firewood consumption, and agricultural land.	7
25	Mitigation	Forestry	Promoting sustainable sourcing of fuel wood in the garment industry	19.6
34	Mitigation	Agriculture	Increasing the effectiveness and sustainability of agricultural land management techniques through conservation agriculture	25
REDD+	Mitigation	Forestry	Strategic Objective 1 – Improving management and monitoring of forest resources and forest land use (2022-2030)	16.3

REDD+	Mitigation	Forestry	Strategic Objective 2 – Strengthening implementation of sustainable forest management (2022-2030)	53.5
REDD+	Mitigation	Forestry	Strategic Objective 3 – Mainstreaming approached to reduce deforestation, building capacity and engaging stakeholders (2022-2030)	11
Total				273

Source: From data in NCSD (2020, 2021) and RTS (2020)

Table 9: FOLU policies and GHG emission reduction targets

Scenarios	Objectives	Emission reduction (MtCO ₂ eq./year)	Expected change by 2030
FOLU NDC Scenario 1	Forest covering 60% of Cambodia's territory by 2030. Deforestation stops by 2045	302.8 (92% of total emission reduction)	Forest coverage: +15% (+2.7 Mha) or +339,923 ha/yr
FOLU NDC Scenario 2	Halving historical emissions from FOLU (76.3 MtCO ₂ e) by 2030. An average annual reduction of 21 MtCO ₂ e/year.	38.1 (59% of total emission reduction)	Deforestation rate reduction of 50% (38.1 MtCO ₂ e)
FOLU NDC Scenario 3	Maintaining 60% forest cover of the total land area with the assumption that the starting point was 57%.	3.8 (12.5% of total emission reduction)	Forest coverage: +3%

4.1.3 Results and discussion

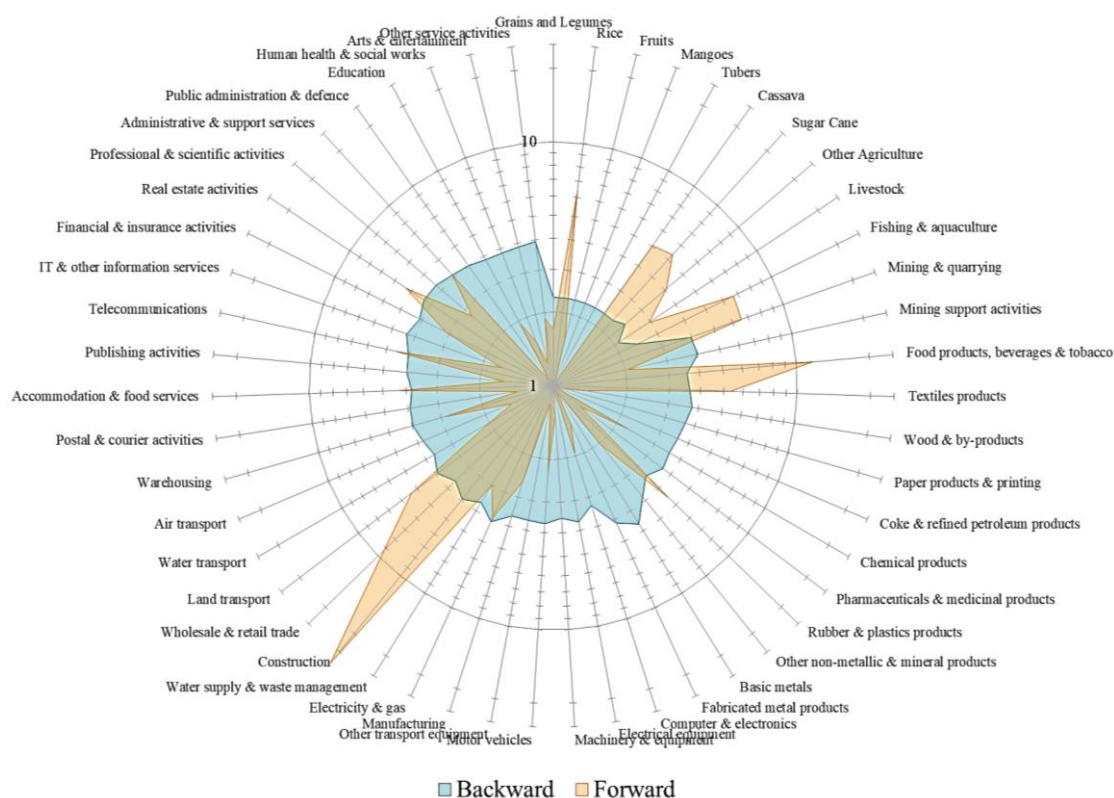
Here, we present investments and policy reform results through four simulations. The first simulation displays the prevailing economic and environmental linkages between upstream and downstream sectors subdivided in the SAMEA. The second simulation analyses the decoupling trends for the various Cambodian economic sectors. The third simulation reports the economic impact of implementing the set of interventions foreseen in the LTS4CN. The fourth simulation shows the environmental impact and performance of the FOLU policies against carbon neutrality targets.

4.1.3.1 Economic and environmental linkages and multipliers in Cambodia

The first set of results from the SAMEA is obtained by looking at the economic multipliers and linkages underpinning the economy. The latter describes how exogenous expenditures in one sector are transmitted to the interconnected sectors in the economy and reveals the sector's ability to either absorb or transmit further the economic stimulus through multiplying economic effects. Sectors with low forward multipliers sell their output mostly to final demand. Vice versa, sectors possessing high forward multipliers provide intermediate inputs to upstream industries. Opposite, low backward multipliers indicate that a sector's dependence on other sectors' inputs is comparatively very low (i.e. inputs are sourced mainly from imports). On the contrary, sectors with high backward multipliers show a stronger bond to local industries downstream, which are responsible for the supply of intermediate goods.

Figure 4 shows that Cambodia's highest forward multipliers are in the construction, rice and food products and beverages sectors, which are relevant input suppliers to all other industries and sectors upstream. Other agricultural sectors – except for rice, fishing, sugarcane and cassava – show lower forward multipliers, meaning they mostly sell their produce to final demand rather than agro-processing industries. The backward multipliers in Cambodia present a more homogenous set of results, except for agriculture activities, which are less integrated into the economy.

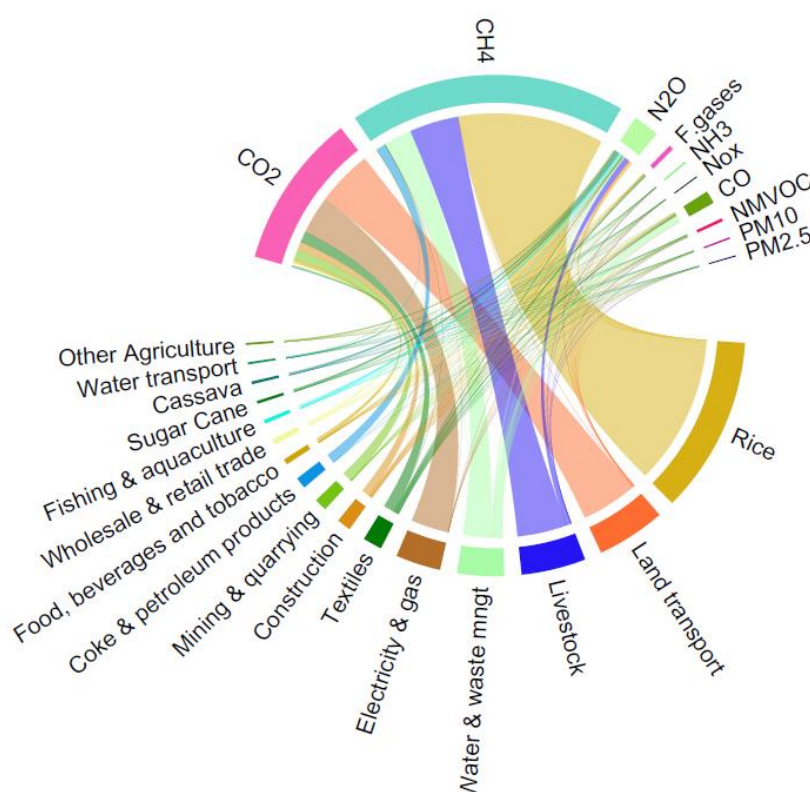
Figure 4: Backward and forward multipliers for Cambodia (log values)



Source: Own elaboration

Another interesting evaluation is determining the origins and destination of the main pollutants in the economy. Figure 5 reports gas emissions for the ten utmost gases by the 16 most polluting economic sectors. The graph excludes emissions from FOLU, which currently accounts for about 50% of the total emission. This exclusion was necessary for a balanced visualisation of other sectors' emissions. In that respect, rice cultivation and livestock farming are the largest contributors to CH₄ emissions. At the same time, land transportation and the generation and distribution of electricity and gas are responsible for most of the CO₂ emissions. The latter is due to the dependency on coal-based technologies that are still largely used in the Cambodian energy sector. Meanwhile, CH₄ and CO₂ account for the greatest share of gas emissions.

Figure 5: Disaggregated view of gas emissions by sources



Note: CO₂ emissions from FOLU are excluded. They account for over 50% of the total country's GHG emissions.
Source: Own elaboration

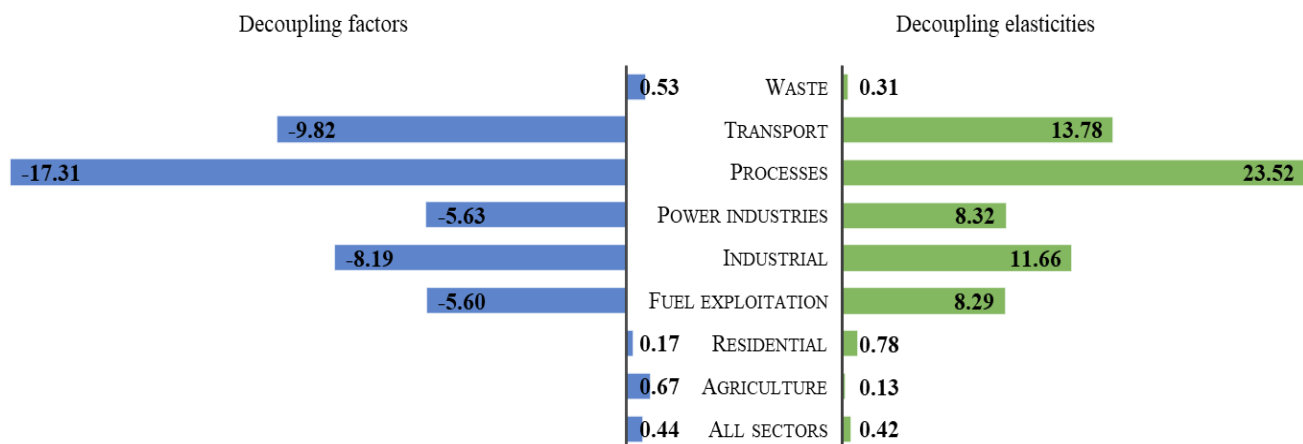
Nonetheless, with a total population of about 17 million people (UNDESA, 2022), Cambodia has an annual per capita GHG emission of 3 tCO₂ eq. This is lower than the world average of 4.7 tons of CO₂ eq. and less than half the European average of 7.3 tCO₂ eq. However, GHG emissions in Cambodia have increased by 73% since 2005, registering an emissions increase of more than 300% in transportation, industrial processes and energy power supply (Crippa *et al.*, 2023).

4.1.3.2 An analysis of the economy decoupling trends

The analysis of carbon intensity reveals that weak decoupling is occurring in Cambodia, where results from all sectors combined report a factor of 0.44 (Figure 6). However, since the indicator is less than 1, the decoupling is classified as weak. The detailed analysis reveals similar results in agriculture, residential, and waste management sectors, all of which present weak decoupling. Only in the agricultural sector are the decoupling elasticities approaching the strong decoupling thresholds. However, when comparing the agriculture sector emission with the agriculture GDP, the decoupling factor and elasticity bounce back to 0.28 and 0.44, respectively,

meaning that weak decoupling also prevails in the agricultural sectors. Opposite, all the remaining sectors present expansive negative decoupling trends, with sectors' emissions growing much faster than the related GDPs.

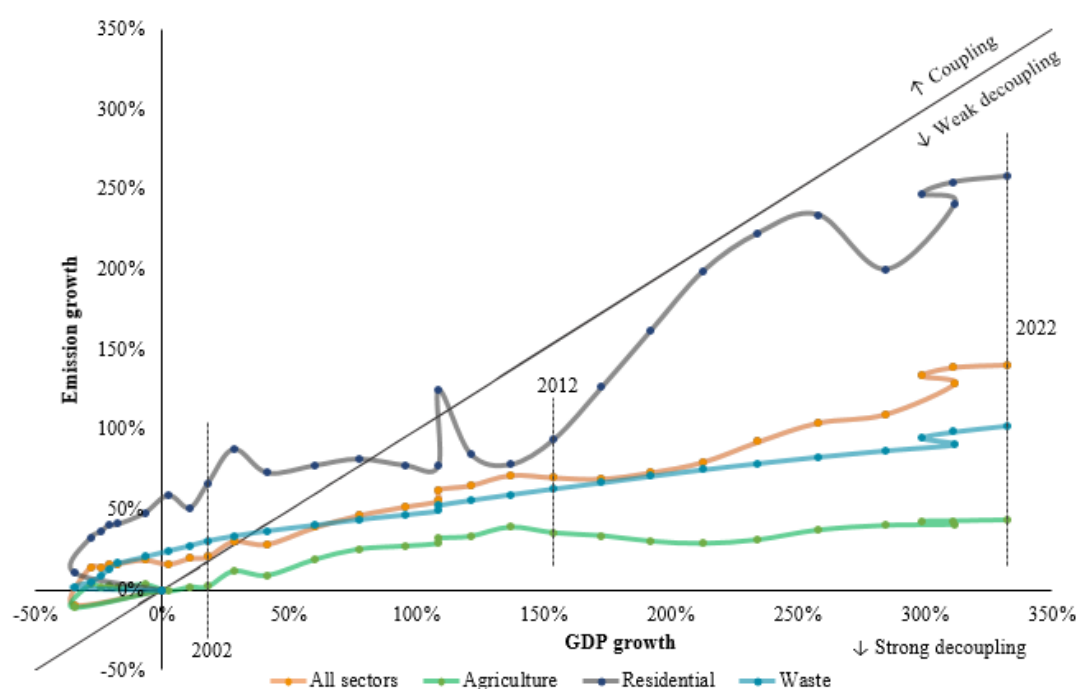
Figure 6: Cambodia decoupling factors and elasticities in 2022



Note: Power industry includes power and heat generation plants. Industrial includes industrial manufacturing and process. Transport includes all transportation means and modalities. Residential relates to services and other sectors. Waste includes solid waste management and wastewater handling. Fuel exploitation relates to fuel extraction, transformation and refining. Agriculture relates to all agricultural activities, including livestock rearing and fishing.

A time-series analysis of decoupling indicators for those sectors presenting weak decoupling reveals that between 1993 and 2006, these sectors went from strong negative decoupling (i.e. strong coupling) to weak decoupling (Figure 7). However, this positive trend led to an upward shift towards coupling since 2011, leading to a relative deterioration of results. In other words, the decoupling trend tends to deteriorate rather than improve.

Figure 7: Decoupling trends for selected sectors from 1993 to 2022



4.1.3.3 Economic impacts of the mitigation and adaptation policies in forestry and agriculture

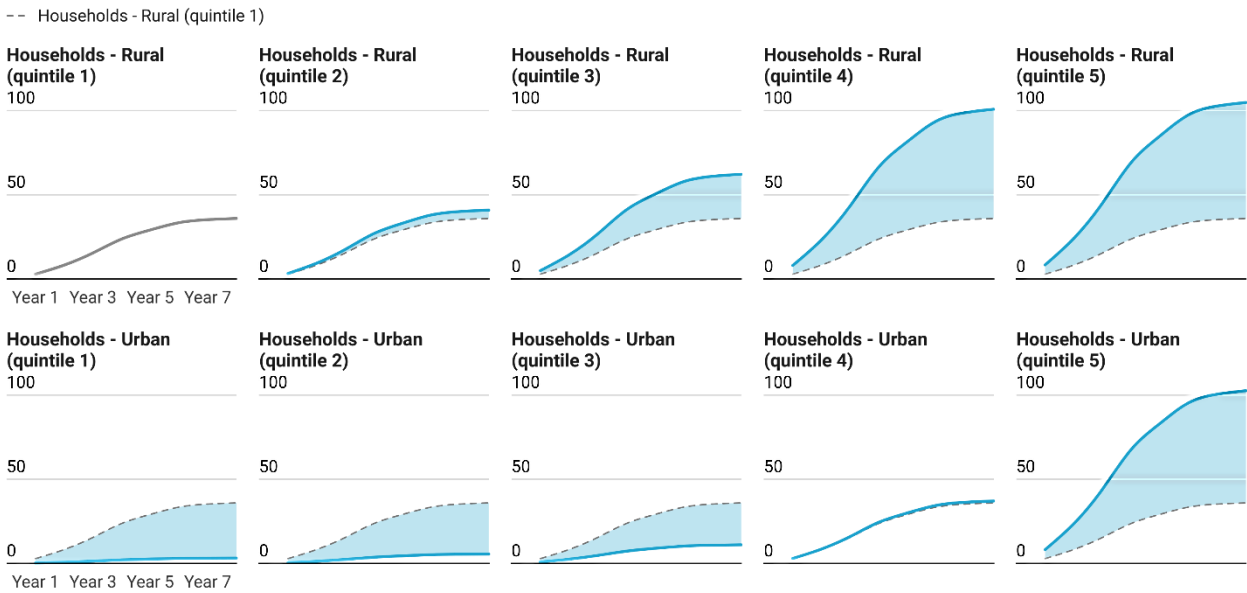
Economic impacts in the short run are related to the implementation of the policies contained in Table 8. Over the expected enactment period, the budgetary expenditure will impact national GDP, household and government incomes, per capita consumption and value-added.

Policy implementation contributes to national GDP growth. In particular, our simulation determined a cumulative impact on the Cambodian economy of USD 352 million over the eight-year implementation period, with a multiplier effect of 1.57 per USD spent. Out of the total value created, USD 339 million is linked to the direct and indirect sectoral stimulus generated by the policies. An additional USD 13.4 million is associated with incremental tax revenues for the government. Therefore, the set of mitigation and adaptation policies analysed will contribute to an annual GDP growth of 0.17 percent.

We could also analyse the economic impact on household income distribution, differentiated by wealth consumption quintiles (per capita) and living settings (rural or urban). As shown in Figure 5, policy implementation would benefit primarily rural households and predominately the richest quintiles (fourth and fifth). However, the present value of the total

income generated gets distributed more evenly across rural households rather than within the urban household cluster, where most of the increase is captured by the richest quintile (Figure 8).

Figure 8: Cumulative economic impact on households' income by quintile (mil. USD)



Source: Own elaboration

4.1.3.4 Short-term economic and environmental effects of the GCF Investment in Cambodia

We monitored the emission of main substances and the consequences of relevant environmental themes (see Table 1 and Table 3) linked to the implementation of mitigation and adaptation policies compared to the business-as-usual scenario. In particular, to validate the forecast of FOLU emission reductions (Table 7), we assumed that the current deforestation rate will remain unchanged without LTS4CN policies. Currently, forests in Cambodia cover 47% of the total country area (8.06 Mha), and the average annual deforestation between 2015-2020 equals 193,270 ha (FAO, 2022b). Five simulation scenarios are tested to verify the expected results on the carbon neutrality targets for Cambodia (Table 10).

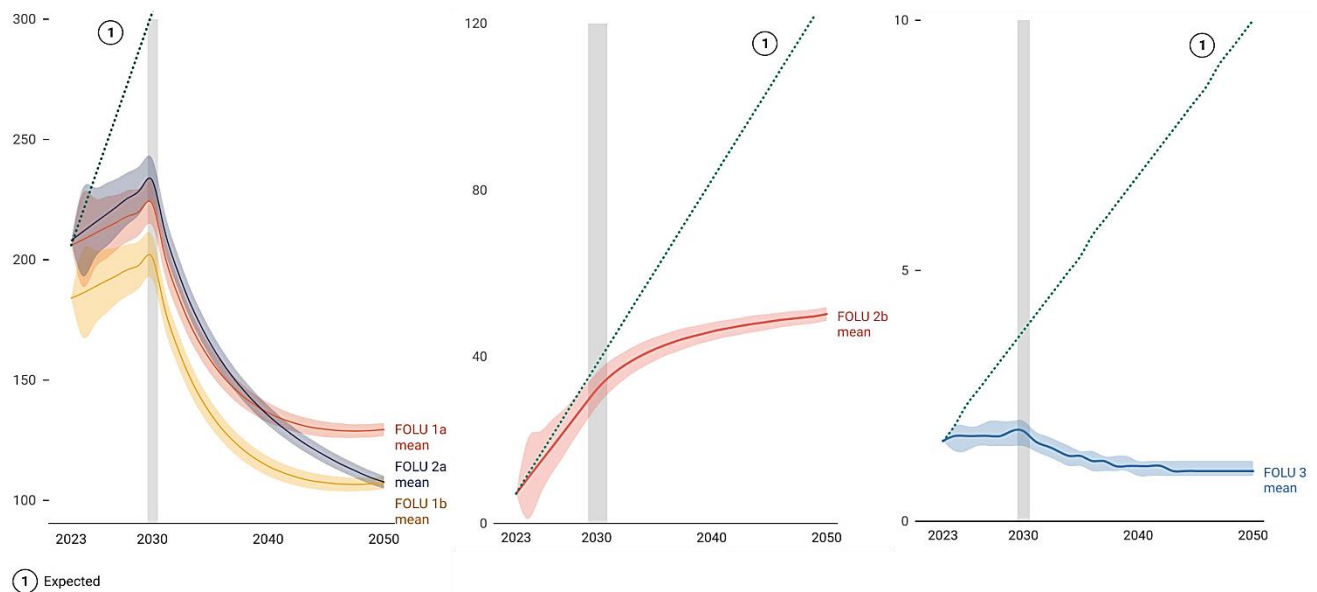
Table 10: Policy simulation scenarios

Simulation Scenarios	Afforestation	Deforestation	2030 reduction targets (MtCO₂eq./year)	Emission reduction coefficient (tCO₂eq.year.ha)
FOLU 1a	+339,923 ha/year till 2030 +37,580 ha every year	-8,403 ha/year till 2045	302.8	591
FOLU 1b	+339,923 ha/year till 2030	-8,403 ha/year till 2045	302.8	591
FOLU 2a	+339,923 ha/year till 2030 +37,580 ha every year	-12,079 ha/year till 2030	302.8	591
FOLU 2b	+37,580 ha every year	-12,079 ha/year till 2030	38.1	591
FOLU 3	+339,923 ha/year till 2030 +37,580 ha every year	-8,403 ha/year till 2045	3.8	4.7

Our analysis indicates that none of the scenarios foreseen in LTS4CN are consistent with emission reductions that allow for carbon neutrality by 2050. In all scenarios, the afforestation and deforestation rates fall far from the 2030 targets, and the gap from the planned route widens further as time goes by (Figure 9). Without a bolder afforestation strategy and continued efforts to halt deforestation, carbon neutrality through FOLU action remains an elusive goal. However, the targets set in the FOLU sectors already appear optimistic, and it is hard to imagine how much more they can be stretched. The 2030 emission reduction targets can be achieved only if yearly afforestation exceeds 520,000 ha (+150%) from the assumed reforestation target), with a monthly afforestation of 43,333 ha. These targets are particularly ambitious, especially when considering that between

2015 and 2020, deforestation – notably rolling out faster than afforestation – has taken place at an average monthly pace of 7,727 ha.

Figure 9: Annual GHG emission reduction against 2030 targets (MtCO₂eq/year)



Finally, Table 11 shows the results of selected environmental themes concerning policies and the BAU scenarios for 2030. The FOLU scenarios substantially improve the GHG and the effects of ocean acidification. The three scenarios with the highest sequestration potential (i.e. FOLU 1a, 1b and 2a) are those where the greatest effects are marked, also since these environmental themes rely heavily on carbon sequestration. On the contrary, the effect on the other environmental themes – especially on health-related indicators – is minimal, considering the lowest weight exerted by carbon sequestration on the environmental theme calculation.

Table 11: Policy simulation results on environmental themes

Environmental themes	2022	BAU 2030	Policy 2030	% change Policy & BAU 2030
GHG effect (MtCO ₂ eq)	44.39	234.47	-871.41	-472%
FOLU 1a			-1551.38	-762%
FOLU 1b			-1373.72	-686%
FOLU 2a			-1629.59	-795%
FOLU 2b			-22.57	-110%
FOLU 3			220.20	-6%
Ocean acidification (MtCO ₂ eq)	38.88	194.67	-911.21	-568%
FOLU 1a			-1591.17	-917%
FOLU 1b			-1413.51	-826%
FOLU 2a			-1669.39	-958%
FOLU 2b			-62.37	-132%
FOLU 3			180.41	-7%
Terrestrial acidification (MtSO ₂ eq)	0.10355	0.00223	0.00224	0.344%
Human health damage (MtPM _{2.5} eq)	0.096339	0.324698	0.324699	0.0002%
Human health damage due to fine dust formation (Mt)	0.0699245	0.2058518	0.2058522	0.0002%
Human health ozone formation potential (MtNo _x eq)	0.266242	0.005205	0.005207	0.039%
Marine biodiversity loss (PDF yr ⁻¹ kg ⁻¹)	0.00000000417	0.00000000914	0.00000000914	0.000005%
Ecosystem ozone formation potential (MtNo _x eq)	0.1279344	0.0025117	0.0025124	0.027%

4.1.4 Conclusions

In this paper, we researched the potential and limitations of green growth policies in supporting Cambodia's transition to carbon neutrality by 2050. We found that green growth policies and strategies in the forestry sector can promote pro-poor growth. However, they are unable to meet emission reduction targets that align with the Paris Agreement.

Further, our analysis has demonstrated that economic growth and emissions are not decoupling and that sectoral growth and GHG emissions are not linked by a one-way linear transmission. Instead, their causal relationship is mostly determined by the strength of both backward and forward linkages between sectors. Ultimately, the multiplier coefficients tying the economic and environmental sectors are responsible for the direct-indirect changes in both the economy and the environment.

The contribution of this paper also relates to constructing the first-ever SAMEA in Cambodia. We showed that comprehensive analytical tools, such as this one, can effectively display the connections between the socio-economic and the environmental domains. Furthermore, we demonstrated that such a type of analysis could be used to better advise policy preparation and inform stakeholders on the expected economic and environmental consequences that development strategies may bring about.

The findings from this study could impact policy decisions in Cambodia related to environmental management strategies, resource allocation, economic development, and climate change adaptation and mitigation. By using our findings and data on environmental factors and impact, policymakers can create specific strategies to address urgent issues such as deforestation, water pollution, and habitat loss and promote more sustainable land use practices. Additionally, by analysing the distribution of ecosystem services and their economic value, policymakers can allocate resources effectively and direct investments toward preservation (refer to paper 2) or climate adaptation measures. In short, our study aims to provide policymakers with insights based on data, bridging the gap between environmental science and effective policy formulation.

Finally, a limitation of our study is the inability to analyse the long-term policy impact within the context of technological progress. Nonetheless, we have established a methodological framework that may help define how development strategies, such as those promoted under Cambodia's LTS4CN, could enhance the environment and reduce greenhouse gas emissions. Another solution is integrating the analysis with computable general equilibrium models, which can account for changes in consumption and production functions and accurately forecast the emission

reduction trajectory by 2050 resulting from technological advancements. However, while offering improved predictability for long-term outcomes, these models also introduce greater uncertainty due to the required modelling assumptions.

4.2 Paper 2: Green vs. Grey: Evaluating Economic and Ecosystem Services Values of Nature-based Solutions in Small Island States

Abstract: This study employs economic evaluation methods and remote sensing techniques to estimate the monetary value of coastal ecosystems in nine Small Island Developing States (SIDS). The study zeroes in on mangrove forests, coral reefs, and seagrass beds, evaluating their economic relevance via cost-benefit analysis. The findings show that coastal ecosystems provide vital services, including carbon sequestration, coastal protection, tourism, and fishery, which are crucial to supporting local livelihoods and business opportunities. Despite the viability of restoration through Nature-based Solutions (NbS), focusing on mangrove forests yields better results. While coral reefs and seagrass beds are relevant, they are more expensive to restore. Therefore, effective conservation requires systematically integrating biological, environmental, and socio-economic components in ecological and conservation activities. Restoration efforts must address multiple ecosystems simultaneously. Moreover, global-level data remains incomplete and fragmented, underscoring the importance of conducting in-country assessments before defining restoration solutions. In conclusion, informed decisions and standardised tools are crucial for the sustainable management of coastal ecosystems.

4.2.1 Introduction

4.2.1.1 Study scope and structure

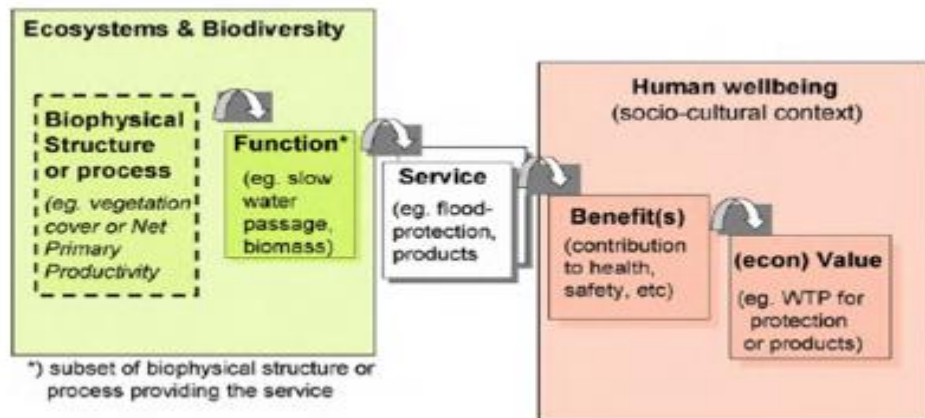
This study investigates the cost and benefit of coastal nature-based solutions (NbS) for the restoration and conservation of mangroves, coral reefs and seagrass beds in nine Small Islands Development States (SIDS) (i.e. Belize, Comoros, Dominican Republic, Guinea-Bissau, Papua New Guinea, São Tome, Solomon Islands, Suriname and Vanuatu). Chapters 4.2.1.2 and 4.2.1.3 provide an overview of the methods and challenges in evaluating ecosystem services. Chapter 4.2.2 includes a review of the existing literature and estimates of the economic values of the various ecosystem services globally. Chapter 4.2.3 reports a cost comparison between green and grey solutions for coastal protection. Chapter 4.2.4 describes the research method employed in this study and indicates the limitations encountered. Finally, chapter 4.2.5 shows the analysis results for the nine SIDS under review.

4.2.1.2 The need for the valuation of ecosystem services

The evaluation of ecosystem services (ES) is often neglected, although human beings depend entirely on the continuation of natural cycles for their existence (Agarwala *et al.*, 2014; Daw *et al.*, 2016; West, Igoe and Brockington, 2006). More than 80 percent of the value of ES is not captured in markets (Groot *et al.*, 2010), and according to Costanza *et al.* (1997), missed or improper valuation of ES is due to its minimal consideration in policy decisions, which in turn explains their exclusion from commercial markets. Similarly, their misevaluation can be linked to the asymmetry of information regarding the parties benefitting from them (i.e. those who enjoy the benefits are unaware of their values). Also, Carpenter *et al.* (2009) linked the missed ES evaluation with the disconnect between the benefits provided and the locations where the costs are borne. Irrespective of the reasons behind it, our society should pay attention to the continuous degradation and unsustainable use of ecosystems, which eventually will affect human welfare and the delivery capacity of ES in the future.

Figure 10 shows the linkages between ES and human well-being, and it describes how an environmental function provides a service, which in turn offers a benefit to human well-being.

Figure 10: Linkages between ES provision and human welfare (Groot et al. 2010)



The possibility of estimating the economic value of ES should be considered along with the needs and advantages that an economic valuation might deliver. Supporters of quantitative economic assessments believe that valuations can (i) improve understanding of problems and trade-offs; (ii) be used directly to increase the range of evidence-based decision-making, therefore increasing efficiency; (iii) illustrate the distribution of benefits and thus facilitate cost-sharing for management initiatives, and (iv) spur the creation of innovative institutional and market instruments that promote sustainable ES management (Dasgupta, 2021; Dasgupta, Levin and Lubchenco, 2000; Salzman, Thompson Jr and Daily, 2001).

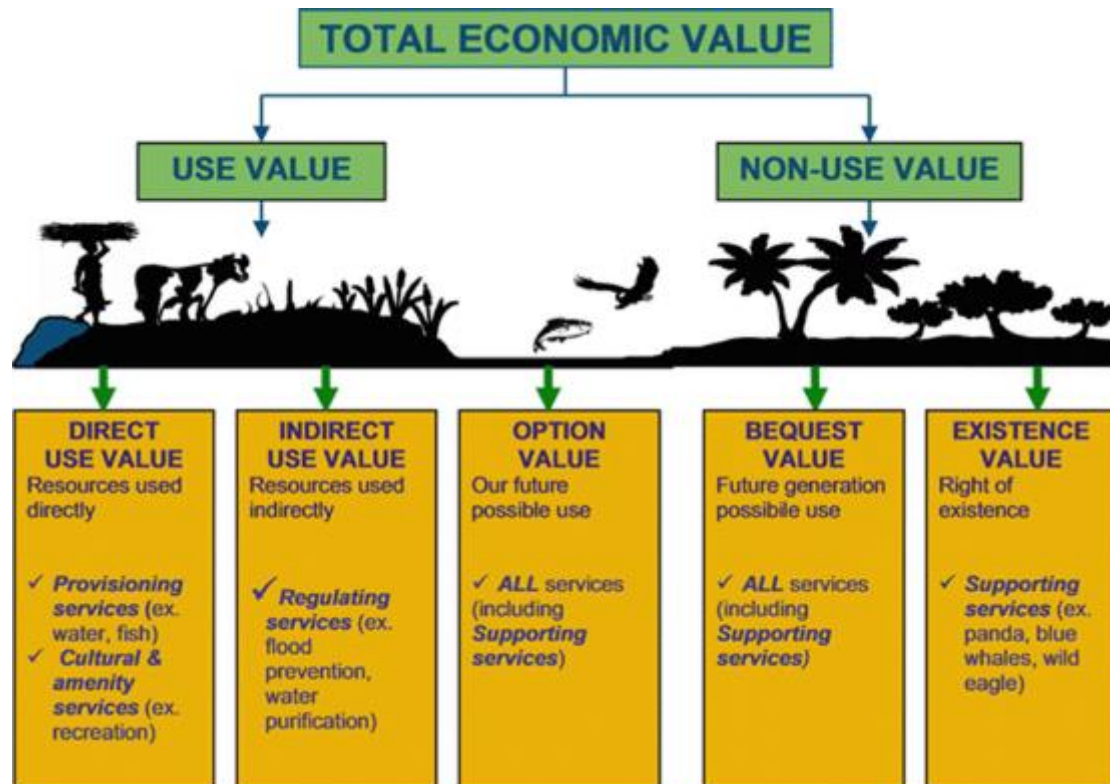
Although there is still a long way to go regarding the comprehensive evaluation of ES, many authors tried to identify how these services can be measured in monetary terms (Bishop *et al.*, 2010; Boyd *et al.*, 2015; Christie *et al.*, 2012; Fisher and Turner, 2008; Hearne, 1996; Rietbergen-McCracken and Abaza, 1998; Stern, N., 2007). A detailed discussion of ES global and local estimates is provided in Chapter 4.2.2.

4.2.1.3 Types of values and valuation methods

Multiple methods were developed to estimate ES's total economic value (TEV). The TEV is the sum of all the use and non-use values for the goods or services that the ecosystem can provide over a certain period. Use values are based on actual use of the environment (e.g. hunting, potable water consumption). Instead, non-use values are associated with the ecosystem's existence rather

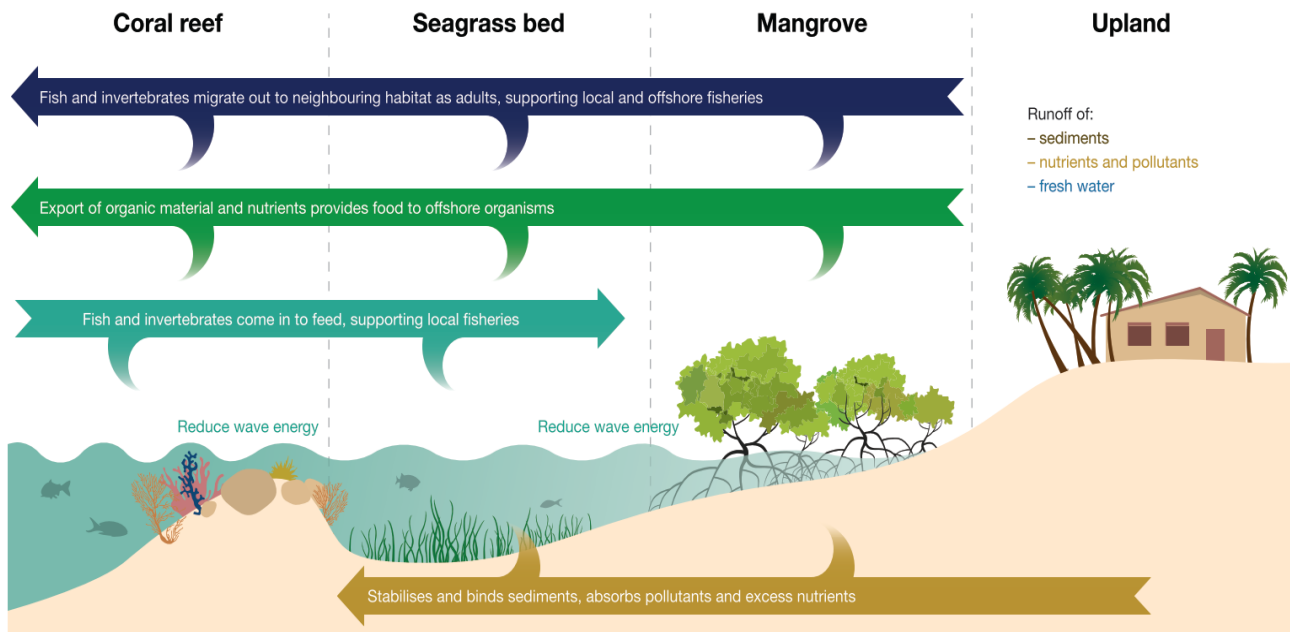
than the actual or optional use of its services. Figure 11 summarises the main elements considered in the TEV framework.

Figure 11: The TEV framework (Groot et al., 2006)



However, the interactions between coastal ecosystems can be multiple, and, in principle, the related ES would need to be understood and valued comprehensively. However, many ES are not traded in markets and remain unpriced in practice. In this case, it is necessary to assess the relative economic worth of these goods or services using non-market valuation techniques. Figure 12 shows the numerous interactions existing in coastal ecosystems.

Figure 12: Interaction between coastal ecosystems (Duke et al., 2014)



Different valuation methods are used to evaluate various ES. They can be summarised as follows:

- Production function analysis:** This method values the contribution of an ES to the production of some markets/commercial services such as drinking water or fish harvest (Crentsil and Ukpong, 2014; Daily and Matson, 2008; Elsin, Kramer and Jenkins, 2010; Sharma and Leung, 2000). This analysis considers the cause-effect relationship between the valued ES and the marketed commodity's output level.

The main limitation of this method is the lack of adequate data and understanding of cause-effect linkages between the valued ES and the marketed commodity, as well as the reconciliation of complex environmental dynamics within a simplified theoretical approach (Müller *et al.*, 2011). Other authors question how an ES can be priced within markets that do not value it sufficiently (Norgaard, 2000).

- Travel cost method (TCM):** The value of an ES relies on travel expenditures incurred to visit a site (Farber, Costanza and Wilson, 2002; Garrod and Willis, 2001). This approach is applied to recreational and other site-specific activities or resources that necessitate travel costs to experience the goods or services associated with the site. For example, to evaluate recreational

fishing, this method will consider the travel costs, license fees, onsite expenses and capital expenditure on fishing equipment (Shrestha, Seidl and Moraes, 2002). The main limitation of this method concerns the incapacity to value ES that is invisible yet commonly appreciated or well understood, such as nutrient cycling capacity or sediment control. Moreover, in TCM, the analyst's judgements can significantly influence the final value of an ES (Bingham *et al.*, 1995).

- c. *Hedonic pricing (HP)*: This approach is mainly used to evaluate environmental amenities (e.g. natural resorts and famous natural landmarks). It relies on the assumption that a person values a service based on the attribute it possesses (Farber, Costanza and Wilson, 2002; Garrod and Willis, 2001). Through this methodology, analysts might compute the price of specific services through econometric analysis and by observing market prices – when available – of the intrinsic goods' characteristics. The HP method applies to situations where the price of a market good reflects access to an ecosystem good or service (Garrod and Willis, 2001). For example, price differences between houses in areas with different air pollution levels would provide a proxy variable for air pollution's contribution to real estate pricing (Azmi, Azhar and Nawawi, 2012; Reichert, 2002) and – more importantly – the monetary value of the ecosystem regulating services (Barbier and Hanley, 2009; Czembrowski and Kronenberg, 2016; Wilson and Carpenter, 1999).
- d. *Defensive/averting expenditures*: This method relies on the existence of technological substitutes for some ES. For example, a water treatment plant is a substitute for the water purification services of a forested watershed. The costs of building a water treatment plant can be used to measure the value of the water treatment service provided by the forest (Gómez *et al.*, 2023; Osuna *et al.*, 2012).
- e. *Commercial value*: This method is used for those ES with market prices. For example, coastal wetlands provide critical habitats for commercial fish species with market values. In this case, the commercial value of the wetlands for fish production is the present value (based on market prices) of the future fish resulting from protecting the wetland (Krieger, 2001). However, that commercial value is incomplete since it excludes non-commercial species.

- f. *Economic impact*: Many studies estimate the total economic impact of an activity on the regional economy. This method includes income, profits, employment and tax revenue. It consists of the indirect effect of local businesses that benefit from the activity under analysis.
- g. *Contingent valuation (CV)*: Many shortcomings of evaluation methods are due to the lack of formal or informal markets – hence prices – for those goods and services being reviewed. Under these circumstances, a solution is to consider the people’s willingness to pay for using an ecosystem good or service. The procedure is based on hypothetical market scenarios in which people get asked to express their desire for an environmental good or service to define a demand function. The drawback of this method is that it is not based on actual market behaviour but on hypothetical scenarios. Numerous critics have pointed out that this technique is fraught with technical and conceptual problems (Chee, 2004). For instance, prior knowledge, preconceived opinions, and level of understanding of those interviewed can substantially influence their willingness to pay, thus influencing their CV results.
- h. *Benefits transfer (BT)*: This method is used to circumvent time and resource constraints while carrying out new environmental valuations. This procedure tries to transfer to the new study site (known as the ‘policy site’) the economic estimates drawn from past study sites (known as ‘study sites’). A correct application of this method requires that the two locations share similar ecological features. The transfer can be positive (benefit) or negative (damage). Thus, a more general term would be value transfer (Navrud and Ready, 2007).

Table 12 summarises the two main approaches to benefit transfer with two variations each. In calculating the country ES values, we applied the unit transfer with income adjustment to adjust international estimates to the prevailing economic conditions in SIDS.

Table 12: Types of benefit transfer methods

Unit value transfer	Simple unit transfer
	Unit transfer with income adjustments
Function transfer	Benefit Function from one study
	Meta-analysis

Source: Adapted from Navrud and Ready (2007)

The 'Unit value transfer' assumes that the marginal value to an average individual at the study site from the environmental good is the same as the average individual will enjoy at the policy site. Thus, we can directly transfer the mean willingness-to-pay (WTP) estimate from the study site to the policy site.

The 'Function transfer' approach is conceptually more appealing than just transferring unit values because it uses more information about the differences between the policy site, the study site, and the affected populations. The benefit relationship to be transferred from the study site(s) to the policy site could be estimated using the travel cost approach, hedonic pricing methods, or contingent valuation method. The main problem with the benefit function approach is the exclusion of relevant variables in the supply or demand functions estimated in a single study. This problem can be resolved by transferring results from several valuation studies and combining them in a meta-analysis to assess one joint benefit function.

Meta-analysis has been used to synthesise research findings and improve the quality of literature reviews of valuation studies to develop adjusted unit values. In a meta-analysis, original studies are analysed as a group. The results from each study are then treated as a single observation in a new analysis of the larger combined data set. The aggregation allows for pooling evidence from multiple sources and validating findings from single studies to determine the average effect and magnitude of the phenomenon analysed. This method is particularly useful for defining the most likely result in cases where single case studies are inconclusive and contradicting (Magalhães Filho *et al.*, 2021; Schmidt, 2008). In addition, this enables evaluating: a) the effects related to the environmental good's characteristics; b) the features of the samples used in each analysis (including aspects of the population affected by the change in environmental quality); c) the valuation method used, and d) the modelling assumptions (Magalhães Filho *et al.*, 2021; van den Bergh *et al.*, 1997).

4.2.2 A meta-review of the economic values of ecosystem services

4.2.2.1 Global and local values of ecosystem services

4.2.2.2 Coastal wetlands

Several recent attempts have been made to quantify the economic relevance of coastal ecosystems globally (Beck *et al.*, 2018; Groot *et al.*, 2012; Losada *et al.*, 2018; Rao *et al.*, 2015;

Spalding *et al.*, 2017). However, there is no consensus on the findings and results across studies, which are highly heterogeneous.

For Costanza *et al.* (2021), the median annual global value of coastal wetlands for storm protection equals US\$ 447 billion yr⁻¹ (in 2015 US\$) and saves up to 4,620 lives each year. Their estimates indicate that the 40 million hectares (ha) of coastal wetlands in storm-prone locations are worth an average of US\$ 11,000 ha⁻¹ yr⁻¹ in avoided storm damages.

In another recent analysis, Eric *et al.* (2022) reviewed 45 studies and found a high variation in ES values across countries. The mean value for wetland-based provisioning ES averaged US\$ 1,645 ha⁻¹ yr⁻¹ (in 2018 US\$; standard deviation (SD): ±US\$ 3,168). Instead, the mean value for wetland-based regulating ES was US\$ 8,711 (SD: ±US\$ 22,375) ha⁻¹ yr⁻¹. They also estimated wetland values through a meta-regression model. Results ranged from US\$ 0.62 to US\$ 11,216 ha⁻¹ yr⁻¹ for regulating services and from US\$ 0.95 to US\$ 2,122 ha⁻¹ yr⁻¹ for provision services. The estimates diverge significantly depending on the wetland location and other changes in the model variables.

Table 13 summarises detailed estimates by type of service provided and the TEV from recent global evaluations of the coastal wetland ecosystems.

Table 13: Global economic values of coastal systems (US\$ ha⁻¹ yr⁻¹)

Provision services	Regulating services	Cultural services	Habitat services	TEV	Price level	Source
2,396	25,847	300	375	28,917	2007 (Int. US\$)*	Groot <i>et al.</i> , 2012
—	—	—	—	8 944	2007 (Int. US\$)*	Kubiszewski <i>et al.</i> , 2017
44	193	127	—	364	2015 (€)	Magalhães Filho <i>et al.</i> , 2021
3 – 12,400	81 – 3,596	—	—	—	2018 (US\$)	Eric <i>et al.</i> , 2022

*: International dollars refer to dollar values adjusted for the purchasing power parity level of the local currency

4.2.2.3 Mangroves

Additional studies have focused on the economic benefits of mangroves both at the global and national scale (Brander, Florax and Vermaat, 2006; Groot *et al.*, 2013; Losada *et al.*, 2018;

Menéndez *et al.*, 2020; Rao *et al.*, 2015; Su, Friess and Gasparatos, 2021). The increasing number of storm protection, erosion control, and waste treatment studies allowed refining value estimates on some biomes – producing significant increases in worth. For instance, (Costanza *et al.*, 2014) report that globally, mangroves' value increased from about US\$ 14,000 to around US\$ 194,000 ha⁻¹ yr⁻¹. Worldwide, Menéndez *et al.* (2020) estimated that mangroves reduce annual flooding for about 15 million people and save about US\$ 65 billion annually from flood damage. Without mangroves, 29 percent more land, 28 percent more people, and 9 percent more property would be damaged annually.

At a national level, these values vary greatly depending on the country's coastal settings and national gross domestic product (GDP). In absolute terms, the countries receiving the most significant risk reduction benefits from mangroves are Guinea, Mozambique, Guinea-Bissau, Sierra Leone, Madagascar and Solomon Islands (Losada *et al.*, 2018). However, in relative terms, mangroves provide critical flood protection in countries with lower GDPs – where exposure is concentrated along vulnerable coastlines. Belize, Suriname and Mozambique are among the nations where flood protection from mangroves accounts for more than 17 percent of the national GDP (Menéndez *et al.*, 2020).

However, as pointed out by Su, Friess and Gasparatos (2021), there is no established consensus on mangroves worth, as estimates of mangrove restoration benefits range widely (US\$ 146 – US\$ 510,759 ha⁻¹ yr⁻¹), reflecting the significant differences in valuation methods, mangrove productivity, valued ES, and wealth disparities between regions.

Additionally, economic estimates vary significantly depending on the number of services considered. For instance, in India's coastal areas, values ranged between US\$ 177 ha⁻¹ (Das and Crépin, 2013) and US\$ 232 ha⁻¹ (Hussain and Badola, 2008). Instead, much higher values from US\$ 10,158 to US\$ 12,392 ha⁻¹ are reported for Thailand (Barbier, 2007). Fishery and storm protection are among the most frequently valued services, reflecting the importance of these two services provided by mangroves. Table 14 shows the estimates from recent studies on the economic value of mangroves.

Table 14: Global economic values of mangroves (US\$ ha⁻¹ yr⁻¹)

Provision services	Regulating services	Cultural services	Habitat services	TEV	Price level	Source
2 998	171,515	2,193	17,138	193,845	2007 (Int. \$)	Groot <i>et al.</i> , 2012
—	—	—	—	193,843	2007 (Int. \$)	Kubiszewski <i>et al.</i> , 2017
28	238	492	—	758	2015 (€)	Magalhães Filho <i>et al.</i> 2021
3,000	232					Hussain and Badola 2008
	10,158–12,392					Quoc Vo, Kuenzer and Oppelt 2015
	177					Barbier 2007
	4,185				2007	Das and Crépin 2013
	822				2010	M. Brander <i>et al.</i> 2012
12,351–13,580	5,538–6,466	2,047–5,790		20,844–33,258	2019	Stewart-Sinclair <i>et al.</i> , 2021
81–3,596	3.4–12,500					Su, Friess and Gasparatos 2021
						Eric <i>et al.</i> , 2022

4.2.2.4 Corals

Globally, coral reefs avert substantial flood damages and provide significant annual expected benefits for coastal protection. However, the protection offered by reefs is higher for lower intensity and frequent storms and lower – yet still substantial – for more extreme events. According to Costanza *et al.* (2014), the estimated global value of coral reefs varied from around US\$ 8,000 to US\$ 352,000 ha⁻¹ yr⁻¹, including benefits from storm protection, erosion protection, and recreation.

More recently, Beck *et al.* (2018) reported that the 71,000 km of reef coastlines worldwide reduce the annual expected storm damages by more than US\$ 4 billion. Without reefs, annual damages would more than double (+118%), and land flooding would increase by 69 percent – affecting 81 percent more people annually.

Once again, flood protection benefits from reefs can be compared in absolute terms (i.e. total dollar value of national avoided losses) or relative to the national GDP and population (i.e. total dollar value of national avoided losses/GDP). According to Beck *et al.* (2018), reefs are essential for averting damages and losses in Indonesia, the Philippines and Malaysia. Meanwhile, the annual averted damages for the Dominican Republic (ranked 7th) and Belize (ranked 15th) amount to US\$ 96 million and US\$ 9 million, respectively. When considering GDP levels, reefs are relevant to SIDS in the Caribbean and the South Pacific. Cayman Islands, Belize and Grenada have more than 30 percent of their GDP protected by coastal reefs. Meanwhile, Dominican Republic (ranked 9th) and Solomon Islands (ranked 15th) get respectively 11 percent and 4 percent of their GDP protected by reefs (*ibid.*).

Another primary benefit attributable to coastal ecosystems comes from the national revenues produced by domestic and international tourism. Spalding *et al.* (2017) calculated reef tourism to be worth US\$ 35.8 billion globally every year – equivalent to the travelling and spending of 70 million tourists. Based on the above figure, they estimated the annual mean value of coral reefs of US\$ 482,428 km⁻² yr⁻¹ (median US\$ 48,000 km⁻² yr⁻¹). These estimates are much lower than those provided by Groot *et al.* (2012), where coral reef value reached US\$ 9,630,200 km⁻² yr⁻¹ (median value US\$ 156,200 km⁻² yr⁻¹). This discrepancy in estimates can be attributed to the more conservative approach used in recent studies and an increasing availability of findings at the national level. Table 15 summarises the global estimates of coral reefs from some studies.

Table 15: Global mean economic values of coral reefs (US\$ ha⁻¹ yr⁻¹)

Provision services	Regulating services	Cultural services	Habitat services	TEV	Price level	Source
55,724	171,478	108,837	16,210	352,249	2007 (Int. \$)	Groot <i>et al.</i> , 2012
—	—	—	—	352,915	2007 (Int. \$)	Groot <i>et al.</i> , 2013
3	390	1,520	—	1,913	2015 (€)	Magalhães Filho <i>et al.</i> , 2021
	1,131				2010	Stewart-Sinclair <i>et al.</i> , 2021

Given the heterogeneity in the economic estimates and to better reflect the range of monetary values of coral reefs that could be considered reasonable, we extracted and analysed estimates from several observations (n=26) included in the Ecosystem Services Valuation Database (ESVD). Table 16 shows the summary statistics of our review.

Table 16: International estimates of coral reefs values (Int. US\$ ha⁻¹ yr⁻¹)

n	Min	Median	Mean	Max
24	0.04	79	304	2,198

Source: Own estimates from data included in Foundation for Sustainable Development (2021)

4.2.2.5 Seagrass beds

Very few global economic studies exist on seagrass beds. Globally, the TEV of seagrass beds varies from US\$ 1,912 (Stewart-Sinclair *et al.* 2021) to about US\$ 29,000 (Kubiszewski *et al.*, 2017).

Meanwhile, the value of other coastal ecosystems, such as marshes, dunes and pelagic systems, has been estimated using various methods. For example, Emerton and Kekulandala (2003) estimated the total value of flood control services provided by marshes in Sri Lanka to be US\$ 5 million yr⁻¹. In British Colombia, Batker, Kocian and Molnar (2012) estimated the importance of food provision services in the marine area to be between US\$ 23 and 44 billion in 2004. Table 17 summarises the TEV of seagrass beds, meadows and other coastal ecosystems.

Table 17: Global economic values of coastal seagrass beds and meadows (US\$ ha⁻¹ yr⁻¹)

TEV	Price level	Source
28,916	2007 (Int. \$)	Kubiszewski <i>et al.</i> , 2017
1,912	2010	Stewart-Sinclair <i>et al.</i> 2021

4.2.3 The cost of nature-based solutions

4.2.3.1 Literature review on the cost of ecosystem restoration

In the following sections, we report the costs from studies that analysed the restoration of coastal ecosystems, mangroves, coral reefs (artificial or natural), seagrass and meadows. However, not all studies provided cost data consistently or comprehensively. For instance, data on capital and operating costs or the cost breakdown by project activity type (e.g. planning, construction, maintenance) were often missing. Thus, we reported the estimates as provided by the source.

4.2.3.2 Restoring coastal ecosystems

According to Groot *et al.* (2013), the restoration cost of coastal ecosystems ranges between US\$ 250,000 – 500,000 ha⁻¹ (2007 US\$ values). Seemingly, restoring coastal marine ecosystems is 10 – 400 times more expensive than other ecosystems, as shown in Table 18.

Table 18: Maximum restoration costs per type of ecosystem (2007 US\$ values)

Ecosystem	Restoration cost (US\$ ha ⁻¹)
Coastal	250,000 – 500,000
Inland wetlands	40,000
Freshwater systems	16,000
Tropical	7,000
Temperate forests	3,000
Grassland	1,500
Woodlands	1,000

Source: Data from Groot et al. (2013)

Similar conclusions have been reached in other cost analyses. In reviewing over 235 studies, Bayraktarov *et al.* (2016) found that the overall reported median and average costs for restoration of coastal marine habitat were around US\$ 80,000 ha⁻¹ and US\$ 1,600,000 ha⁻¹ (2010 US\$ values), respectively. However – when properly accounting for capital and operating costs – the actual total

costs are likely to be two to four times higher, increasing the median costs to US\$ 150,000 – 400,000 ha⁻¹.

Meanwhile, Armsworth (2014) estimated that conserving terrestrial protected areas costs between US\$ 10,000 – 50,000 ha⁻¹ (2012 US\$ values) – eight times less than restoring coastal marine ecosystems. However, when accounting for the local values of the U.S. dollar in developing countries, the costs for restoration would be up to 200 times less than in developed countries (Bayraktarov *et al.*, 2016).

4.2.3.3 Restoring mangroves

Several reviews indicated that restoring coastal environments by reintroducing mangrove forests is among the most cost-effective solutions (Aerts, 2018; EDF, 2015; Narayan *et al.*, 2016; Ruangpan *et al.*, 2020). In a recent review and meta-analysis of 167 studies, Su, Friess and Gasparatos (2021) show that – by excluding extreme values – the total mangrove restoration costs range from US\$ 23 to US\$ 371,326 ha⁻¹ with a median value of US\$ 1,097 ha⁻¹ (2019 US\$ values). The restoration technique is one of the factors affecting the final cost. In fact, the minimum restoration cost (i.e. US\$ 23 ha⁻¹) was reported in a project using the direct propagule dibbling method, which is different from conventional seedling transplantation.

Similarly, estimates from Bayraktarov *et al.* (2016) report that the most cost-effective technique reported from a mangrove restoration project was the planting of seeds, seedlings, or propagules (US\$ 786 ha⁻¹ in a developing country), while the least cost-effective project used a combination of planting and construction (estimated at US\$ 749,215 ha⁻¹ in a developed country). The median total restoration costs for mangroves are equal to US\$ 3,000 ha⁻¹ (2010 US\$ values), which is well below the maximum restoration cost of other ecosystems (for a comparison, see Table 18).

However, according to Adame *et al.* (2015), the cost-effectiveness of mangrove restoration actions depends highly on the compatibility of biodiversity and carbon targets, as well as the suitability of restoration areas to support these objectives. In addition, the lower cost of mangrove restoration can be related to a) lower labour costs – given the high number of community and volunteer workers involved; b) the high availability of mangrove seeds, seedlings, and propagules, and c) the easier access to mangrove restoration sites (*ibid.*). Table 19 summarises the range of possible restoration costs extracted from the literature.

Table 19: Restoration cost of mangrove forests (US\$ ha⁻¹)

Min	Median	Max	Price level	Source
23	1,097	371,326	2019	Su, Friess and Gasparatos, 2021
786	3,000	749,215	2010	Bayraktarov <i>et al.</i> , 2019
69	1,221	141,303	2010	Stewart-Sinclair <i>et al.</i> 2021

4.2.3.4 Restoring coral reefs

The cost of constructing or rehabilitating coral reefs can vary substantially depending on the restoration techniques. In a study by Bayraktarov *et al.* (2016), the variability in cost estimates between the different restoration techniques spanned five orders of magnitude – from US\$ 6,000 to US\$ 143 million ha⁻¹ (in 2010 US\$ values) – with a median cost of US\$ 404,147 (SD: ±US\$ 18.9 million ha⁻¹).

Ferrario *et al.* (2014) indicate that restoring coral reefs is much cheaper than constructing or adding a substrate. In their review, the cheapest restoration technique was the collection and nursery phase component of the coral gardening approach, with a median cost of US\$ 28,000 (2012 US\$ values; SD: ±US\$ 20,000 ha⁻¹). Similarly, data included in a review from Stewart-Sinclair *et al.* (2021) reports that coral gardening costs an average of US\$ 30,000 ha⁻¹ (2010 US\$ values). On a similar range, Bayraktarov *et al.* (2016) found that the most cost-effective coral reef restoration technique concerned transplanting coral reef fragments from donor colonies to degraded reefs with costs averaging US\$ 11,717 ha⁻¹ (2010 US\$ values) in developing countries.

Meanwhile, construction costs of structural coral reefs are much higher. Data from restoration projects put it between US\$ 20 and US\$ 155,000 (2012 US\$ values) per linear metre (m⁻¹), with a median project cost of US\$ 1,290 m⁻¹ (Ferrario *et al.*, 2014). The median cost for substrate addition and artificial reefs was the highest, with a median cost of US\$ 3.3 million ha⁻¹ (SD: ±US\$ 44 million ha⁻¹) (*ibid.*).

In contrast, the least cost-effective approach concerned coral reef restoration through a combination of stabilising substrate and transplantation of fragments. The cost reached US\$ 2.8

million ha⁻¹ in a developing country (Bayraktarov *et al.*, 2016). Table 20 summarises the cost data of coral reef restorations reported in various studies.

Table 20: Summary results of cost review on coral reef restoration (US\$)

Interventions	Min	Median	Max	Unit	Price level	Source
Overall	6,000	404,147	143,000,000	ha ⁻¹	2010	Bayraktarov <i>et al.</i> , 2016
	103	117,220	142,667,803	ha ⁻¹	2010	Stewart-Sinclair <i>et al.</i> , 2021
Construction	20	1,290	155,000	m ⁻¹	2012	Ferrario <i>et al.</i> (2014)
		3,300,000		ha ⁻¹	2012	Ferrario <i>et al.</i> , 2014
		0		ha ⁻¹	2010	Bayraktarov <i>et al.</i> , 2016
		2,879,773		ha ⁻¹	2012	Ferrario <i>et al.</i> , 2014
Restoration		28,000		ha ⁻¹	2012	Ferrario <i>et al.</i> , 2014
		11,717		ha ⁻¹	2012	Bayraktarov <i>et al.</i> , 2016

4.2.3.5 Restoring seagrass beds

Few studies have looked into the cost and benefits of seagrass restoration. Among the few available evidence, the review of 64 studies on seagrass beds by Bayraktarov *et al.* (2016) reports that the most cost-effective seagrass restoration technique concerned transplanting seagrass cores or plugs and required US\$ 29,749 ha⁻¹, in a developed country. Opposite, the least cost-effective project used mechanical seagrass transplantation and required about US\$ 1.2 million ha⁻¹ in a developed country (*ibid.*). Table 21 summarises the median and mean costs for seagrass bed restoration.

Table 21: Restoration cost of seagrass beds (International 2010 US\$ ha⁻¹)

Intervention	Median	Mean	Source
Restoration	144,981	444,604	Bayraktarov <i>et al.</i> , 2016
Total restoration*	419,646	725,326	
Restoration	74,594	257,980	Stewart-Sinclair <i>et al.</i> , 2021

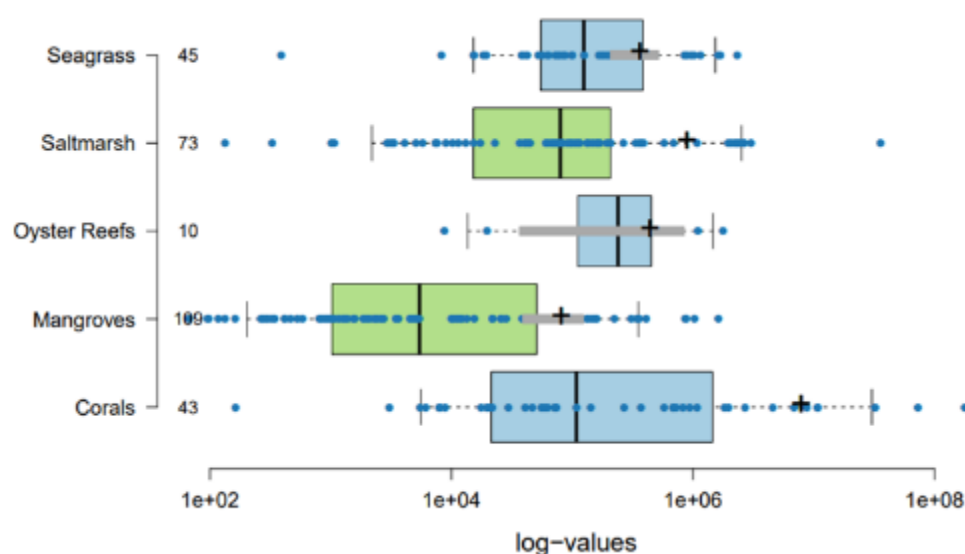
*: From observations including both capital and operating costs
Source: From data included in Bayraktarov *et al.* (2016)

4.2.3.6 Cost comparison of green solutions and grey infrastructures

The conservation of coastal communities and ecosystems can be carried out through investments in NbS or green infrastructures. Such investments reduce the impact of coastal hazards (e.g. floods, coastal erosion, habitat degradation and biodiversity depletion) and provide socio-economic and environmental benefits (Cambridge Econometrics, 2020; Glick *et al.*, 2020; Hynes *et al.*, 2022; Ruangpan *et al.*, 2020; Ruckelshaus *et al.*, 2016). Various data and information were retrieved from multiple studies to estimate the cost of investing in NbS (Aerts, 2018; Bayraktarov *et al.*, 2016; Reguero *et al.*, 2018; Resilient Islands, 2019a; Rogers *et al.*, 2019; Stewart-Sinclair *et al.*, 2021). Collecting a large dataset allowed us to conduct a systematic review and establish average restoration values for each ecosystem under analysis.

Our review showed the presence of considerable heterogeneity and variance in the findings due to the different timeframes, years and locations where these investments took place. All estimates were converted to 2021 US dollars to reduce data heterogeneity and adjusted for income differences through the GDP (PPP) conversion factors (The World Bank, 2023). These figures indicate the average cost value per hectare for the restoration of each ecosystem. However, due to the inclusion of studies from developed and developing countries, the significant variability within the dataset does not allow us to establish an average standard cost per hectare applicable in all contexts. To identify it, a more detailed analysis of a subset of the larger sample was performed to better capture the reality of the study context. Table 22 summarises the cost sample's descriptive statistics, while Figure 13 shows the variability within the dataset.

Figure 13: Costs sample boxplot (2021 Int.US\$ log-values)



Source: Own elaboration from data in Table 22

Note: The + sign indicates the sample mean, while the grey bar reports the 95 percent confidence interval of the mean

Table 22: Summary statistics of ecosystem restoration costs (2021 Int.US\$ ha⁻¹PPP)

	Country	n	Min	Mean	Max
Coral	Developing	25	22	43,905	207,904
	Developed	18	1,889	18,330,175	177,700,952
Mangroves	Developing	43	1	1,286	30,594
	Developed	46	133	115,115	876,839
Oyster reefs	Developing	-	-	-	-
	Developed	10	8,749	439,928	1,764,404
Saltmarsh	Developing	-	-	-	-
	Developed	67	133	804,368	24,854,000
Seagrass	Developing	-	-	-	-
	Developed	49	389	367,250	2,319,289

Source: Own estimates from data included in Bayraktarov et al. (2016)

Similarly, our analysis gathered information on investment, operation and maintenance (O&M) costs of both green and grey infrastructures that are set to protect coastal zones from the effect of climate hazards and impact (e.g. sea surges, floods, storms, etc.). Green infrastructures include investments for the conservation and restoration of natural environments obtained through – for example – the restoration of mangrove forests and coral reefs or the construction of oyster

reefs, marshland creation and restoration. Among the several investments considered, grey infrastructures conceive investments for flood protection (e.g. rural, earthen and sea dikes, sandbag or concrete walls etc.), coastal protection (e.g. beaches and dunes restoration, coastal groins etc.), water channel management (e.g. bank dredging, river widening etc.), flood proofing and prevention (e.g. pumping station, infiltration trench, concrete storage tanks, green roofs, relocation and elevation of buildings, basements waterproofing etc.). Table 23 reports the average cost value (for both investment and O&M) divided into two subsets of low and high-range values (i.e. min and max).

Table 23: Average investment and O&M value of green and grey infrastructures (2021 US\$)

Type	Investment cost		O&M cost	
	Min average	Max average	Min average	Max average
Green	1,389,765	1,392,168	133,242	133,537
Grey	56,840,164	123,488,794	271,832	1,263,838
No. of observations				
Total	103	103	76	76
of which Green	32	32	32	32
of which Grey	71	71	44	44

Source: Own estimates from data included in Aerts (2018) and Bayraktarov et al. (2016)

Table 23 shows how the investment and O&M cost of grey infrastructure vastly exceeds the cost of green infrastructure (Table 22). Nature-based coastal protection infrastructures are two to one hundred times cheaper than artificial infrastructures. This is further observed by Hynes et al. (2022), who indicate that the cost of constructing an oyster reef on a 1,076-mt-long shore amounts to about EUR 260,000 (US\$ 277,000).¹⁰ Labour and small equipment for monitoring and maintaining the reef cost EUR 12,640 (US\$ 13,466) each year. In comparison, a grey infrastructure of rock armour, impermeable revetments and seawalls covering the same area would cost between EUR 1.09 – 12.36 million (US\$ 1.16 - 13.16 million) for an average total initial capital cost of EUR 5.57

¹⁰ At the time of this study, the prevailing average exchange rate used to convert Euro to US\$ was 1:1.06

million (US\$ 5.93 million). An additional annual maintenance cost of EUR 68,480 (US\$ 73,000) from year one onward must also be considered.

4.2.4 Methodology

4.2.4.1 The cost-benefit analysis framework

In conducting the ES evaluation of mangrove forests, coral reefs and seagrass beds, we employ the cost-benefit analysis (CBA) method. CBA is an analytical technique used to estimate the cost and benefits associated with an investment to measure the total wealth generated or the impact on social well-being. This method helps evaluate which alternative project options can maximise the result for all stakeholders.

In the CBA analytical framework, the concept of opportunity cost is particularly relevant to determining the approach used to evaluate project resources. The opportunity cost of a good or service is defined as the loss resulting from using that good or service in the project rather than in another best alternative use. The logic of the CBA is that a valuation of project inputs and outputs that is done at their market value may lead to suboptimal and undesirable results, especially from the social standpoint. In fact, market prices do not reflect the actual value of the goods and services because their levels are altered by market inefficiency and failure. The absence of markets is an example of market failure, which mainly affects the environmental services that coastal ecosystems provide. These services are not traded in markets; hence, their valuation is challenged by the lack of a reference value for their evaluation. To overcome this issue, the ecosystems evaluations are based on the direct (e.g. revenues for coral reef tourism) or indirect benefits (i.e. avoided damage thanks to mangrove presence) provided to the people receiving them (for more details, see section 4.2.1.3).

4.2.4.2 Mapping ecosystems through remote sensing

Coastal areas are constantly affected by ocean waves and storms, leading to erosion and flooding. These processes pose a significant threat to human populations, activities, and infrastructure, especially when considering climate change and increasing human presence along coasts. Human activities can worsen these risks and weaken the protective functions of coastal ecosystems during storms. Therefore, it is crucial to understand how biological and geological factors interact in coastal erosion and flooding to plan and protect these vulnerable communities

effectively. The critical role of natural habitats in mediating coastal dynamics requires a nuanced understanding of how environmental changes affect vulnerability to storm-induced erosion and flooding.

Through remote sensing, we verified the presence and extent of coastal ecosystems to support the integrity and strength of our research findings. Our work is underpinned by a robust remote sensing-based mapping approach, a technology that has proven its effectiveness in understanding and protecting coastal ecosystems (Quoc Vo, Kuenzer and Oppelt, 2015). This approach allows us to gather spatial information and quantify the extent and productivity of coastal ecosystems. For mangroves, remote sensing provides critical insights into metrics such as area (hectares), density (trees per hectare), structure (density and basal area), and productivity (Mg C per hectare per year). Mangrove extent and change between 2016 and 2020 are calculated based on the most recent data from the Mangrove Forest Watch project (Bunting et al., 2022). Similarly, monitoring for coral reefs encompasses area (hectares) and the percentage of effectively protected coral reefs. Furthermore, measurements of area (hectares) and productivity (Mg C per hectare per year) are conducted for seagrass beds. The extent of coral reef and seagrass habitat is analysed through a 5m pixel resolution with raw data extrapolated from the Allen Coral Atlas (2022). The appendix showcases detailed maps highlighting the presence of each ecosystem in the countries analysed.

The variables described above are then used to outline the role of each ecosystem in providing essential ecosystem services (provisioning, regulating, cultural, habitat, and the total economic value) and for the economic valuation of the respective ecosystems to the local population and neighbouring communities. Additionally, through it, we can identify geographic hotspots at heightened risk of floods, coastal erosion, and habitat loss due to declines in ecosystem coverage. Overlaying vulnerability data in coastal areas with population density information and coastal settlement locations helps us identify regions where economic losses associated with extreme weather events are most significant.

4.2.4.3 Selecting unit values for the ecosystem benefits

We employed the BT technique to quantify the ecosystem benefits, transferring estimates and average (or median) values from past studies to the evaluation site. Where available, we used

detailed estimates from previous studies in the countries of interest (i.e. Belize, Comoros, Dominican Republic, Guinea-Bissau, Papua New Guinea (PNG), São Tomé and Príncipe, Solomon Islands, Suriname and Vanuatu). If country estimates were unavailable, we chose the most appropriate values from the review of global estimates of ES presented in section 4.2.2.1.

Unit values of ecosystem benefits from one study site were transferred to the policy site after adjusting for differences in inflation and living standards across countries, consistently to the unit value transfer technique described in section 4.2.1.3. Values were adjusted for changes in the Consumer Price Index (CPI) that occurred between the estimation period and the latest year for which CPI values are available. If CPI values were unavailable, we retained the original estimate as indicated in the study. We also adjusted these values for GDP conversion factors and the country purchasing power parity (PPP) to account for differences in income level and living standards across countries. The GDP (PPP) indicator accounts for differences in the general price level of countries (The World Bank, 2023).

4.2.4.4 Defining the costs for green and grey infrastructures

The cost of NBS and green and grey infrastructures are presented in section 4.2.3.6. This analysis does not allow in-country validation of the ecosystem's health status and the investment needed to preserve it. More detailed in-country diagnostics are required to determine the investment potential accurately.

This study uses the costs from Table 18 to Table 23 to estimate country-level costs. These costs are then compared to the economic value of the ES to determine the suitability of NBS, green and grey investments in some of the countries under review.

4.2.4.5 Profitability results

The project evaluation through CBA allows for the imputation of monetary values to the positive and adverse effects of the investment on social well-being. These values are then discounted and added together to calculate the total net benefit of the project. The project performance is finally measured through indicators such as the Net Present Value (NPV) and the Benefit–Cost Ratio (BCR), thus allowing the comparison of competing options.

The NPV of the investment compares project costs with project revenues. This profitability indicator measures the project's net worth and the ability of the project's net revenues to repay the

initial investment, regardless of the financing sources or methods (European Commission, 2015). The NPV is the sum of the difference between discounted costs and revenues. Thus, the NPV formula would be:

$$NPV = \sum_{i=1}^n \frac{B_i - C_i}{(1+r)^i} - I_0$$

where:

- B_t is the amount of revenues (or benefits) at time i ,
- C_t is the amount of costs borne during the operation phase of the investment at time i ,
- I_0 is the investment cost,
- $(1+r)$ is the discount factor chosen for discounting at time i , where r is the discount rate,
- i is the project economic life.

The NPV is expressed in monetary terms, and its magnitude depends on the size of the project. A positive NPV denotes that the project can compensate fully for the expenses and yield a profit — equal to the NPV. The opposite is true for negative values of the NPV. Thus, projects generating positive NPVs would be approved, and those presenting negative NPVs would be discarded. When comparing two or more mutually exclusive investments, the alternative that maximises the NPV needs to be preferred (World Bank, 2001).

Another profitability indicator is the BCR. The ratio compares the project benefits' present value to the project costs' present value. The ratio must equal or exceed one for the project to be acceptable. The BCR may give an incorrect ranking among independent projects and cannot be used to choose mutually exclusive alternatives (Bojö, Mäler and Unemo, 1990). The BCR formula is:

$$BCR = \frac{\sum_{i=1}^n \frac{B_i}{(1+r)^i}}{\sum_{i=1}^n \frac{C_i}{(1+r)^i}} = \frac{PV(B_i)}{PV(C_i)}$$

where:

- $PV(B_i)$ is the present value of revenues (or benefits) at time i ,

$PV(C_i)$ is the present value of costs at time t .

Each profitability result has been discounted for varying rates of 2, 5 and 10 percent and combined into three cost scenarios (i.e. min, mean and max cost) to determine nine analysis scenarios of high, medium and low profitability. The analysis timeframe is equal to 20 years.

4.2.4.6 Monte Carlo simulation and uncertainty

To account for uncertainty and variability in the cost and benefit estimates, we perform a sensitivity analysis using the Monte Carlo simulation method, whereby probability distribution functions are associated with cost and benefits variables. Probability distributions are fitted to cost variables after removing outliers from the cost sample. Cost variables are related to log-normal distribution or triangular distribution functions. Instead, revenues are fitted with normal or log-normal distribution functions. The NPVs are tested against 10,000 randomised trials, and results are presented for various discount rate scenarios (i.e. 2, 5 and 10 percent).

4.2.4.7 Study limitations

In standardising values, we faced the problem of distinguishing between mean and marginal values, which can be expressed in monetary terms. The majority of studies we reviewed provided mean value estimates per hectare. However, ecosystem changes should be valued using marginal changes. In contrast, mean values can be used to compare an ecosystem's value relative to the area's size (Bergstrom *et al.*, 1990). In addition, expressing ecosystem values in per-hectare terms gives the impression that each hectare is equally productive. Without being able to convert average values to marginal values, we assume a linear provision capacity of each ecosystem, and we overlook possible interactions across multiple biomes. However, coastal ecosystems are highly interactive and complex environments whose health status and service delivery capacity change over time non-linearly (Barbier *et al.*, 2008; Grantham *et al.*, 2022; IPBES, 2019; Koch *et al.*, 2009). In addition, standardised measures of ecosystem value do not reflect that coastal ecosystems and services are valued differently by different people at different times (Grantham, Lau and Kleiber, 2020).

Overall, the economic evaluation of ES grasps only a part of a more significant contribution of natural capital to human welfare, especially in SIDS, where the reliance of vulnerable coastal populations on natural resources is high (Folkersen, 2018). Also, this leads to thinking about whether

ES should be valued in monetary terms, given the multiplicity of services lacking a reference market for their evaluation (Groot *et al.*, 2012). Finally, this analysis was performed using secondary data, whose availability is limited and reliability – for both costs and benefits – uncertain.

4.2.5 Country case studies

4.2.5.1 Belize

Flood protection benefits from mangroves are high and account for over 28 percent of the national GDP in Belize, proving once again the importance of natural barriers in countries with lower GDPs, where exposure is concentrated along vulnerable coastlines (Menéndez *et al.*, 2020). The erosion protection service provided by coastal wetlands was estimated to range between US\$ 231–347 million (Cooper, Burke and Bood, 2009). Other estimates on the avoided damages thanks to the protection service of coastal wetlands amount to US\$ 536 million yr⁻¹ and to 5,301 lives saved annually (Costanza *et al.*, 2021).

Additionally, economic services related to tourism and environmental regulation are particularly relevant. Our estimates indicate mangroves' carbon sequestration amounting to US\$ 2,403 ha⁻¹ yr⁻¹. This is followed by tourism – mainly associated with coral reefs – and the coastal protection of mangroves, coral reefs and seagrass beds. Meanwhile, the monetary contribution to subsistence fishing of the entire coastal ecosystem is worth US\$ 321 ha⁻¹ yr⁻¹.

Table 24 summarises each ecosystem's relative annual contribution per hectare and its aggregate sum (i.e. coastal ecosystem). All values from the indicated sources were adjusted for the country CPI index and reported into 2022 US\$ values.¹¹

¹¹ <https://tradingeconomics.com/belize/consumer-price-index-cpi>

Table 24: Ecosystem services values in Belize (2022 US\$ ha⁻¹ yr⁻¹)

Sector	Ecosystem service	Coastal ecosystem	Mangroves	Coral reefs	Seagrass beds
Fisheries	Subsistence fishing	321	102	78	140
	Commercial fishing	161	51	39	70
Tourism	Tourism and recreation	1,269	380	888	-
Regulating services	Coastal protection	2,137	681	520	936
	Carbon sequestration	3,433	2,403	343	686
Total		7,321	3,619	1,869	1,833

Source: Own estimates from data by Arkema et al. (2015), Cisneros-Montemayor et al. (2013) and Cooper, Burke and Bood (2009)

The restoration cost of coastal ecosystems varies significantly between developing and developed countries. Table 25 summarises cost estimates for restoring or maintaining coastal ecosystems in Belize. In some cases (e.g. Oyster reefs and seagrass), the data used for the calculation were drawn from projects implemented in developed countries. However, the original estimates get substantially reduced when adjusting for the differences in income and PPP levels in developing countries.

Table 25: Restoration cost by type of coastal ecosystem in Belize (2021 Int.US\$ ha⁻¹)

	Mangroves	Corals	Oyster reefs*	Salt marshes*	Seagrasses*
Min cost	40	13	5,249	103	80
Average cost	2,467	26,343	263,957	690,888	534,747
Max cost	46,766	124,742	1,058,642	27,744,000	21,473,856

Source: Own estimates from data extracted from Bayraktarov et al. (2016)

*: These estimates are sourced from studies in developed countries and adjusted to Belize's PPP level

The results of the preliminary profitability analysis for mangroves, coral reefs and seagrass beds are summarised in table 26- table 28. Nine scenarios are considered for the calculation of the

BCR and the NPV. The scenarios account for changes in restoration costs and discount rates to determine the present values of each ecosystem. The analysis timeframe is 20 years.

For mangroves, the analysis shows that the conservation and restoration are profitable under most circumstances, except when restoration costs exceed US\$ 30,866 ha⁻¹ (in the scenario foreseeing a 10 percent discount rate). In the latter case, the NPV becomes negative, and the cost outweighs the benefits (i.e., the BCR is lower than one).

Differently, restoring coral reefs and seagrass beds depends highly on the conservation work required and the health status of the ecosystem. Especially for seagrass beds, average restoration costs are high, leading to uneconomic results. To break even, restoration costs in Belize should not exceed US\$ 15,938 and US\$ 15,608 ha⁻¹ for coral reefs and seagrass beds, respectively.

Table 26: Profitability of mangrove restoration in Belize (2021 US\$ ha⁻¹)

	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	1,472	1,122	766	COST	min	59,136	45,060	30,770
	average	24	18	12		average	56,709	42,633	28,343
	max	1.27	0.96	0.66		max	12,409	-1,666	-15,956

Table 27: Profitability of coral reef restoration in Belize (2021 US\$ ha⁻¹)

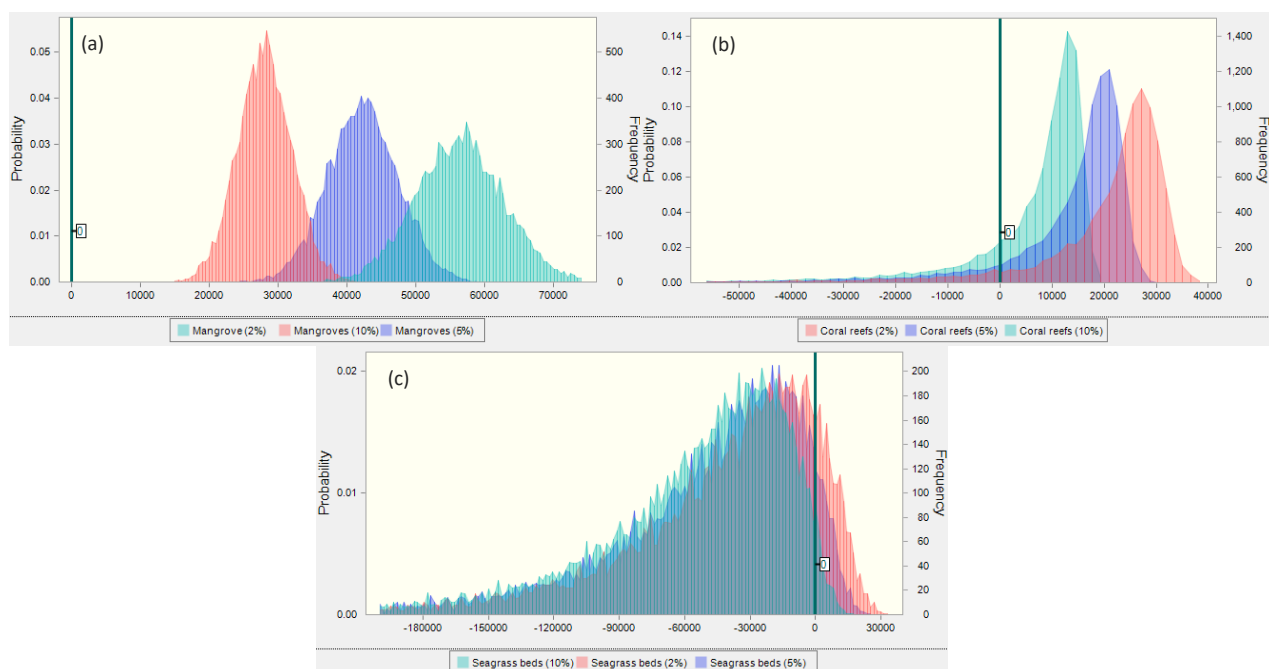
	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	2,315	1,765	1,206	COST	min	30,551	23,281	15,901
	average	1.16	0.88	0.60		average	4,221	-3,048	-10,429
	max	0.25	0.19	0.13		max	-94,178	-101,448	-108,829

Table 28: Profitability of seagrass bed restoration in Belize (2021 US\$ ha⁻¹)

	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	128	97	66	COST	min	29,751	22,619	15,378
	average	0.15	0.11	0.08		average	-175,379	-182,511	-189,751
	max	0.02	0.02	0.01		max	-1,361,589	-1,368,721	-1,375,962

The sensitivity analysis results provided in Figure 14 show that, even after accounting for variations in average restoration costs and expected benefits, mangrove NPVs remain steadily in the positive result region of the graph (i.e. to the right-hand side on the zero-line threshold). Similarly, coral reef NPVs are also largely positive, only slightly crossing into negative results on a few unlikely occasions. Instead, seagrass bed results are expected to remain negative, and only in fewer instances will they turn positive – shall benefits surge concurrently with reduced restoration costs.

Figure 14: Sensitivity analysis results for mangroves (a), coral reefs (b) and seagrass beds (c) NPV (US\$) in Belize



4.2.5.2 Comoros

A detailed study on the marine ecosystem of Comoros revealed that the annual total economic value of coastal ecosystem goods and services equates to 22 percent of the country's GDP (ASCLME, 2012). The diagnostic quantified the TEV of the marine and coastal ecosystem to about US\$ 15 billion, with direct and indirect use values amounting to about 56 and 44 percent, respectively. According to these estimates, regulating services are more significant than provisioning and cultural services. However, both tourism and fisheries are critical sectors for the direct and indirect use value they provide. Coral reefs alone produce 3,000 tons of reef fish annually, with a 2 billion Comorian francs turnover yearly (ibid.).

Much lower estimates are instead provided by other studies. In their evaluation of the marine ES values in the Northern Mozambique Channel, Ghermandi *et al.* (2019) estimated the TEV of the Comoros marine ecosystem to be US\$ 237 million yr⁻¹. Most of the value relates to carbon sequestration services provided by seagrass (US\$ 150 million) and tourism (US\$ 79 million). Due to their limited extent, mangroves and wetlands are irrelevant from an economic viewpoint (*ibid.*). Similarly, a study from UNEP/Nairobi Convention Secretariat (2009) estimated the TEV of Marine ecosystems totalling US\$ 700 million, with coral reefs, coastal forests and mangroves valued at US\$ 436 million, US\$ 261 million and US\$ 1 million, respectively. In that study, the economic value of seagrass beds for Comoros is not quantified. Meanwhile, Costanza *et al.* (2021) valued the protection service provided by coastal wetlands at US\$ 1.6 million yr⁻¹.

On a broader analysis of the African Marine Ecosystem value, Trégarot *et al.* (2020) estimated the total value of the Agulhas Large Marine ecosystem – including Comoros – equal to US\$ 305,408 ha⁻¹ yr⁻¹, with US\$ 242,543 ha⁻¹ yr⁻¹ for coral reefs, US\$ 32,491 ha⁻¹ yr⁻¹ for mangroves, and US\$ 30,345 ha⁻¹ yr⁻¹ for seagrass beds. Table 29 shows the estimates of Comoros Marine ecosystem values, adjusted for the country's CPI and PPP levels.

Table 29: Ecosystem services values in Comoros (2021 US\$ ha⁻¹ yr⁻¹)

Sector	Ecosystem service	Coastal ecosystem	Mangroves	Coral reefs	Seagrass beds
Fisheries	Subsistence fishing	-	-	-	-
	Commercial fishing	494	4.74	337	151
Tourism	Tourism and recreation	7,557	2,267	5,290	-
Regulating services	Coastal protection	20	0.20	13	6.27
	Carbon sequestration	14,021	-	-	14,021 [‡]
	Total	22,093	2,272	5,641	14,179

[‡]: This value was calculated from estimates by Ghermandi *et al.* (2019). However, we remark that estimated seagrass bed values in Comoros are much higher than those obtained in other countries.

Source: Own estimates from data and information in Ghermandi *et al.* (2019), ASCLME (2012) and UNEP/Nairobi Convention Secretariat (2009)

Similarly, Table 30 shows the cost of restoring or maintaining the various coastal ecosystems in Comoros after accounting for the differences in income and PPP levels in the country.

Table 30: Restoration cost by type of coastal ecosystem in Comoros (2021 Int.US\$ ha⁻¹)

	Mangroves	Corals	Oyster reefs*	Salt marshes*	Seagrasses*
Min cost	27	9	3,500	53	156
Average cost	1,645	17,562	175,971	356,498	136,909
Max cost	31,178	83,162	705,762	14,315,904	927,716

Source: Own estimates from data by Bayraktarov et al. (2016)

*: For the following ecosystems, estimates are sourced from studies in developed countries and adjusted to Comoros PPP level

The results of the preliminary profitability analysis on the preservation of mangroves, coral reefs and seagrass beds are summarised in Table 31 – Table 33.

The analysis shows that the conservation is profitable under most circumstances for mangroves, except when restoration costs exceed US\$ 19,344 ha⁻¹ (at a 10 percent discount rate). Beyond that level, the NPV becomes negative.

Differently, restoring coral reefs and seagrass beds is more sensitive to cost levels. Especially for seagrass beds, average restoration costs are high, which could lead to uneconomic results. The cost break-even point for coral reefs and seagrass bed restoration is US\$ 48,031 and US\$ 120,716 ha⁻¹, respectively.

Table 31: Profitability of mangrove restoration in Comoros (2021 US\$ ha⁻¹)

		Discount rate					Discount rate		
		2%	5%	10%			2%	5%	10%
COST	Min	1,386	1,056	721	COST	min	37,125	28,289	19,317
	average	22	17	11		average	35,507	26,671	17,699
	Max	1.19	0.91	0.62		max	5,974	-2,862	-11,834

Table 32: Profitability of coral reef restoration in Comoros (2021 US\$ ha⁻¹)

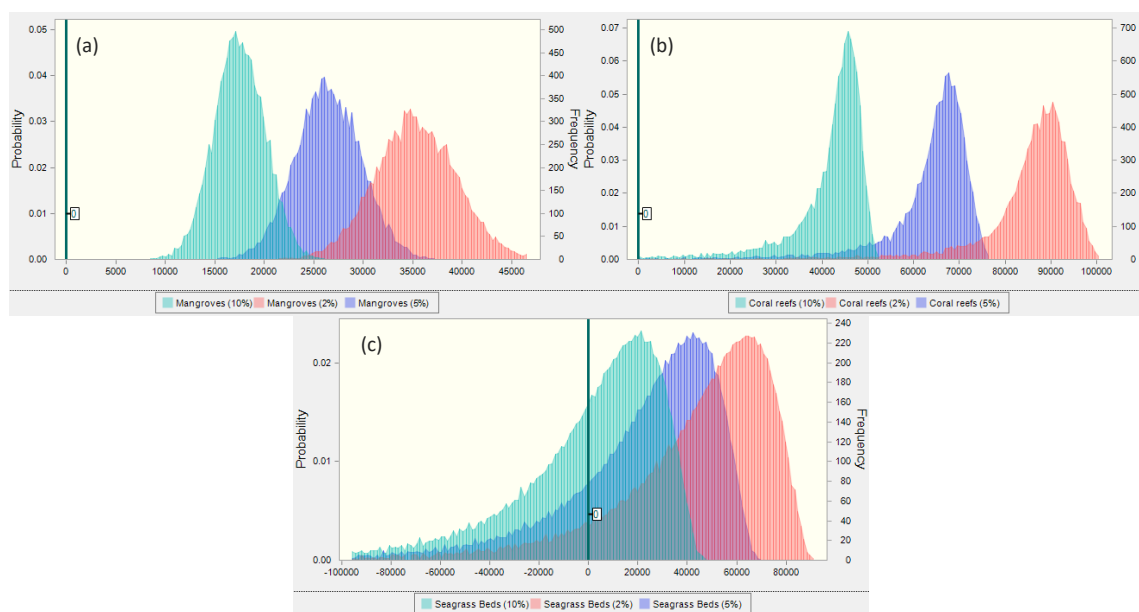
		Discount rate					Discount rate		
		2%	5%	10%			2%	5%	10%
COST	Min	4,193	3,195	2,183	COST	min	92,228	70,286	48,009
	average	2.10	1.60	1.09		average	48,345	26,403	4,126
	Max	0.44	0.34	0.23		max	-115,654	-137,596	-159,873

Table 33: Profitability of seagrass bed restoration in Comoros (2021 US\$ ha⁻¹)

		Discount rate					Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	596	454	310	COST	min	231,462	176,316	120,327
	average	0.68	0.52	0.35		average	-110,421	-165,567	-221,556
	max	0.10	0.08	0.05		max	-2,087,438	-2,142,584	-2,198,573

The graphs in Figure 15 show that NPV results for the three ecosystems in Comoros would remain largely positive under changes in costs and benefits. Seagrass ES results show a slight tendency to fall into the negative results area. However, adverse impacts are less likely as most probability function volumes hang over positive outcomes.

Figure 15: Sensitivity analysis results for mangroves (a), coral reefs (b) and seagrass beds (c) NPVs (US\$) in Comoros



4.2.5.3 Dominican Republic

Marine ES are particularly valuable in the Dominican Republic and are estimated to be about US\$ 690 million (Resilient Islands, 2019b). The avoided damages from coastal wetlands are estimated at US\$ 5 billion yr⁻¹, and 56,389 lives are saved annually (Costanza *et al.*, 2021). Coral reef tourism is the prevailing economic sector, worth about US\$ 525 million annually (Beck *et al.*, 2019; Resilient Islands, 2019b). According to Spalding *et al.* (2017), reefs tourism revenues amount to about 1 percent of the national GDP, bringing a mean value of reef of nearly US\$ 611,000 km⁻² (US\$ 6,110 ha⁻¹). Meanwhile, beach erosion due to decay in reefs' health would result in revenue losses to the resorts of US\$ 52 – US\$ 100 million (Wielgus *et al.*, 2010).

In addition, coral reefs protect the shorelines and the livelihoods of those residing along them. Beck *et al.* (2019) estimated that reefs avert US\$ 96 million from flood damages yearly. Moreover, flood protection is particularly relevant for more than 170,000 vulnerable people (i.e. PPP < US\$ 15 day⁻¹) living along the coastline.

The monetary value of the services provided by the three target ecosystems is summarised in Table 34. In economic terms, the most important service is the cultural service associated with tourism, worth US\$ 590 million yr⁻¹. Instead, the yearly value of commercial and subsistence fishing amounts to US\$ 90 million. The regulating service that the three ecosystems provide amounts to US\$ 105 million.

Table 34: Ecosystem services values in the Dominican Republic (2021 US\$ ha⁻¹ yr⁻¹)

Sector	Ecosystem service	Coastal ecosystem	Mangroves	Coral reefs	Seagrass beds
Fisheries	Subsistence fishing	979	330	415	232
	Commercial fishing	490	165	207	116
Tourism	Tourism and recreation	9,691	2,907	6,783	-
Regulating services	Coastal protection	244	82	103	58
	Carbon sequestration	1,477	1,033	147	295
Total		12,879	4,518	7,658	702

Source: Own estimates from information and data included in *Resilient Islands (2019b)*

The figures above will be considered the annual benefit of preserving or restoring the coastal ecosystem in our preliminary CBA. Meanwhile, Table 35 shows the cost of repairing or maintaining the various coastal ecosystems in the Dominican Republic.

Table 35: Restoration cost by coastal ecosystem in the Dominican Republic (2021 US\$ ha⁻¹)

	Mangroves	Corals	Oyster reefs*	Salt marshes*	Seagrasses*
Min cost	27	9	3,500	133	53
Average cost	1,645	17,562	175,971	891,245	356,498
Max cost	31,178	83,162	705,762	35,789,760	14,315,904

Source: Own estimates from data by Bayraktarov et al. (2016)

*: For the following ecosystems, the estimates are sourced from studies in developed countries only

The following tables (table 36 -table 38) present the NPV and BCR for conserving mangroves, coral reefs and seagrass beds. The costs considered for the calculations are those given in Table 35, while the benefits are those provided in Table 34. The results from the analysis show that preserving mangroves and coral reefs is viable under most analysis scenarios. In contrast, seagrass beds results

are penalised by the high restoration costs. The cost break-even point for the restoration of mangroves, coral reefs and seagrass beds is equal to US\$ 38,470, US\$ 65,240 and US\$ 5,980 ha⁻¹.

Table 36: Profitability of mangroves restoration in the Dominican Republic (2021 US\$ ha⁻¹)

	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	2,756	2,101	1,435	COST	min	73,860	56,286	38,443
	average	44	34	23		average	72,242	54,668	36,825
	max	2.37	1.81	1.23		max	42,710	25,135	7,293

Table 37: Profitability of coral reef restoration in the Dominican Republic (2021 US\$ ha⁻¹)

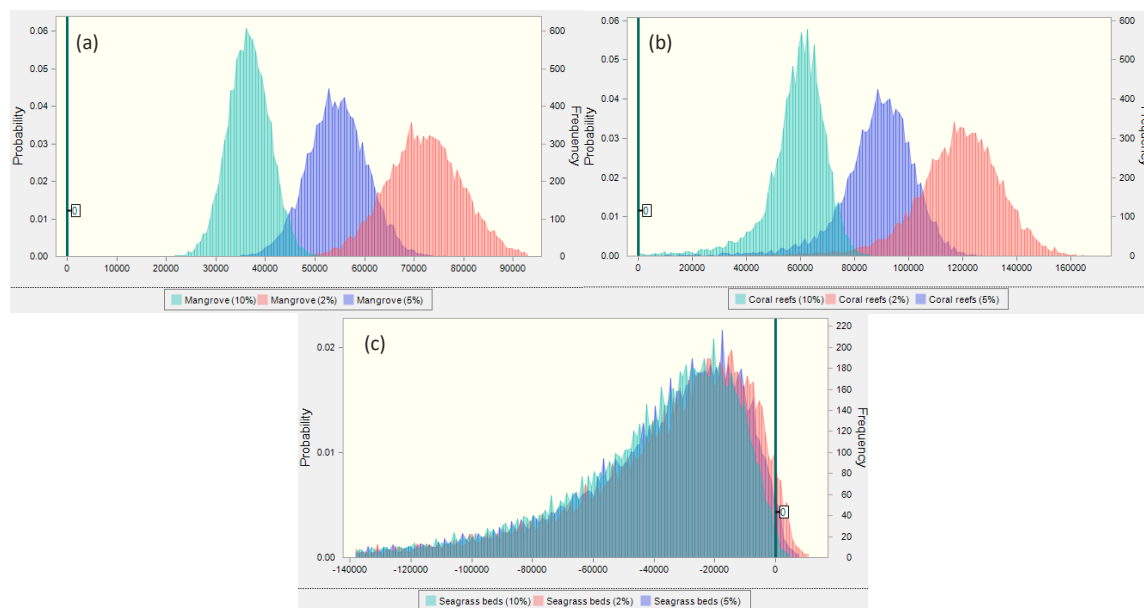
	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	14,231	10,846	7,409	COST	min	125,224	95,437	65,195
	average	7.13	5.43	3.71		average	107,671	77,884	47,642
	max	1.51	1.15	0.78		max	42,071	12,284	-17,958

Table 38: Profitability of seagrass bed restoration in the Dominican Republic (2021 US\$ ha⁻¹)

	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	73	56	38	COST	min	11,330	8,598	5,825
	average	0.08	0.06	0.04		average	-125,423	-128,155	-130,929
	max	0.01	0.01	0.01		max	-916,230	-918,962	-921,735

The sensitivity analysis result in Figure 16 shows that both mangroves and coral reefs NPVs are resilient to changes in costs and benefits. Opposite, the graphs on seagrass beds indicate that the negative results shown in table 38 are confirmed in most cases.

Figure 16: Sensitivity analysis results for mangroves (a), coral reefs (b) and seagrass beds (c) NPVs (US\$) in Dominican Republic



4.2.5.4 Guinea Bissau

Guinea Bissau's natural wealth is estimated at US\$ 3,874 per capita (in 2010 US\$ values), including resources from minerals, fishery, croplands, biodiversity and ecosystems (Edmundson, 2014).

A study in the Canary current large marine ecosystem for West African countries – including Guinea Bissau - indicates that mangroves provide ES worth US\$ 2,235 ha⁻¹ yr⁻¹ mainly for coastal protection, climate regulation and fishery enhancement (Interwies and Görlitz, 2013). In addition, mangroves in Guinea Bissau are found to contribute between 1,700 and 2,600 kg ha⁻¹ to rice production (The State's General Office of the Environment, 2015) and have a potential carbon absorption capacity of 1,143 mgCO₂eq. ha⁻¹ (Vasconcelos *et al.*, 2015).

Meanwhile, seagrass bed services are valued at about US\$ 450 ha⁻¹ yr⁻¹ (Interwies and Görlitz, 2013). This value falls within US\$ 401 – 726 ha⁻¹ yr⁻¹ as estimated in another recent review on seagrass ES in seven countries of West Africa, including Guinea Bissau (ResilienSEA, 2019).

On a broader scale, the analysis from Trégarot *et al.* (2020) estimated the marine ES in the Canary and Guinea Large Marine ecosystem – including Guinea Bissau – ranging between US\$ 18,017 – 30,282 ha⁻¹ yr⁻¹ for mangroves and US\$ 19,351 – 45,379 ha⁻¹ yr⁻¹ for seagrass beds.

In performing the CBA, we compiled data from the above-cited studies. The main figure used is the per capita estimate of ecosystem services value provided by Edmundson (2014). This figure was adjusted for the latest available CPI value (i.e. 2019) and PPP levels. Additionally, the benefits are attributed exclusively to the country's coastal population living below five meters of elevation – about 337,000 people (Neumann *et al.*, 2015).

In the case of Guinea-Bissau, only the mangroves' results are reported due to the lack of data on the geographical extension of coral reefs and seagrass beds. The yearly monetary value of mangroves is US\$ 1,061 ha⁻¹, as shown in Table 28. Commercial fishing is the prevalent service from an economic point of view. The net annual value of fishing amounts to US\$ 109 million and US\$ 47 million for commercial and subsistence fishing, respectively. The value of tourism amounts to US 36 million annually, while the annual value regulating service for the coastal protection and carbon sequestration provided by mangroves sums up to US\$ 106 million.

Table 39: Ecosystem services values in Guinea Bissau (2019 US\$ ha⁻¹yr⁻¹)

Sector	Ecosystem service	Mangroves
Fisheries	Subsistence fishing	168
	Commercial fishing	388
Tourism	Tourism and recreation	127
Regulating services	Coastal protection	311
	Carbon sequestration	68
Total		1,061

Source: Own estimates from data included in Edmundson (2014)

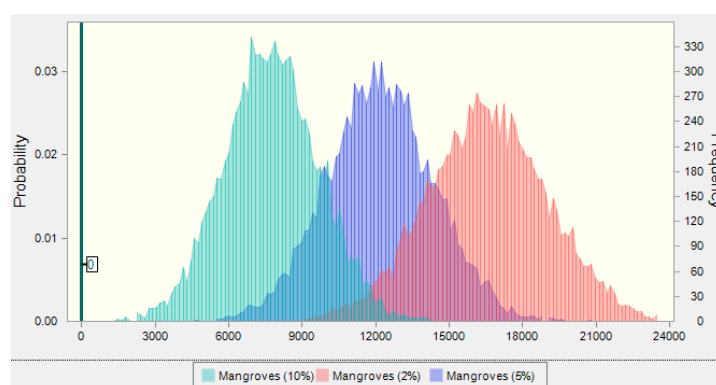
The cost of replanting one hectare of mangroves in Guinea Bissau averages US\$ 1,645 (min US\$ 27 and max. US\$ 31,178). The profitability analysis is presented in Table 40 and indicates an NPV (discounted at 5%) of US\$ 11,582 ha⁻¹ over 20 years. The break-even point is reached at restoration costs of US\$ 9,604 ha⁻¹. Accounting for the 280,200 hectares of mangrove in the country, the total annual worth equates to US\$ 162 million.

Table 40: Profitability of mangroves restoration in Guinea Bissau (2021 US\$ ha⁻¹)

BCR		Discount rate		
		2%	5%	10%
COST	min	647	493	337
	average	10.55	8.04	5.49
	max	0.56	0.42	0.29

NPV		Discount rate		
		2%	5%	10%
COST	min	17,327	13,200	9,009
	average	15,709	11,582	7,391
	max	-13,824	-17,951	-22,142

Figure 17 reports the sensitivity analysis on mangroves. The results from the ecosystem restoration remain positive under all scenarios envisioned. In other words, the negative results presented in Table 40 are very unlikely.

Figure 17: Sensitivity analysis results for mangroves NPVs (US\$) in Guinea Bissau

4.2.5.5 Papua New Guinea

According to Seidel and Lal (2010), the share of the blue economy in PNG is equal to 6.6 percent of the GDP.¹² Nature-based tourism is a relevant economic sector with great potential for further development. With an average spending of US\$ 1,208 per visitor, PNG could generate US\$ 99 million – US\$ 124 million in this sector. Other estimates from Spalding *et al.* (2017) report reef tourism revenues to nearly 0.2 percent of the national GDP, thus bringing a mean value of reef of US\$ 2,203 km⁻² (US\$ 22 ha⁻¹). More recent estimates from Chaiechi *et al.* (2022) estimated reef values by assigning US\$ 6.65 ha⁻¹ for the WTP for recreational fishing in PNG reefs. Considering the

¹² Blue Economy defined by the World Bank (2017) as the joint revenues from fisheries, tourism, maritime transport, aquaculture, off-shore renewable energy, seabed extractive activities, marine biotechnology and bioprospecting.

1.3 million hectares of PNG coral reefs, the option value of recreational fishing is estimated to be around US\$ 9 million.

Similarly, the direct and indirect use value of mangroves has been estimated. The indirect use-value of air quality regulated by the 458,600 hectares of mangroves in the coastal wetlands – valued at US\$ 3,669 ha⁻¹ (2020 US\$ prices) – amounted to around US\$ 1.6 billion (ibid.). Meanwhile, the indirect use-value of erosion prevention services in PNG coastal wetlands was estimated at US\$ 314 million (ibid.).

Table 41 reports our estimates that originated based on the findings from other evaluations as cited below and adjusted to the 2021 price level.

Table 41: Ecosystem services values in PNG (2021 US\$ ha⁻¹ yr⁻¹)

Sector	Ecosystem service	Coastal ecosystem	Mangroves	Coral reefs	Seagrass beds
Fisheries	Subsistence fishing	528	327	132	67
	Commercial fishing	276	171	69	35
Tourism	Tourism and recreation	392	117	274	-
Regulating services	Natural purification	208	145	20	41
	Coastal protection	401	248	101	51
	Carbon sequestration	175	122	17	34
	Total	1,980	1,132	616	231

Source: Own calculation from data included in Chaiechi et al. (2022), Spalding et al. (2017) and Burke et al. (2012)

Table 42 shows the cost of restoring or maintaining the various coastal ecosystems in PNG after adjusting for the country's prevailing PPP and CPI rates.

Table 42: Restoration cost by type of coastal ecosystem in PNG (2021 Int.US\$ ha⁻¹)

	Mangroves	Corals	Oyster reefs*	Salt marshes*	Seagrasses*
Min cost	27	9	3,500	53	156
Average cost	1,645	17,562	175,971	356,498	136,909
Max cost	31,178	83,162	705,762	14,315,904	927,716

Source: Own estimates from data extracted from Bayraktarov et al. (2016)

*: For the following ecosystems, estimates are sourced from studies in developed countries and adjusted to PNG PPP level

The results of the preliminary profitability analysis for the mangrove, coral reefs and seagrass beds are summarised in table 43–table 45. For mangroves, the analysis shows that conservation and restoration are profitable under most circumstances, except when restoration costs exceed US\$ 9,642 ha⁻¹ (at a 10 percent discount rate). Beyond that level, the NPV becomes negative (i.e., BCR is lower than one).

Differently, restoring coral reefs and seagrass beds depends highly on the conservation work required. Especially for seagrass beds, average restoration costs are high, which could lead to uneconomic results. Cost break-even points are reached at US\$ 5,247 and US\$ 1,971 ha⁻¹ for coral reefs and seagrass beds, respectively.

Table 43: Profitability of mangroves restoration in PNG (2021 US\$ ha⁻¹)

		Discount rate					Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	394	300	205	COST	min	18,473	14,068	9,596
	average	6.43	4.90	3.35		average	15,641	11,236	6,764
	max	0.34	0.26	0.18		max	-36,041	-40,446	-44,918

Table 44: Profitability results of coral reef restoration in PNG (2021 US\$ ha⁻¹)

		Discount rate					Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	654	498	340	COST	min	10,062	7,665	5,231
	average	0.33	0.25	0.17		average	-20,656	-23,053	-25,487
	max	0.07	0.05	0.04		max	-135,456	-137,853	-140,286

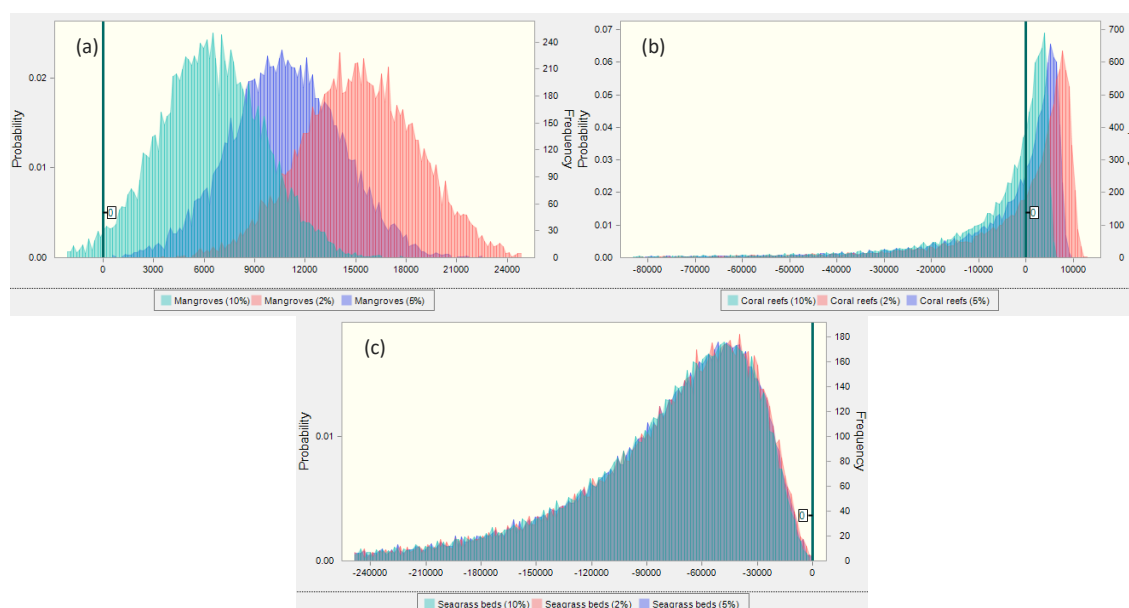
Table 45: Profitability results of seagrass bed restoration in PNG (2021 US\$ ha⁻¹)

		Discount rate		
		2%	5%	10%
COST	BCR			
	min	13.90	10.59	7.24
	average	0.02	0.01	0.01
	max	0.00	0.00	0.00

		Discount rate		
		2%	5%	10%
COST	NPV			
	min	3,512	2,612	1,698
	average	-235,806	-236,706	-237,620
	max	-1,619,718	-1,620,618	-1,621,532

The graphs in Figure 18 show that mangrove results remain positive under most circumstances. Meanwhile, results from coral reefs indicate that NPVs will remain positive in most cases. However, there is a chance of variation in costs and benefits that would negatively affect the NPV. The analysis confirms that positive results are improbable for seagrass beds and that the previously indicated negative NPVs (Table 45) are confirmed after almost all iterations.

Figure 18: Sensitivity analysis results for mangroves (a), coral reefs (b) and seagrass beds (c) NPVs (US\$) in PNG



4.2.5.6 São Tomé

There is little evidence from the academic or grey literature supporting the economic evaluation of São Tomé ES. Two recent studies from Afonso *et al.* (2021, 2022) have looked at mangrove ES and their relevance to the local population but came short of providing a compelling quantitative evaluation of their economic value. Therefore, we extracted and analysed data from ESVD. Our review of 19 observations indicates that the average value of mangrove ES is US\$ 1,075 ha⁻¹ yr⁻¹. The summary statistic of our estimates is provided in Table 46.

Table 46: Summary statistics of international mangrove values (2021 Int. US\$ ha⁻¹ yr⁻¹)

n	Min.	Median	Average	Max	SD
19	3.1	625	1,075	4,328	± 1,235

Source: Own elaboration from data of the ESVD database

The cost of replanting one hectare of mangroves in São Tomé averages US\$ 2,056 (min US\$ 34 and max. US\$ 38,972).

The CBA shows that restoring mangroves is generally profitable, except in cases where the restoration costs exceed US\$ 9,152 ha⁻¹ (at a 10 percent discount rate). Beyond that level, the NPV becomes negative, and costs would outweigh the benefits (Table 47).

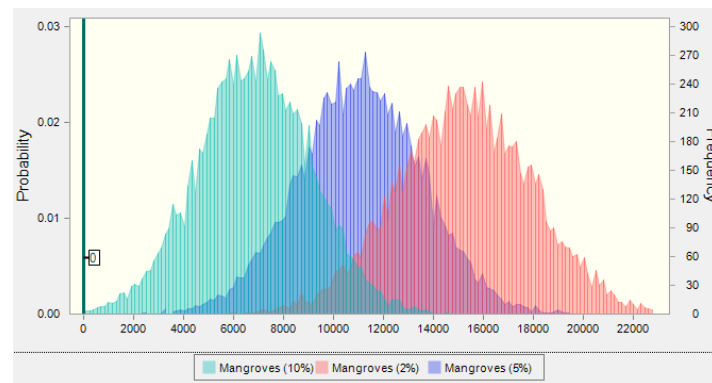
Table 47: Profitability of mangroves restoration in São Tomé (2021 US\$ ha⁻¹)

BCR		Discount rate		
		2%	5%	10%
COST	min	524	399	273
	average	8.55	6.52	4.45
	max	0.45	0.34	0.23

NPV		Discount rate		
		2%	5%	10%
COST	min	17,544	13,363	9,119
	average	15,522	11,341	7,096
	max	-21,394	-25,575	-29,820

Figure 19 presents the result of the sensitivity analysis on the Mangroves NPV. The graph shows that the restoration of mangroves is economically viable, and these positive results are confirmed even after the uncertainty has been factored in.

Figure 19: Sensitivity analysis results for mangroves NPVs (US\$) in São Tomé



4.2.5.7 Solomon Islands

The share of the blue economy in Solomon Island approaches 13.3 percent of the GDP (Seidel and Lal, 2010), and tourism accounts for a large part of this value. According to Spalding *et al.* (2017), reef tourism revenues were nearly 2 percent of the national GDP, for a mean value of reefs of nearly US\$ 3,206 km⁻² (US\$ 32 ha⁻¹). The average spending per visitor in the Solomon Islands is significantly higher than in other Pacific islands. With an average outlay of US\$ 2,803 per tourist, revenues for the sector reached between US\$ 19.5 million – US\$ 24.3 million in 2013 (2iis Consulting, 2015). Other estimates of tourism expenditures from foreign visitors amounted to about US\$ 68 million yr⁻¹, out of which 22 percent (US\$ 15 million) is the contribution of marine ES to tourism (Arena *et al.*, 2015).

Coral reefs are the most relevant marine ecosystem in the Solomon Islands and are responsible for most marine ES value generation. Coral reefs present a diverse array of direct, indirect and non-use values. In a study of four communities, Albert *et al.* (2012) estimated the direct use value of coral reefs between US\$ 75,000 - US\$ 170,000 km⁻² reef per year (for subsistence and cash fishing). The indirect use value of coral reefs – using the replacement value of shoreline protection as a proxy – had a total value between US\$ 18,000 - US\$ 270,000 km⁻² yr⁻¹, while the non-use value ranged from US\$ 1,200 to US\$ 7,700 per km⁻² yr⁻¹. Hence, the TEV for the direct, indirect (through coastal protection) and non-use value was estimated between US\$ 100,000 to US\$ 420,000 per km⁻² yr⁻¹ (US\$ 1,000 – US\$ 4,200 ha⁻¹ yr⁻¹).

One attribute that all coral reefs have in common is their direct and significant benefit to the people in the coastal communities of the Solomon Islands. Their presence allowed for protecting

inshore fisheries for a total value of about US\$ 58.8 million yr⁻¹, corresponding to US\$ 114 person⁻¹ yr⁻¹ (Arena *et al.*, 2015). Instead, the protection service value of the entire coastal wetland nears US\$ 2 million yr⁻¹ (Costanza *et al.*, 2021). Other studies reported the importance of provisioning services brought by coral reefs to yield a mean cash equivalent of US\$ 7,386 (SE: ±7,796) person⁻¹ yr⁻¹ for reef products harvested for both subsistence and cash needs (Albert *et al.*, 2015). Additionally, the coastal protection provided by coral reefs against damage from storm surges is estimated to be US\$ 3.3 – 5.6 million yr⁻¹ (Arena *et al.*, 2015).

Meanwhile, past estimates on mangrove contribution looked at actual and potential carbon sequestration. Arena *et al.* (2015) estimated the value of carbon sequestration provided by mangrove forests at US\$ 21 million yr⁻¹ with additional potential revenue from the sale of carbon offsets – from the protection of stored carbon in mangroves – in the range of US\$ 7.3 – US\$ 9.7 million yr⁻¹.

In the Solomon Islands, mangrove forests and coral reefs represent the most relevant ecosystems. Table 48 summarises each ecosystem's relative annual contribution per hectare and its aggregate sum (i.e. coastal ecosystem), while ecosystem restoration costs are provided in Table 49.

Table 48: Ecosystem services values in the Solomon Islands (2021 US\$ ha⁻¹ yr⁻¹)

Sector	Ecosystem service	Coastal ecosystem	Mangroves	Coral reefs	Seagrass beds
Fisheries	Subsistence fishing	498	126	275	97
	Commercial fishing	87	22	48	17
Tourism	Tourism and recreation	133	40	93	-
Regulating services	Coastal protection	253	64	140	49
	Carbon sequestration	182	128	18	36
Total		1,154	380	574	200

Source: Own estimates from data included in Arena et al. (2015)

Table 49: Restoration cost by type of coastal ecosystem in Solomon Islands (2021 Int.US\$ ha⁻¹)

	Mangroves	Corals	Oyster reefs*	Salt marshes*	Seagrasses*
Min cost	27	9	3,500	53	156
Average cost	1,645	17,562	175,971	356,498	136,909
Max cost	31,178	83,162	705,762	14,315,904	927,716

Source: Own estimates from data extracted from Bayraktarov et al. (2016)

*: For the following ecosystems, the estimates are sourced from studies in developed countries and adjusted to Solomon Islands PPP level

The data included in Table 50 - Table 52 present the CBA results for each ecosystem. For mangroves, the analysis shows that conservation and restoration are profitable until restoration costs exceed US\$ 3,235 ha⁻¹ (at a 10 percent discount rate). Differently, restoring coral reefs and seagrass beds is less profitable and depends highly on the conservation work required. Especially for seagrass beds, average restoration costs are high, which could lead to negative results. The break-even for coral reefs and seagrass bed restoration is reached for restoration costs of US\$ 4,890 and US\$ 1,699 ha⁻¹, respectively.

Table 50: Profitability of mangroves restoration in the Solomon Islands (2021 US\$ ha⁻¹)

BCR		Discount rate		
		2%	5%	10%
COST	min	103	78	53
	average	1.68	1.28	0.87
	max	0.09	0.07	0.05

NPV		Discount rate		
		2%	5%	10%
COST	min	6,152	4,675	3,174
	average	2,512	1,034	-466
	max	-63,937	-65,415	-66,915

Table 51: Profitability of coral reef restoration in the Solomon Islands (2021 US\$ ha⁻¹)

BCR		Discount rate		
		2%	5%	10%
COST	min	474	361	247
	average	0.24	0.18	0.12
	max	0.05	0.04	0.03

NPV		Discount rate		
		2%	5%	10%
COST	min	9,373	7,139	4,871
	average	-30,122	-32,356	-34,624
	max	-177,721	-179,955	-182,223

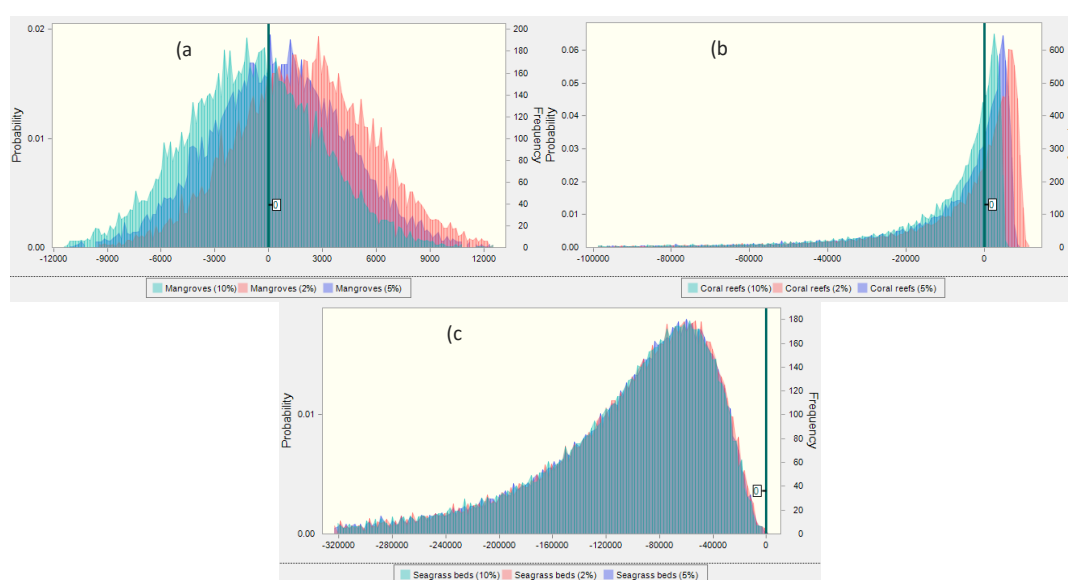
Table 52: Profitability results of seagrass bed in the Solomon Islands (2021 US\$ ha⁻¹)

BCR		Discount rate		
		2%	5%	10%
COST	min	9.32	7.10	4.85
	average	0.01	0.01	0.01
	max	0.00	0.00	0.00

NPV		Discount rate		
		2%	5%	10%
COST	min	2,913	2,137	1,349
	average	-304,782	-305,558	-306,346
	max	-2,084,097	-2,084,873	-2,085,661

The sensitivity analysis graphs shown in Figure 20 indicate that the odds of obtaining positive or negative results are spread even for mangroves and coral reefs, with mangroves leaning slightly towards positive and coral reefs towards negative results. Opposite, the reading of seagrass beds results is straightforward and negative in almost all variations considered.

Figure 20: Sensitivity analysis results for mangroves (a), coral reefs (b) and seagrass beds (c) NPVs (US\$) in Solomon Islands



4.2.5.8 Suriname

Very few studies evaluated ES in Suriname, and none investigated the economic value of those services. Only one study from Sieber *et al.* (2021) provided a spatial ES assessment in Suriname. However, it did not contain any estimate regarding the actual monetary contribution of mangrove ES to the local population. Therefore, we analysed data from the ESVD and considered

the average value of mangrove forests to equal US\$ 1,075 ha⁻¹ yr⁻¹. The summary statistic of our estimates is provided in Table 53.

Table 53: Summary statistics of international mangrove values (2021 Int. US\$ ha⁻¹ yr⁻¹)

n	Min.	Median	Average	Max	SD
19	3.1	625	1,075	4,328	± 1,235

Source: From data included in the ESVD database

The cost of replanting one hectare of mangroves in Suriname equals US\$ 1,234 (min US\$ 20 and max. US\$ 23,383). The CBA results in Table 54 show that conserving mangroves is generally profitable, and the NPV becomes negative only for restoration costs exceeding US\$ 9,122 ha⁻¹ (at a 10 percent discount rate).

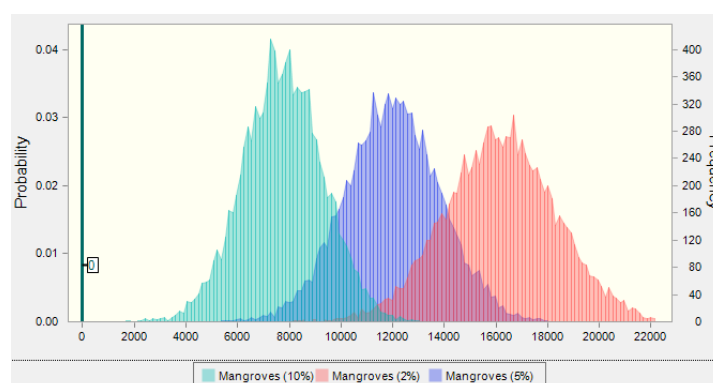
Table 54: Profitability of mangroves restoration in Suriname (2021 US\$ ha⁻¹)

BCR		Discount rate		
		2%	5%	10%
COST	min	874	666	455
	average	14.25	10.86	7.42
	max	0.75	0.57	0.39

NPV		Discount rate		
		2%	5%	10%
COST	min	17,558	13,377	9,132
	average	16,344	12,163	7,918
	max	-5,805	-9,986	-14,231

The Monte Carlo analysis confirms the initial favourable assessment of the mangrove restoration. As shown in Figure 21, NPVs remain positive when discounted at different rates and under all variations in costs and benefits being considered.

Figure 21: Sensitivity analysis results for mangroves NPVs (US\$) in Suriname



4.2.5.9 Vanuatu

According to Seidel and Lal (2010), the share of the blue economy in Vanuatu is highly relevant and equal to 32 percent of the GDP. In their study, Pascal *et al.* (2018) estimate the TEV of Vanuatu ES in US\$ 120,199 (2010 US\$ values) km⁻² yr⁻¹ (US\$ 1,201 ha⁻¹ yr⁻¹), with fishery tourism and coastal protection being valued US\$ 428, US\$ 326 and US\$ 343 ha⁻¹ yr⁻¹. The seven marine and coastal ES categories assessed in Vanuatu had a TEV between US\$ 37 – 64 million yr⁻¹ in 2013 US\$ values (Pascal *et al.*, 2015). The estimated value of avoided damages thanks to coastal wetlands amounts to about US\$ 9.9 million yr⁻¹.

According to Spalding *et al.* (2017), reefs tourism revenues amount to about 6.4 percent of the national GDP, bringing a mean value of reef of nearly US\$ 27,730 km⁻² (US\$ 277 ha⁻¹). Instead, the value of mangrove ES was estimated between US\$ 4,291 and US\$ 8,535 ha⁻¹ (Pascal, 2014; Pascal and Bulu, 2013). Table 55 presents the ecosystem's relative annual contribution per hectare and its aggregate sum (i.e. coastal ecosystem), while Table 56 shows the estimated restoration costs per type of ecosystem in Vanuatu.

Table 55: Ecosystem services values in Vanuatu in (2021 US\$ ha⁻¹ yr⁻¹)

Sector	Ecosystem service	Coastal ecosystem	Mangroves	Coral reefs	Seagrass beds
Fishery	Subsistence fishing	244	14	224	5
	Commercial fishing	274	15	252	6
Tourism	Tourism and recreation	366	109	255	-
Regulating services	Coastal protection	701	40	645	15
	Carbon sequestration	61	42	6	12
Total		1,644	222	1,383	38

Source: Own estimates from data included in Pascal *et al.* (2018), Spalding *et al.* (2017) and Pascal *et al.* (2015)

Table 56: Restoration cost by type of coastal ecosystem in Vanuatu (2021 Int.US\$ ha⁻¹)

	Mangroves	Corals	Oyster reefs*	Salt marshes*	Seagrasses*
Min cost	67	22	8,749	133	389
Average cost	4,112	43,905	439,928	891,245	342,272
Max cost	77,944	207,904	1,764,404	35,789,760	2,319,289

Source: Own estimates from data extracted from Bayraktarov et al. (2016)

**: These estimates are sourced from studies in developed countries and adjusted to Vanuatu PPP level*

The data included in Table 57 - Table 59 report the CBA results per ecosystem. For mangroves, the analysis shows that conservation and restoration are profitable under most circumstances, except when restoration costs exceed US\$ 1,893 ha⁻¹ (at a 10 percent discount rate).

Unlike coral reefs and seagrass beds, the results are more sensitive to conservation costs. Especially for seagrass beds, average restoration costs are high, which could lead to negative results. The cost break-even point for coral reefs and seagrass beds is US\$ 11,782 and US\$ 328 ha⁻¹, respectively.

Table 57: Profitability of mangroves restoration in Vanuatu (2021 US\$ ha⁻¹)

		Discount rate					Discount rate		
		BCR					NPV		
		2%	5%	10%			2%	5%	10%
COST	min	54	41	28	COST	min	3,569	2,704	1,826
	average	0.88	0.67	0.46		average	-476	-1,341	-2,219
	max	0.05	0.04	0.02		max	-74,308	-75,173	-76,051

Table 58: Profitability of coral reef restoration in Vanuatu (2021 US\$ ha⁻¹)

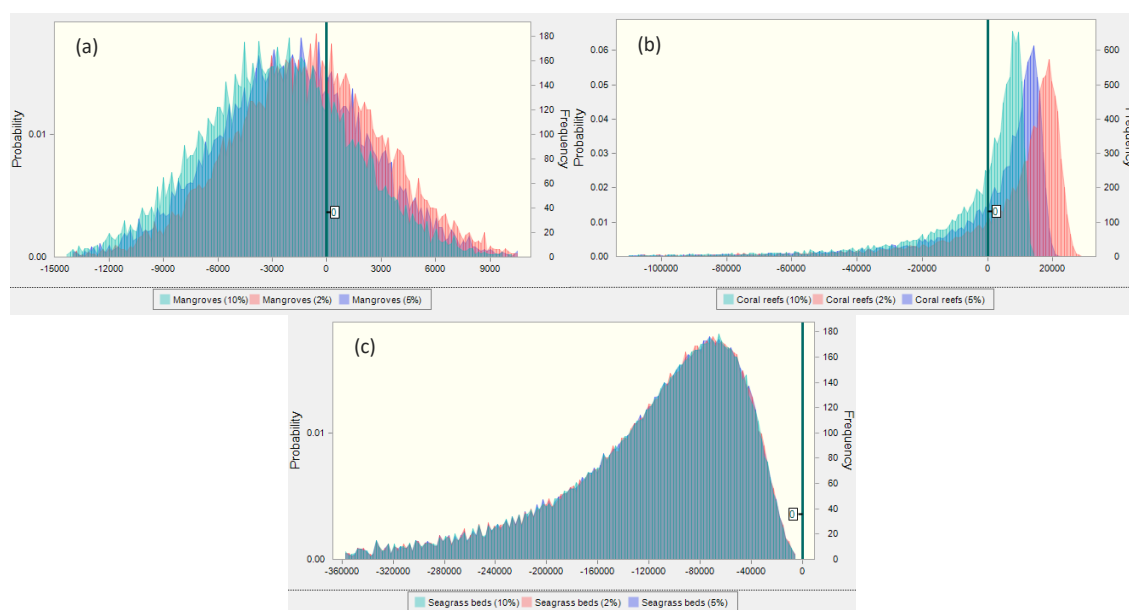
		Discount rate					Discount rate		
		BCR					NPV		
		2%	5%	10%			2%	5%	10%
COST	min	1 028	783	535	COST	min	22,606	17,224	11,760
	average	0.52	0.39	0.27		average	-21,277	-26,659	-32,123
	max	0.11	0.08	0.06		max	-185,276	-190,658	-196,122

Table 59: Profitability of seagrass bed in Vanuatu (2021 US\$ ha⁻¹)

	BCR	Discount rate				NPV	Discount rate		
		2%	5%	10%			2%	5%	10%
COST	min	1.62	1.24	0.84	COST	min	242	92	-61
	average	0.00	0.00	0.00		average	-341,641	-341,791	-341,944
	max	0.00	0.00	0.00		max	-2,318,658	-2,318,808	-2,318,961

The sensitivity analysis of Vanuatu CBA results is presented in Figure 22. The graphs indicate that while NPVs of coral reef restoration are likely to be positive, the NPVs from mangrove restoration show a tendency towards the negative results area. In the latter case, the probability of obtaining a negative NPV is slightly higher than that of getting a positive result. Opposite, the outcomes from seagrass bed restoration are consistently negative under all the scenarios considered.

Figure 22: Sensitivity analysis results for mangroves (a), coral reefs (b) and seagrass beds (c) NPVs (US\$) in Vanuatu



4.2.6 Conclusions

This study contributes novel insights to the literature on ecosystem services by addressing the often-neglected economic evaluation of these services, which are crucial for human survival. Previous works (Groot *et al.*, 2010; Costanza *et al.* 1997) have highlighted the significant gap in market valuation of ecosystem services. Our research fills this gap by combining economic evaluation methods and remote sensing techniques to estimate the monetary value of coastal ecosystems in nine SIDS.

Through this comprehensive CBA, we quantified the economic relevance of mangrove forests, coral reefs, and seagrass beds. This analysis includes a broad spectrum of benefits such as coastal protection, regulating and provisioning services. Through it, we contribute to providing up-to-date and detailed cost estimates for restoring and conserving marine ecosystems through nature-based solutions and grey infrastructures.

Our analysis shows the relevance of each ecosystem in providing direct- and indirect-use benefits to the local population. Carbon sequestration, coastal protection, tourism and fishery are among the most critical services marine ecosystems provide across the countries considered. The latter reiterates the need to preserve and maintain natural endowments that provide substantial revenues through commercial activities – with the potential to generate employment and business opportunities for future local development. Furthermore, these ecosystems often provide invisible but particularly significant protection and regulating services that safeguard the livelihoods and proprieties of coastal populations.

One additional contribution of our study is the demonstration of substantial cost savings and improved outcomes when restoration efforts prioritise mangrove forests. While coral reefs and seagrass beds also offer significant benefits, their restoration is often economically challenging. This finding is critical for policymakers and stakeholders as it highlights the importance of focusing resources on the most cost-effective and impactful ecosystems.

However, in our analysis, we assumed a linear provision capacity of each ecosystem and overlooked interactions and synergies across multiple biomes. In reality, the service capacity of coastal ecosystems is highly interactive and depends on the presence of various biomes at once. Therefore, ecological and conservation activities should be defined and implemented using a systemic and holistic approach that integrates biological, environmental, and socio-economic

components. Depending on the context, we suggest selecting and implementing conservation and restoration strategies focusing on the environmental system as a whole: on the one hand, reducing the inland ecological stressors and, on the other hand, including more than one coastal ecosystem. Despite the higher implementation costs, the benefits will be greater than working on a single ecosystem due to the synergistic interactions among coastal ecosystems (mangroves, coral reefs and seagrass beds) restored to healthier conditions.

Furthermore, our extensive review of secondary data from the existing literature reveals the fragmented and incomplete nature of global-level data on ecosystem services. This underscores the need for more comprehensive and standardised data collection and research on the subject. By highlighting these data limitations, our study provides a foundation for future research to build upon and improve the robustness of comparative analyses. Meanwhile, given the high variability in estimates of costs and benefits, we recommend performing a detailed in-country assessment before defining the most appropriate restoration solutions. CBA, such as the one presented here, can support the selection of conservation strategies. However, primary evidence should be gathered in situ, according to the required spatial analysis scale (i.e. local, national, regional and global).

Additionally, our evaluation underscores the greater economic efficiency of nature-based solutions compared to grey infrastructures for coastal protection, with NbS being two to one hundred times cheaper than artificial. This finding supports the argument of existing literature for integrating NbS into coastal management strategies, providing a clear economic rationale for their adoption.

Finally, we suggest developing standardised procedures and tools to support informed decisions on the sustainable management of natural ecosystems. These procedures should be implemented to compare results across countries, monitor long-term outcomes from conservation and restoration projects in developing countries, and replicate assessments similar to the one presented here.

In summary, this research advances the literature by providing a detailed economic evaluation of marine ecosystem services, identifying cost-effective restoration strategies, and highlighting critical data gaps that need to be addressed to support robust policy-making.

4.2.7 Appendix

Figure 23: Map of coastal ecosystems in Belize

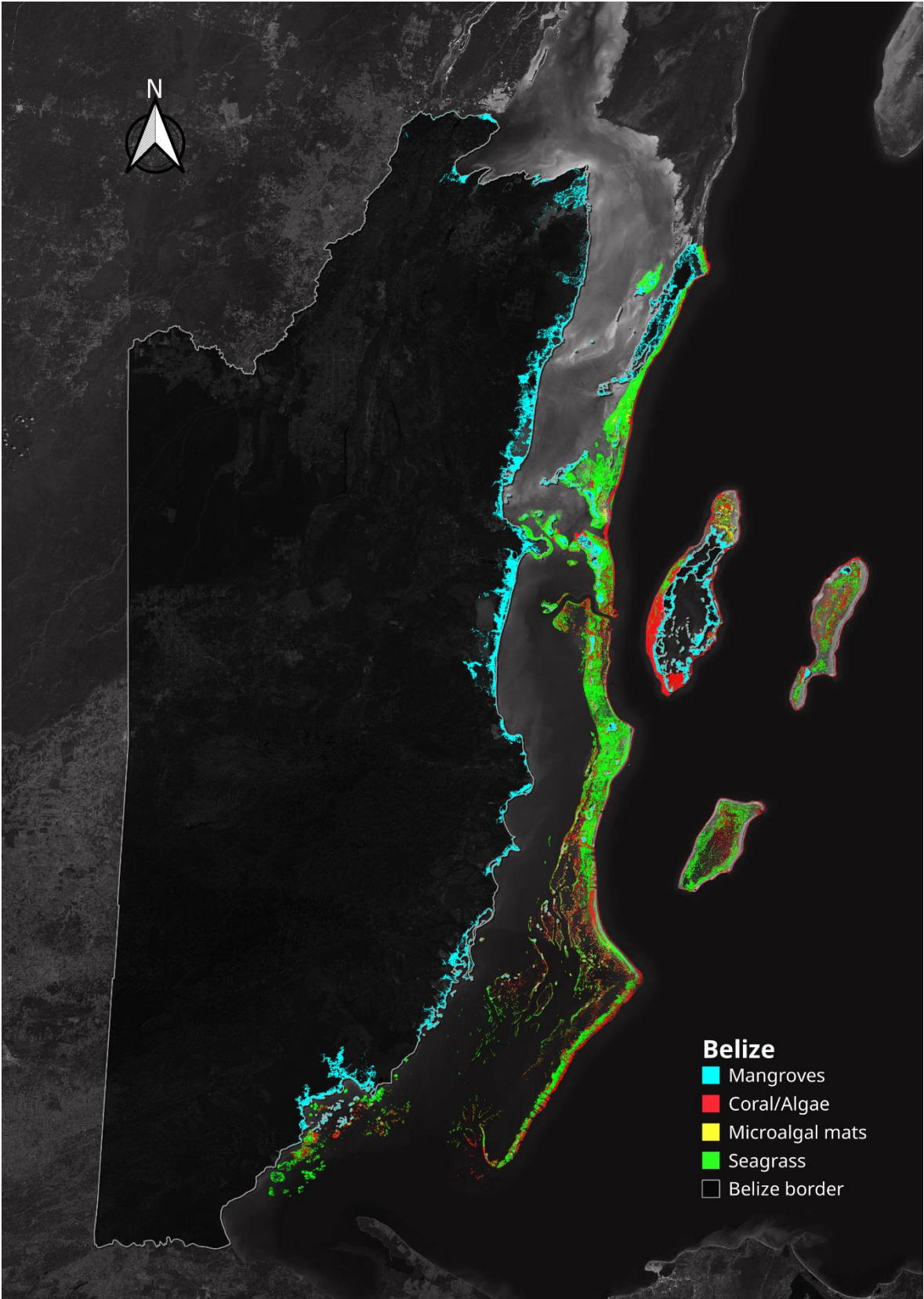


Figure 24: Map of coastal ecosystems in Comoros



Figure 25: Map of coastal ecosystems in Dominican Republic

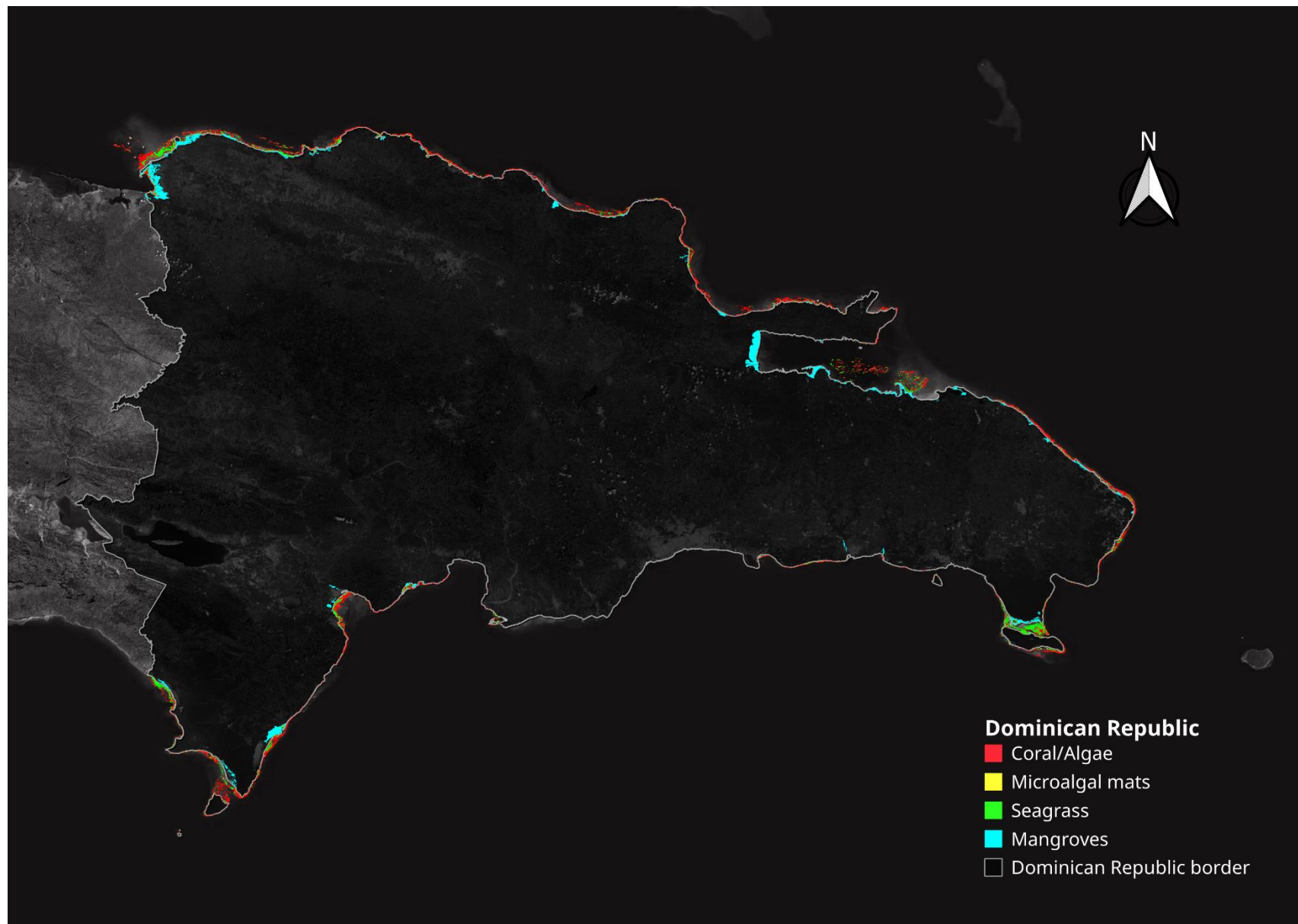


Figure 26: Map of coastal ecosystems in Guinea Bissau

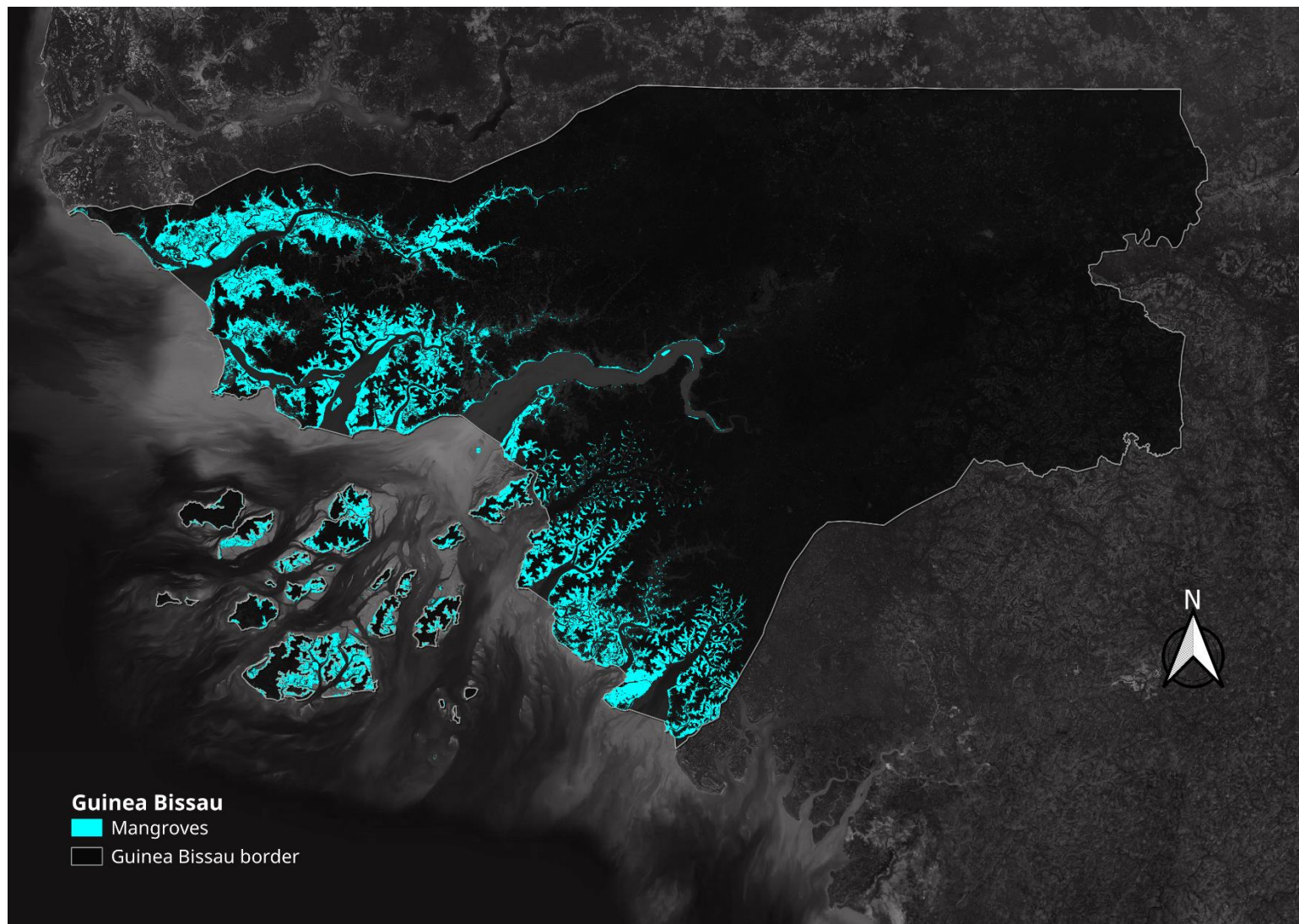


Figure 27: Map of coastal ecosystems in Papua New Guinea

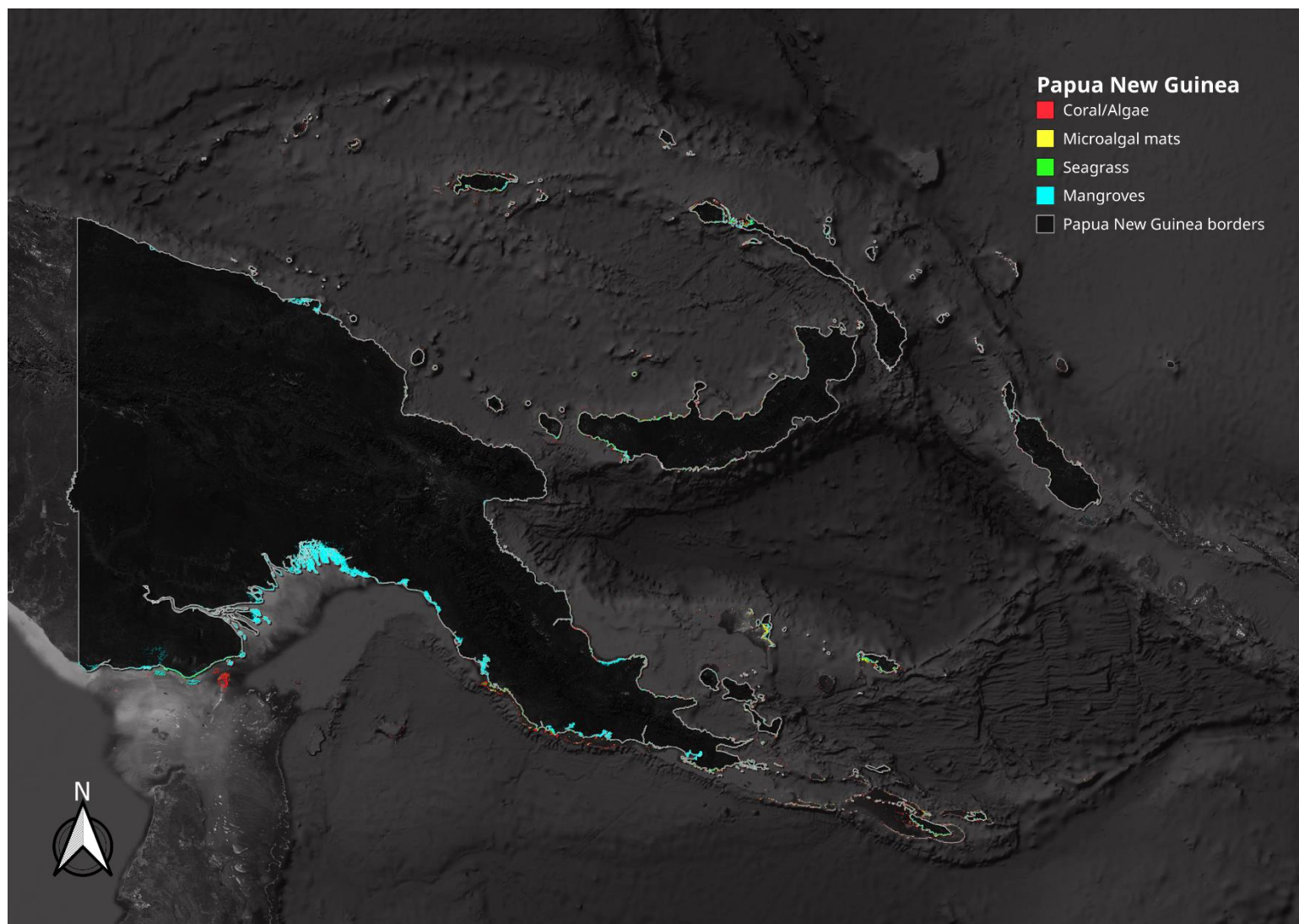


Figure 28: Map of coastal ecosystems in São Tomé



Figure 29: Map of coastal ecosystems in Solomon Islands

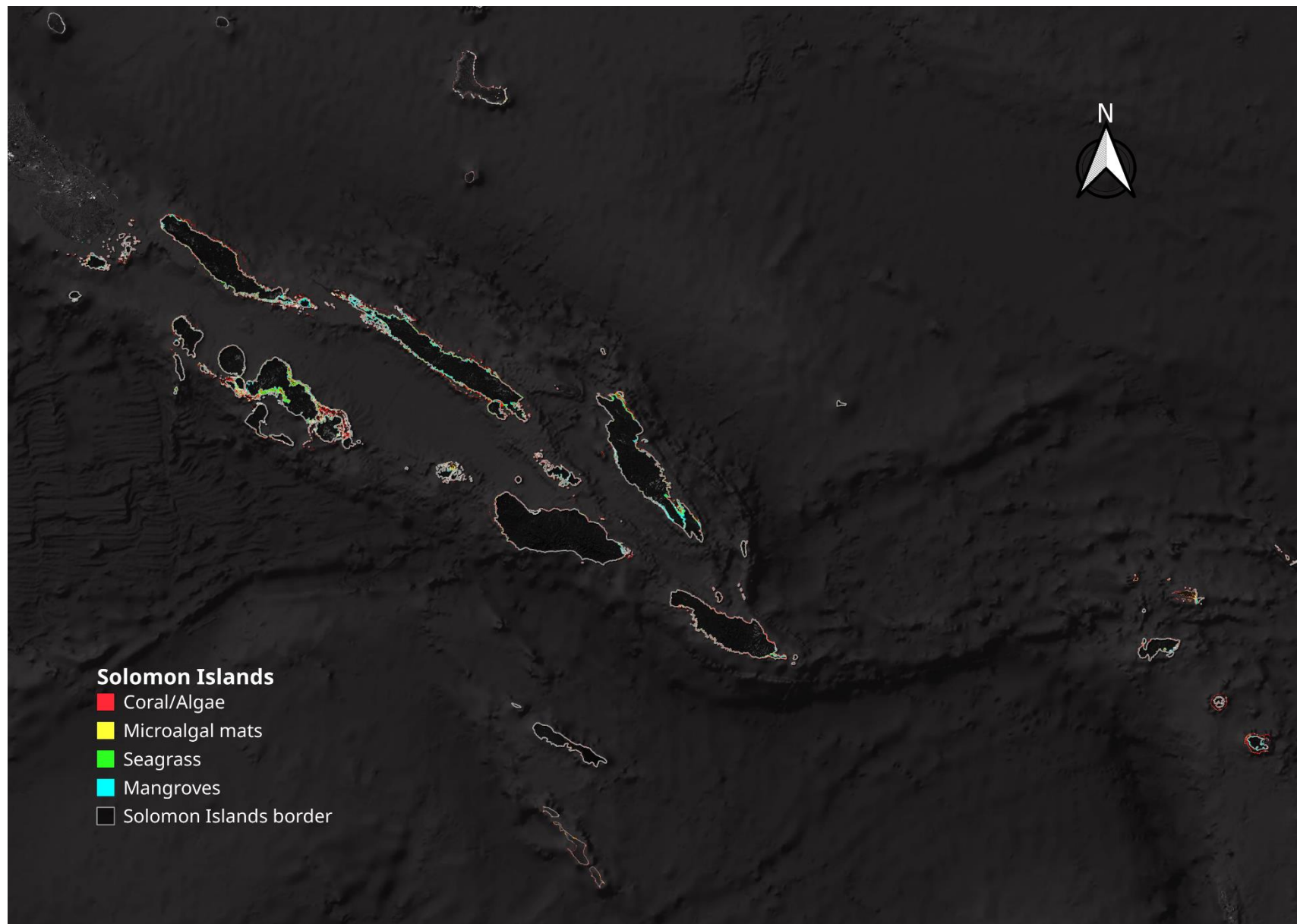


Figure 30: Map of coastal ecosystems in Suriname

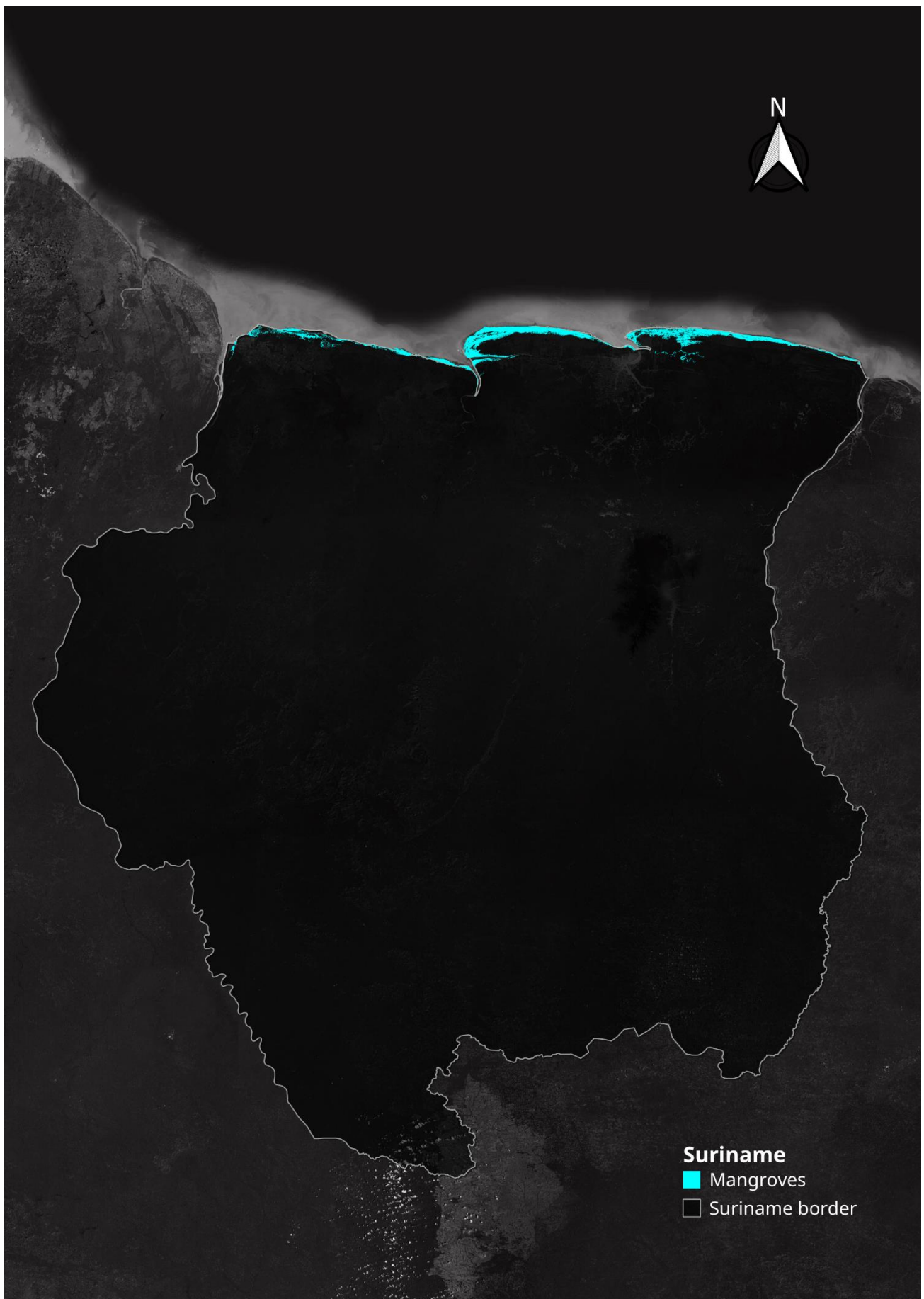
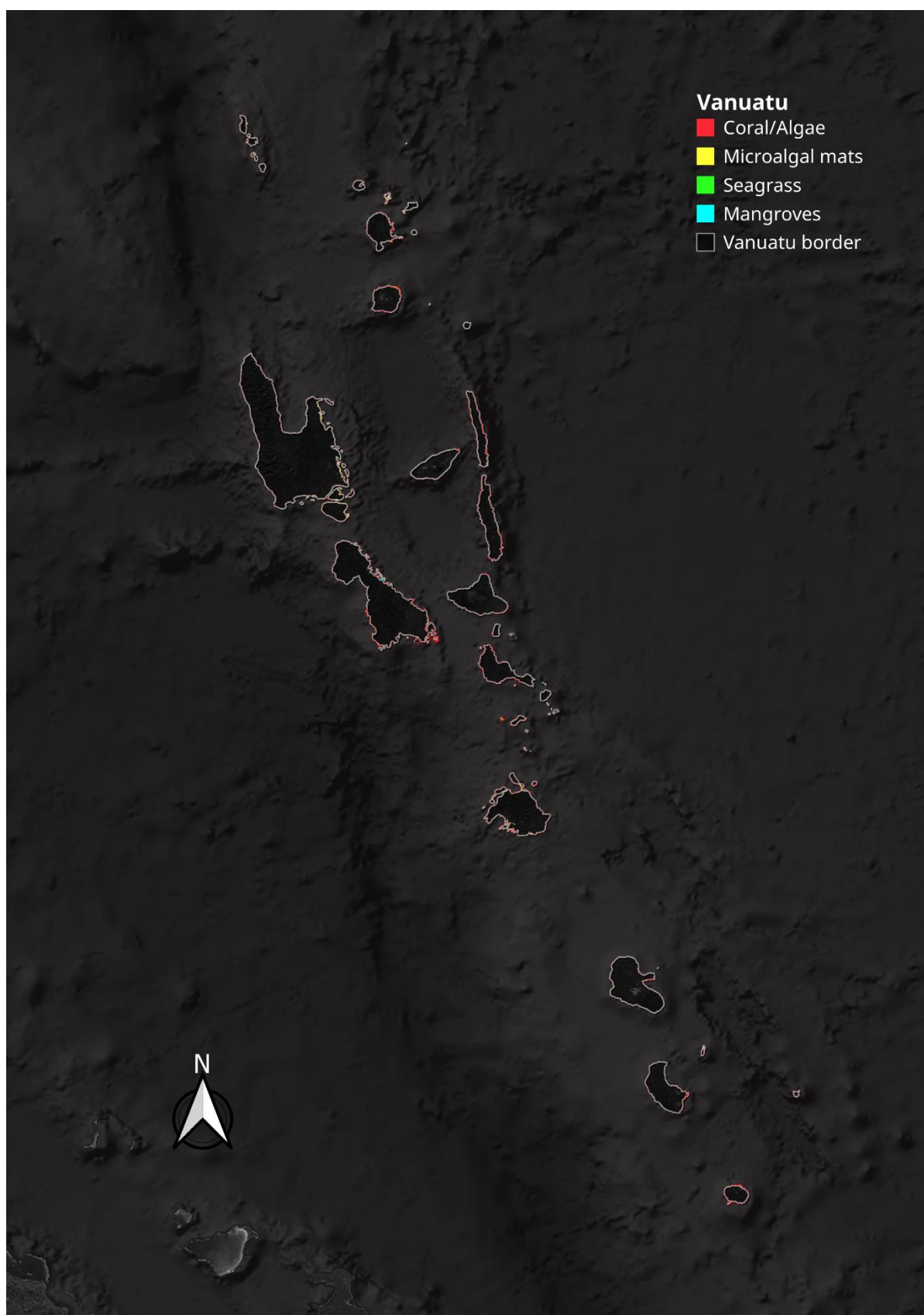


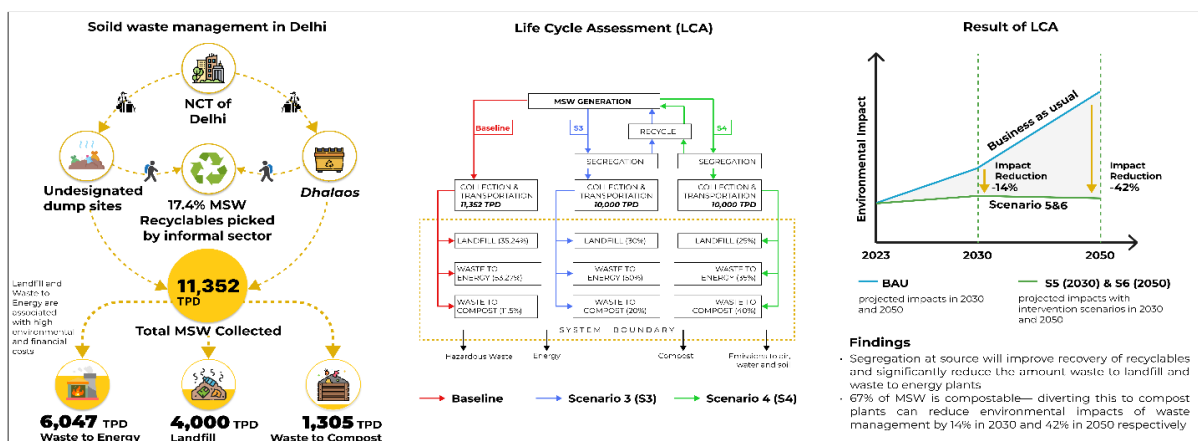
Figure 31: Map of coastal ecosystems in Vanuatu



4.3 Paper 3: Greening the City: A Holistic Assessment of Waste Management Alternatives in India

Abstract: Waste is one of the major urban challenges faced globally today, and the severity of the challenge is further exacerbated by rapid urbanisation, growing populations and increasing per capita waste generation. As one of the largest urban agglomerations in the world, Delhi collects 11,352 tonnes of waste daily. Without adequate segregation, most of this waste is sent to dumpsites and waste to energy plants, often associated with significant capital costs and environmental externalities. This paper conducts a life cycle assessment of the current waste management system and a comparative analysis with a suggested alternative scenario, where the share of recyclables and compostables going to landfills and waste-to-energy plants is reduced through adequate segregation. Our results revealed that landfills and waste-to-energy plants are associated with significant adverse environmental impacts such as climate change, soil and water acidification, freshwater eutrophication, human toxicity, and respiratory health. In comparison, compost plants showed negligible emissions per tonne of waste. The alternative scenario (i.e. reduce waste to landfill through adequate segregation) can help reduce the negative impact on all environmental indicators by an average of 23%. We posit that the prevailing narrative of addressing the waste issue through waste-to-energy plants in Delhi goes against the country's climate neutrality targets. Instead, the circular economy approach offers simpler, faster, and more cost-effective solutions that policymakers should consider to reduce the financial and environmental load of the current and future waste management issue.

Figure 32: Graphical abstract



4.3.1 Introduction

Waste management represents one of the most significant urban challenges faced globally. This issue has received considerable international attention due to its detrimental effects on environmental pollution, public health, and climate change (Gómez-Sanabria et al., 2022; Siddiqua et al., 2022; Vinti et al., 2021). The alarming increase in waste generation is attributed to rapid urbanisation and economic growth (Allesch and Huber-Humer, 2023; Balasubramanian, 2015; Kaza et al., 2018). Rapid urbanisation, especially in developing nations, is characterised by increasing consumption, production, and waste generation (Gutberlet et al., 2017; Rani Yaduvanshi et al., 2016). The global economy produces an estimated 2.1 billion tonnes of municipal solid waste (MSW) per year, of which 30% ends up in landfills (UNEP, 2024). By 2050, MSW generation is expected to reach 3.8 billion tonnes (UNEP, 2024).

The Intergovernmental Panel on Climate Change (IPCC, 2022) and the World Bank (Kaza et al., 2018) estimate that about 4%-5% of global greenhouse gas (GHG) emissions originate from MSW. Since 2010, emissions from waste have been growing steadily at an average rate of 1.6% (IPCC, 2022), and they now represent the second-most relevant emission source in cities (Lu and Li, 2019). Adopting efficient solid waste management (SWM), such as waste prevention, landfill mitigation, and recycling, is a priority (Nisbet et al., 2019) that could help reduce global GHG emissions by 5% to 20% (Coalition for Urban Transitions, 2019; UNEP and ISWA, 2015).

However, the challenge of managing waste efficiently is further compounded by an expected 22% increase in world population by 2050 (Mahendra et al., 2021; UNEP and ISWA, 2015). Kaza et al. (2018) estimate that by then, per capita waste growth will be markedly higher in low- and medium-income countries (+40%) compared to in high-income nations (+19%). Meanwhile, India is already struggling to manage industrial and domestic waste due to its high industrial output and rapid rate of urbanisation. The country has seen a surge in GHG emissions related to waste by a staggering 224% between 1994 and 2016 (MoEFCC, 2021).

Clearly, the current approach to economic growth and resource over-extraction fails to address the environmental consequences of the unregulated market from an ecological perspective (Krausmann et al., 2018). Even from an economic perspective, resource depletion, embedded energy leakages, and high costs of material recovery and waste management are not sustainable.

Thus, the circular economy (CE) approach has been suggested as an effective solution to SWM issues that simultaneously stimulate economic growth while staying within planetary boundaries (Calderón Márquez and Rutkowski, 2020; Fatimah et al., 2020; Gómez-Sanabria et al., 2022; Joshi et al., 2019; Lacy and Rutqvist, 2015). The widely accepted definition provided by the Ellen MacArthur Foundation (2019) describes CE as a "regenerative" economic model that aims "to benefit businesses, society, and the environment" (p. 20) by minimising the resources and energy used in production processes.

CE is based on three principles: eliminate waste and pollution, circulate products and materials (at their highest value), and regenerate nature. Thus, adopting a CE approach to production and consumption can reconcile economic and environmental benefits – two traditionally conflicting aspects of sustainability in practice – and reduce carbon footprints (Fatimah et al., 2020).

As one of the green growth solutions to achieve carbon neutrality, the CE model focuses on decoupling the environmental externalities resulting from economic growth activities. This approach requires a shift from traditional linear economic models, which prioritise the extraction, production, and disposal of goods and services, towards a more circular approach that aims to maximise resource efficiency and minimise waste. Hence, CE can be seen as a paradigm shift in addressing the waste challenge, moving from the perspective of 'waste-disposal' towards 'waste management' and treatment of 'waste' as a 'resource' for the CE (UNEP and ISWA, 2015).

In addition, several Sustainable Development Goals (SDGs) directly or indirectly deal with waste management, climate action, sustainable cities, and combating land degradation and environmental pollution (Cibrario, 2018; UNGA and ECOSOC, 2018). CE offers a systemic approach to address these global challenges, and its inclusion in national and city roadmaps for decarbonisation and climate neutrality can contribute towards achieving nationally determined contributions under the Paris Agreement (Ellen MacArthur Foundation, 2019).

Given the strong positive correlation between CE practices, SDGs and Paris Agreement targets (Schroeder et al., 2018), national governments, international organisations, and industries are now seeing the possibilities that CE has to offer in achieving many of these goals (European Commission, 2015; Korhonen et al., 2018; UNGA and ECOSOC, 2018). The European Union (EU) has been the flagbearer of promoting the concept of CE as an approach towards net zero waste (Gemeente Amsterdam, 2020; Heshmati, 2017), and several high-income countries are already

adopting CE principles across industries (Geissdoerfer et al., 2017; Ghisellini et al., 2016). Even developing nations with high economic growth rates are considering CE to mitigate the impact of increasing production and consumption patterns of rapidly urbanising cities (Gutberlet et al., 2017; Rani Yaduvanshi et al., 2016).

In India, nationwide campaigns such as "Swachh Bharat" aimed at promoting clean and hygienic urban development by closing material cycles within city boundaries (MoHUA, 2021). However, the challenge to close the material loops is immense, as cities are open systems, i.e., the materials enter and exit a city's administrative and geographic boundaries (Chaudhary et al., 2021b).

The application of CE has been hampered by several factors, including inadequate infrastructures, lack of funding, unsustainable consumption and social unawareness of adequate practices (Fiksel et al., 2021). Consequently, the efforts made thus far have been insufficient to manage waste effectively (Joshi and Ahmed, 2016).

Segregation at source is critical to an efficient SWM (Assocham & PwC India, 2017). However, the increased consumption of packaged goods, improper waste disposal practices and lack of penalties have made segregation at source arduous (Manuja et al., 2019). The municipal authorities have attempted several awareness campaigns, such as "Zero-waste colonies", to promote dry and wet waste separation at the household level. Yet, to the frustration of those individuals separating waste at home, most of the waste gets to landfills unsegregated. An audit report by the New Delhi Municipal Council (NDMC, 2018) revealed that the waste received at the Okhla landfill site was mixed at the stage of transferring from auto tippers to compactors, which undermines and defeats the whole purpose of efforts made towards segregation.

Against this backdrop, this study investigates one of the biggest problems Delhi's National Capital Territory (NCT) faces - the challenge of managing its waste. The city's growing population and economy have led to increased waste and inadequate SWM practices (Pujara et al., 2019). The poor segregation of waste has resulted in an excessive load on the landfill sites, leading to a further decline in the urban environment and real estate market in the surrounding areas (Angmo and Shah, 2021; Chaudhary et al., 2021b).

Delhi generates over 11 352 metric tonnes (MT) of MSW per day (DPCC, 2023a), and due to the scarcity of land, landfill sites offer only a temporary solution for waste management. Other

methods, such as waste-to-energy (WtE) plants, have also been scrutinised for their emissions and impact on public health (Devraj and Sambyal, 2019; Dong et al., 2020).

The study assesses the economic and environmental viability of Delhi's current waste management system and investigates the potential for a circular economy approach. We do this by comparing emissions from various waste disposal methods such as recycling, incineration, and upcycling for infrastructure construction.

This study is the first of its kind, offering an innovative and comprehensive analysis of the economic and environmental impacts of Delhi's waste management processes. We used the latest data on waste generation, population, and cost estimates to represent the current situation accurately. Our comparison of different waste management scenarios helps identify areas for improvement. Additionally, our assessment of the circular economy's potential contribution to addressing waste management challenges will help shape policy recommendations and sustainable strategies for solid waste management in large cities, effectively bridging the gap between theory and practice.

Waste management usually falls under the responsibility of municipal bodies, which often work under limited resources and capacity (Kaza et al., 2018; UNEP & ISWA, 2015). Waste management constitutes a significant share of the budget in developing countries and could represent more than 20-25% of the total municipal budget (Kaza et al., 2018; Sharma, 2019; CPHEEO, 2000). In addition, the informal sector handles a fair share of waste and plays a vital role in recovering recyclables, especially in developing countries (Agarwal et al., 2005; ASSOCHAM & PwC India, 2017).

Over the years, there has been a paradigm shift in how we address the challenge of waste. The perspective of 'waste disposal' has transitioned towards 'waste management' and the treatment of 'waste' as a 'resource' for a circular economy (UNEP & ISWA, 2015). Several targets of the Sustainable Development Goals (SDG) deal with waste services, especially SDG 11 - Sustainable Cities and Communities and SDG 12 - Responsible Consumption and Production (Cibrario, 2018).

The Intergovernmental Panel on Climate Change (IPCC) estimates that 3% of global greenhouse emissions can be attributed to municipal solid waste. In comparison, another estimate by the World Bank estimates the share of municipal solid waste in global greenhouse emissions to be 5%. This does not include emissions from waste transportation (Kaza et al., 2018). A total

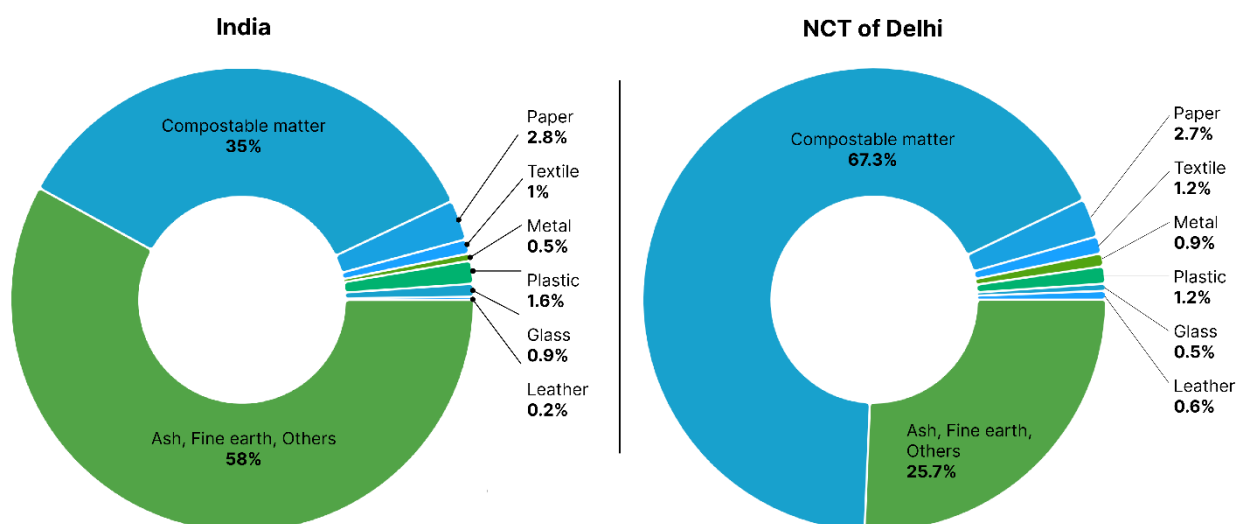
reduction of 15 to 20% in global greenhouse gas emissions is estimated to be achievable through efficient solid waste management such as waste prevention, landfill mitigation, recycling, etc. (UNEP & ISWA, 2015).

4.3.2 Literature review

4.3.2.1 Current State of MSW Management

With an estimated population of 21.4 million, the NCT of Delhi is growing at an average 2% rate each year (National Commission on Population, 2020). Civic bodies collect about 11 352 MT of solid waste daily (DPCC, 2023a). Nearly half of the MSW in Delhi is made up of biodegradable and recyclable material (UNDS, 2024), and yet most of the waste (92%) ends up in landfills and WtE plants (DPCC, 2023a). Figure 33 compares the MSW composition in India and Delhi.

Figure 33: MSW composition (% of total)



Source: Own elaboration from data in Khan et al. (2022), UNDS (2024) and DPCC (2023b)

4.3.2.2 Collection and Transportation

Unsegregated household waste is collected and dumped into larger dumper bins, undesignated dump sites, or intermediary dumping sites called 'dhalao'. These intermediary sites, located almost in every square kilometre, also serve as collection centres where the informal waste

pickers separate metals, plastics, paper and other recyclable waste. According to Agarwal et al. (2005), this informal mechanism collects almost 17.4% of the total generated MSW as recyclables.

After the waste has been collected from various intermediary areas or dhalaos, it is transported to disposal sites by a fleet of different types of vehicles, such as mini-trucks or mechanical trucks with an average load capacity of 8.6m³ (Gupta and Arora, 2016). The waste collection rate is estimated at 77% of the total generated MSW (Kaza et al., 2018). The urban local bodies or private contractors transport the waste under the public-private partnership model to four designated landfill sites: Ghazipur, Narela-Bawana, Bhalswa and Okhla (DPCC, 2023a).

4.3.2.3 MSW Processing

The current landfill sites are equipped with two types of processing facilities — WtE and Waste-to-compost (WtC). The WtE plants generate 84 MWh of energy from approximately 6 550 tonnes per day (TPD) of MSW. In contrast, the combined capacity of the centralised compost plants is estimated at 1 305 TPD. The four landfills receive 4 000 TPD of MSW (DPCC, 2023a).

4.3.2.4 Key Stakeholders

The civic bodies responsible for waste management are the North, South and East Municipal Corporations of Delhi, the New Delhi Municipal Council and the Delhi Cantonment Board. In addition, the informal sector handles a large share of waste and plays a vital role in recovering recyclables and reducing the amount of unsorted waste (Assocham & PwC India, 2017; Sambyal, 2017). Table S1 (supplementary material) highlights the roles of the key stakeholders in New Delhi's waste management sector. Waste management constitutes a significant share of a municipal budget (Kaza et al., 2018), and in India, SWM activities absorb about 5-25% of available budgetary resources each year (CPHEEO, 2016b). The Government of NCT of Delhi (2023) allocated about 8% of its municipal budget in 2023-24 to waste management.

4.3.2.5 Adverse Impacts of the current MSW management scenario

The Indian Ministry of Housing and Urban Affairs, through the Central Public Health & Environmental Engineering Organisation (CPHEEO), released a manual in 2016 outlining seven essential components for an MSW landfill site. These components include a liner system at the base and sides of the landfill, a leachate collection and control facility, and a facility for collecting and

controlling landfill gas (LFG) (CPHEEO, 2016a). The Okhla landfill is the only one equipped with LFG vents (Angmo and Shah, 2021), while Narela-Bawana has the only leachate collection system (Chaudhary et al., 2021b). Poorly maintained landfills that do not adhere to sanitary standards can pose severe health and environmental hazards. Leachate from landfills contains heavy metals such as chromium, cadmium, zinc, lead, and nickel, as well as high concentrations of organic matter containing coliform bacteria and fungi. This leachate can percolate down to the groundwater, aquifers, and surface water, posing a severe epidemiological hazard (Siddiqua et al., 2022; Vinti et al., 2021). A study on all four landfills discovered higher than standard Leachate Pollution Index and significant levels of carcinogenic heavy metals such as cadmium and lead (Chaudhary et al., 2021b). The improper segregation of domestic waste leads to high levels of organic waste being dumped in landfills. When this waste decomposes, it releases methane, a powerful GHG inducing global warming (Gonzalez-Valencia et al., 2016; Manuja et al., 2018; Singh et al., 2018). According to Crippa et al. (2023), waste accounts for about 6% of India's total GHG emissions. Unfortunately, methane recovery is not a common practice in Delhi's dumpsites, and the great energy potential derived from it remains largely unseized (Ghosh et al., 2019; Singh et al., 2016).

In addition to contributing to global warming, uncontrolled waste disposal also poses other hazards. For instance, self-ignited or intentional fires, beyond worsening air quality and posing significant risks to human health – are hard to manage (Chaudhary et al., 2021a). A fire breakout at the Ghazipur landfill in November 2020 took almost 24 hours to be extinguished (Rajput, 2020). Similar fire breakouts in Delhi's landfills have become common in the last few years and bring significant health impacts (Agarwal et al., 2020; Chavan et al., 2022; Sharma et al., 2024). Dumpsites with unconsolidated or steep slopes can also lead to slope collapse, which can contaminate surface water, cause land pollution, and pose a risk to human life.

Due to the limited land availability in Delhi, some landfill sites are still operational, even though they have far exceeded their maximum capacity. For example, the Ghazipur landfill reached a height of 65m, three times the allowed height of 20m. In 2017, part of the landfill mound collapsed, killing two people (Sunny, 2017). Over the last few years, public authorities have been looking into the potential offered by WtE plants to reduce waste volumes and reclaim landfill sites (DPCC, 2023b; Planning Commission, 2014). However, these plants are associated with emissions of

human toxic compounds (fly ash, furans and dioxins) and face strong opposition from residents and the scientific community due to their proximity to residential areas (Chand Malav et al., 2020).

According to the Indian Ministry of Urban Development, landfills should be at least 500m away from any inhabited area, and the area within that radius should be declared a "No-Development Buffer Zone" (CPHEEO, 2016a). Nevertheless, our satellite review revealed that settlements, and even hospitals, are located within 500m of the four landfills.

Despite some technology-driven solutions available (e.g. WtE plants, compost plants and recycling plants), the waste dump rate in the NCT of Delhi supersedes the treatment rate by a large factor. Recent data from the Delhi Pollution Control Committee (2023) shows that over 35% (4,000 TPD) of the total waste collected is not processed and ends up in landfills. With an estimated annual population growth rate of more than 2.7% (UNDESA, 2022) compounded with the 1.3% yearly growth rate of waste generation per capita (CPHEEO, 2016a), the burden of waste is bound to increase by more than 4% every year.

Finally, the excessive saturation of the limited available land carries economic impacts that go beyond the aesthetic factors, the resulting odour, emissions, and groundwater contamination. Uncontrolled landfilling is a significant deterrent to the real estate market in the surrounding area. Properties near a landfill have lower market demand, leading to slower capital appreciation. A study suggests that dumpsites negatively affect all properties' value within a 10 km radius (Abhyankar et al., 2023).

4.3.3 Methods and data

We employed a Life Cycle Assessment (LCA) to evaluate the environmental impact of Delhi's current and future SWM practices. The extensive literature review allowed us to identify actors, processes and data to define inputs, emissions, and environmental externalities associated with each process within the system boundaries. The inputs and outputs for the life cycle inventory data are derived from the literature and the Environmental Footprint database (Pretato *et al.*, 2020), which contains secondary datasets from the European Commission's Single Market for Green Products initiative.

The Life Cycle Impact Assessment (LCIA) was performed by monitoring ReCiPe midpoint and endpoint levels while adopting the hierarchy perspective and related characterisation factors

(Huijbregts *et al.*, 2016). We include the inventory data and characterisation factors for each process in the supplementary material file.

4.3.3.1 System Boundary

The system boundary encompasses the entire process of managing MSW, from waste collection from dhalaos or intermediary collection points to its final treatment, also known as gate-to-grave. This boundary includes the transportation of MSW from secondary collection centres to their final destination and the processes by which MSW is typically processed at landfill sites, namely WtE, landfilling, and WtC (see Figure 34).

4.3.3.2 Simulation scenarios

The study conducts three comparative analyses to simulate MSW policy and practice changes. The first analysis concerns the current SWM practices and reflects the quantities of MSW currently processed at each waste treatment facility. The analysis includes a simulation of the status quo in 2022-2023 (i.e., the baseline scenario) and two future scenarios (S1 and S2) that project the waste generation corresponding to the population growth for 2030 and 2050, respectively. In this first analysis, there are no changes in the system, and the proportion of waste going towards each waste management process remains the same. This first run helps us identify the magnitude of future environmental impacts associated with unchanged SWM practices within the context of urban population growth.

A second analysis compares the baseline with two additional scenarios (S3 and S4). These scenarios assume higher segregation at the source, decentralised composting at the domestic or community level, and efficient collection and recovery of recyclables. The idea behind these scenarios is that waste segregation will improve recycling rates and the availability of biodegradable waste suitable for composting. This will reduce the amount of MSW collected and transported by 12%. We also assumed that this would reduce the share of mixed MSW going to landfills (5%-10%) and WtE plants (3%-18%) and increase the share of MSW for composting (8.5%-38.5%) (Figure 34). These changes are feasible given that the composition of biodegradables in Delhi is nearly 70% while the current share of MSW going to compost plants is only 11.5% (DPCC, 2023a). Comparing the baseline with the two alternative scenarios helps understand the differences in impacts that could

be achieved by changing the share of MSW going into each process. Under each scenario, we quantify the emissions to soil, freshwater, and air within each process.

Finally, the scenarios forecasting an increase in waste due to population growth (S1 and S2) and those assuming a change in processing rates (S3 and S4) are combined into two additional scenarios (S5 and S6). The LCA of these last scenarios helps quantify the impact reductions achieved by 2030 and 2050 by diverting compostable MSW from landfills and WtE plants to composting. Table 60 describes the amount of solid waste generated and the share of waste going to each waste management process for the baseline and six scenarios.

Figure 34: System boundary for the different waste management scenarios

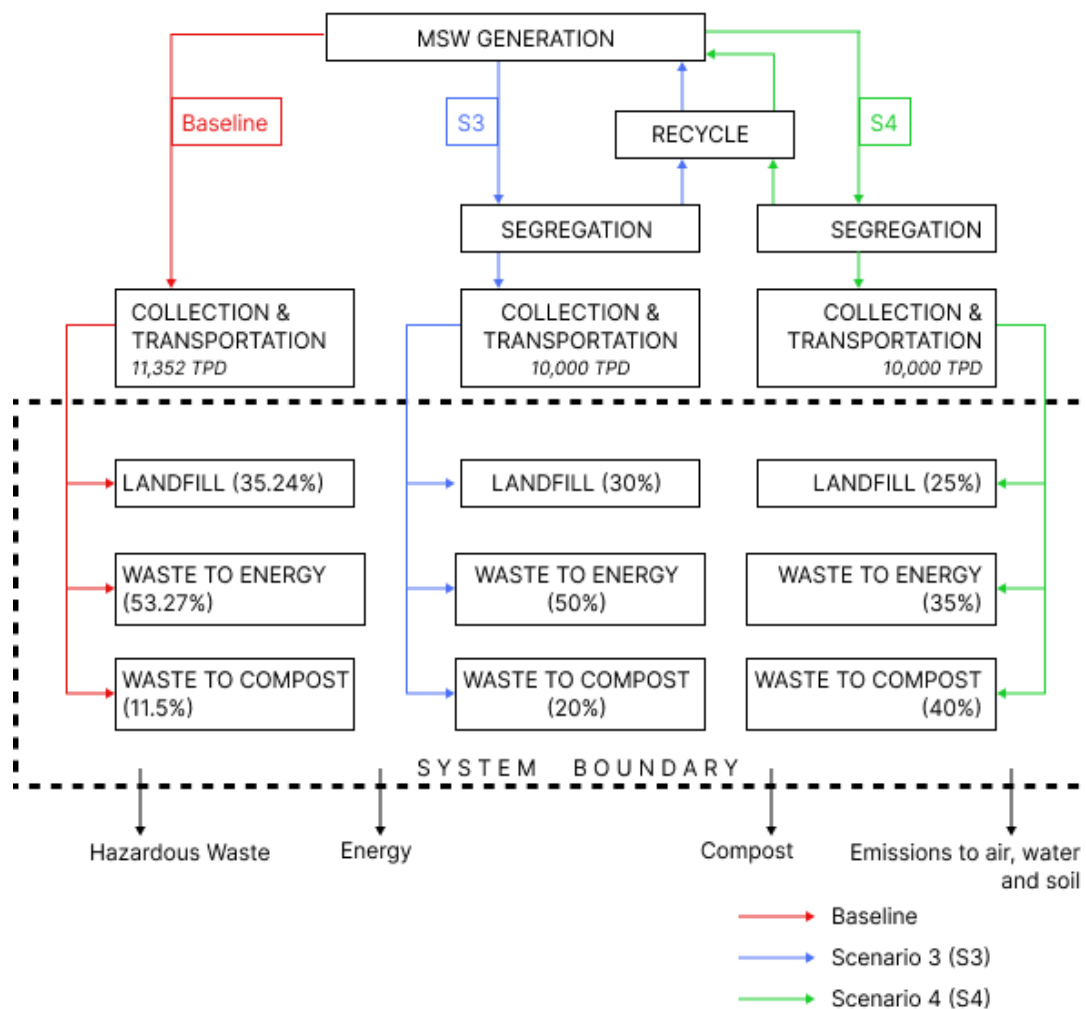


Table 60: Simulation scenarios*

SCENARIO	Population	Total MSW - TPD	Compost - TPD (share)	Landfill - TPD (share)	WtE - TPD (share)	WtC - TPD (share)
Baseline (2022-2023)	21.4 million	11,352	-	4,000 (35.2%)	6,047 (53.3%)	1,305 (11.5%)
S1 – BAU 2030	24.6 million ¹³	12,300 ¹⁴	-	6,177 (35.2%)	9,354 (53.3%)	2,018 (11.5%)
S2 – BAU 2050	36.5 million ³	18,250	-	10,581 (35.2%)	16,022 (53.3%);	3,457 (11.5%)
S3	21.4 million	10,000	2,000 (20%)	3,000 (30%)	5,000 (50%)	-
S4	21.4 million	10,000	4,000 (40%) ¹⁵	2,500 (25%)	3,500 (35%)	-
S5 [2030]	24.6 million	12,300	3,510 (20%)	5,265 (30%)	8,775 (50%)	-
S6 [2050]	36.5 million	18,250	12,024 (40%)	7,515 (25%)	10,521 (35%)	-

*some of the values are rounded to facilitate the reading

4.3.3.3 Life-cycle costing

The process of estimating all costs incurred over a product or service's lifetime is called Life-Cycle Costing (LCC). The LCC of dumpsites includes the initial investment cost and the present values of development costs, operating and maintenance costs, closure and post-closure costs and residual value (Martinez-Sanchez, Kromann and Astrup, 2015; Muthu, 2023). Thus, we used the following formula to estimate the life cycle costs of a landfill in New Delhi:

$$LCC = C_0 + \frac{D}{1+i} + \sum_{t=1}^n \frac{OC}{(1+i)^t} + \sum_{t=1}^n \frac{CL}{(1+i)^t} + \sum_{j=n+1}^m \frac{PC}{(1+i)^j} - \sum_{t=1}^n \frac{SV}{(1+i)^t}$$

Where C_0 is the Initial investment, D is the Development cost, OC is the Operational and management costs, CL is the Closure cost, PC is the Post-closure cost and maintenance, SV is the

¹³ Calculated at an annual compound rate of 2%

¹⁴ Considering per capita waste generation at 500 grams per day.

¹⁵ As per DPCC, 45-50% of MSW is biodegradable, hence 40% seems a reasonable estimation. Source: <https://www.dpcc.delhigovt.nic.in/uploads/pdf/SWMCPT1pdf-0580c09121d08b4d24f254273ace8fc7.pdf> (p7).

Salvage value, i is the real interest rate on capital, and j is the time horizon for operation and post-closure maintenance, respectively.

Based on data from the World Bank (2024a), we estimated that the real interest rate in India (average 2017-2021) equals 4%. This value accurately reflects the opportunity cost of capital for India in financing landfill sites rather than other profitable investments. Further, the estimate is consistent with the average value of long-term deposit interest rates ranging from 3.0% to 7% and the 7% coupon rate of long-term government bonds (Reserve Bank of India, 2024). It is assumed that the development of a landfill site occurs in the first year of its life cycle and takes one year to complete the development and closure.

4.3.3.4 Assumptions and Limitations

In our research, we analyse the current performance of the waste management sector dealing with MSW in the NCT of Delhi, which is the shared responsibility of local urban bodies. Hence, we are only considering the waste collected and managed by the formal sector, excluding the informal sector, which is responsible for recycling before the secondary collection. We have excluded the informal sector from our study because we lacked reliable data on the amount of recyclable waste segregated and recycled by it, the waste categories (such as paper, plastics, metals), and their emission factors.

Secondly, for the LCIA, each process is considered a black box model (i.e., all inputs are converted to outputs), which considers only the inputs and emissions relevant to the waste treatment process. The study does not consider the inventory for the life cycle of each facility (such as material inputs for a WtE plant, land for a landfill, etc.) since it would be the same for all scenarios under comparison.

Thirdly, we employed secondary data from the LCIA inventory database and the available literature as we lacked primary data on emissions from the waste processing and treatment on site. Nevertheless, we triangulated all data from the various sources to ensure a comprehensive inventory and avoid anomalies in emission quantities.

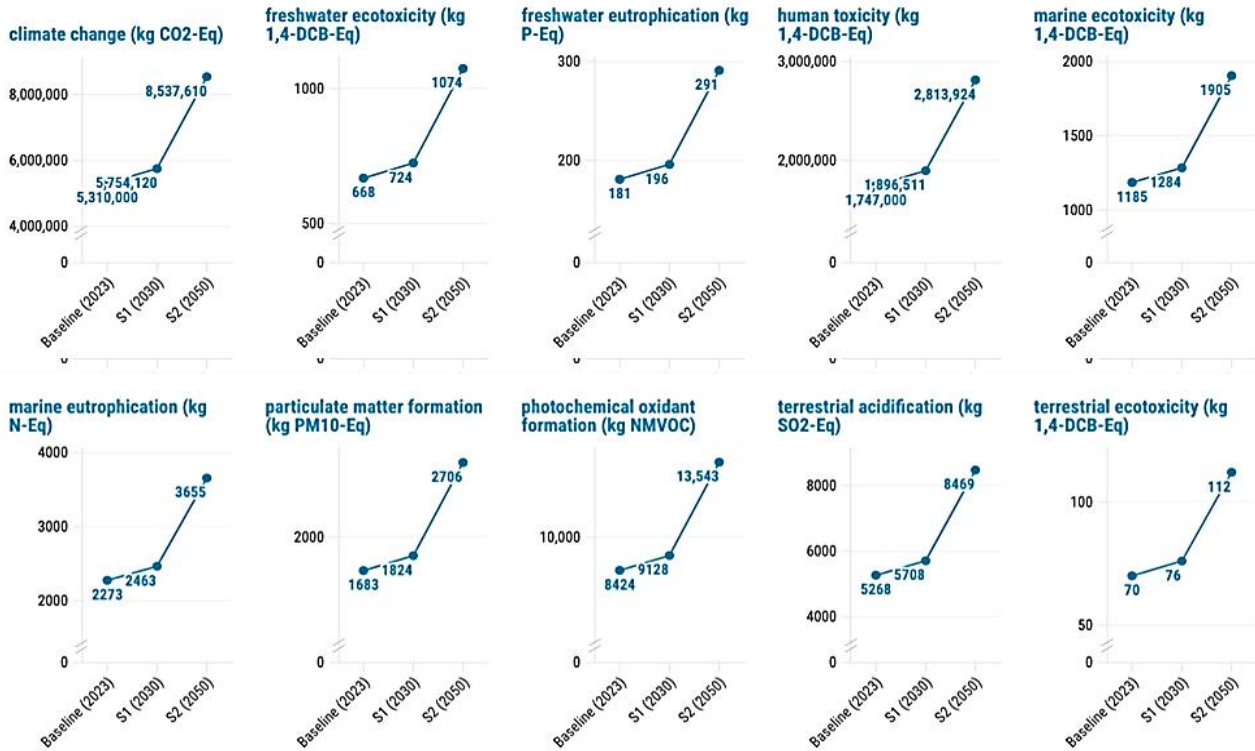
4.3.4 Results and discussion

4.3.4.1 Environmental impact

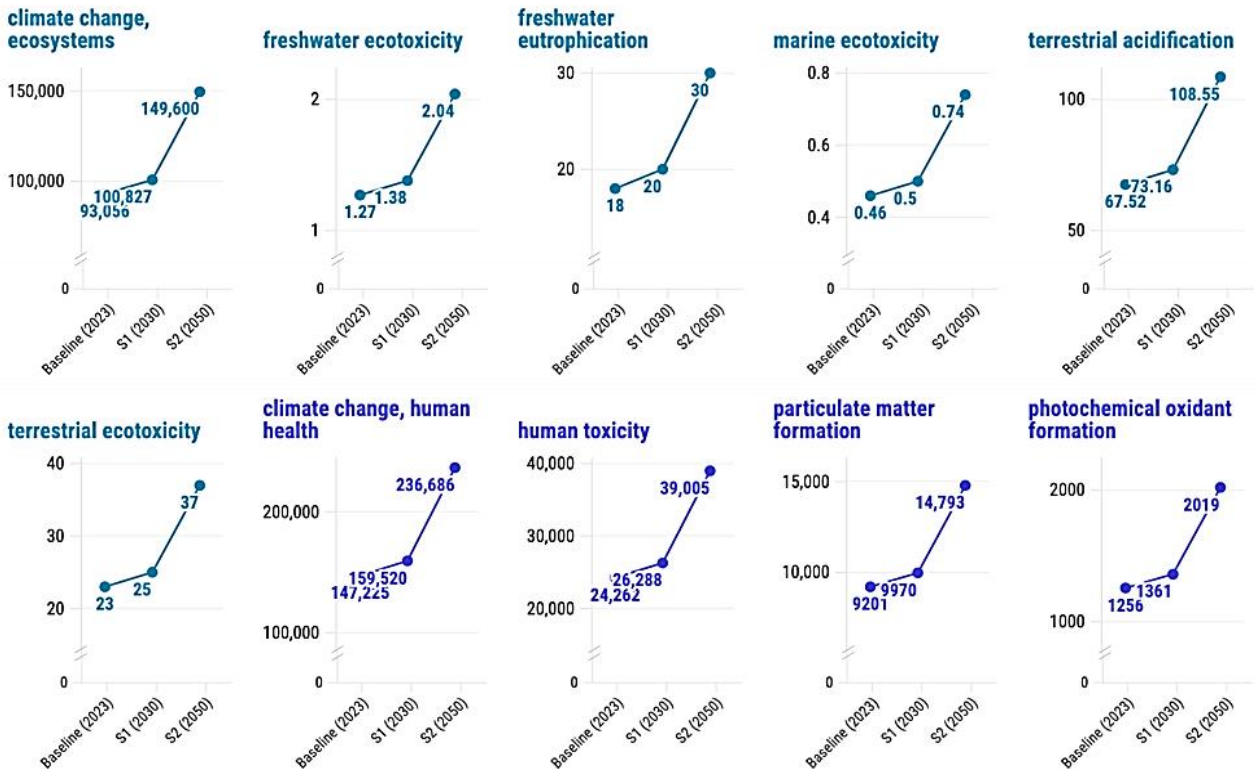
The following analysis compares the environmental impact of municipal waste management for three different scenarios: the baseline, S1, and S2. The LCIA results show that the overall environmental footprint of the waste management system is likely to increase by 8.4% by 2030. This increase is concerning, but even more alarming is that the impact is predicted to surge up to 60.7% in 2050 – compared to the baseline – under the current MSW practices (Figure 35).

Figure 35: Projected impacts on human health and ecosystem quality by 2030 and 2050

Midpoint results



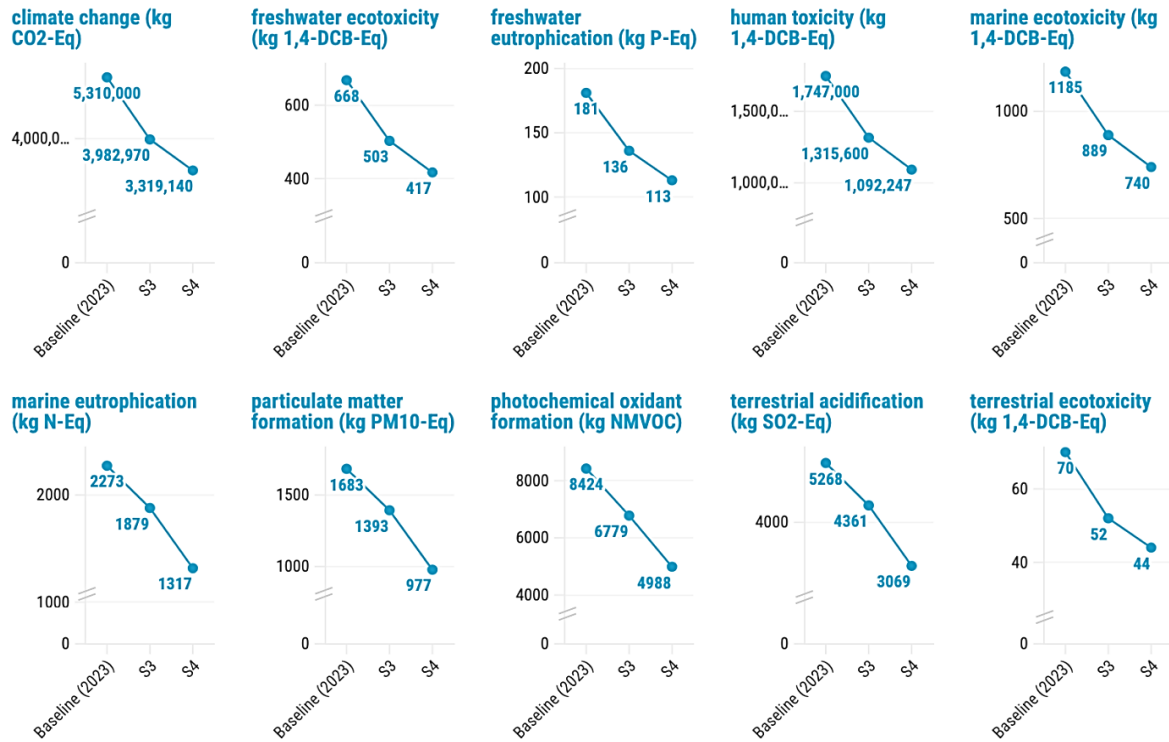
Endpoint results



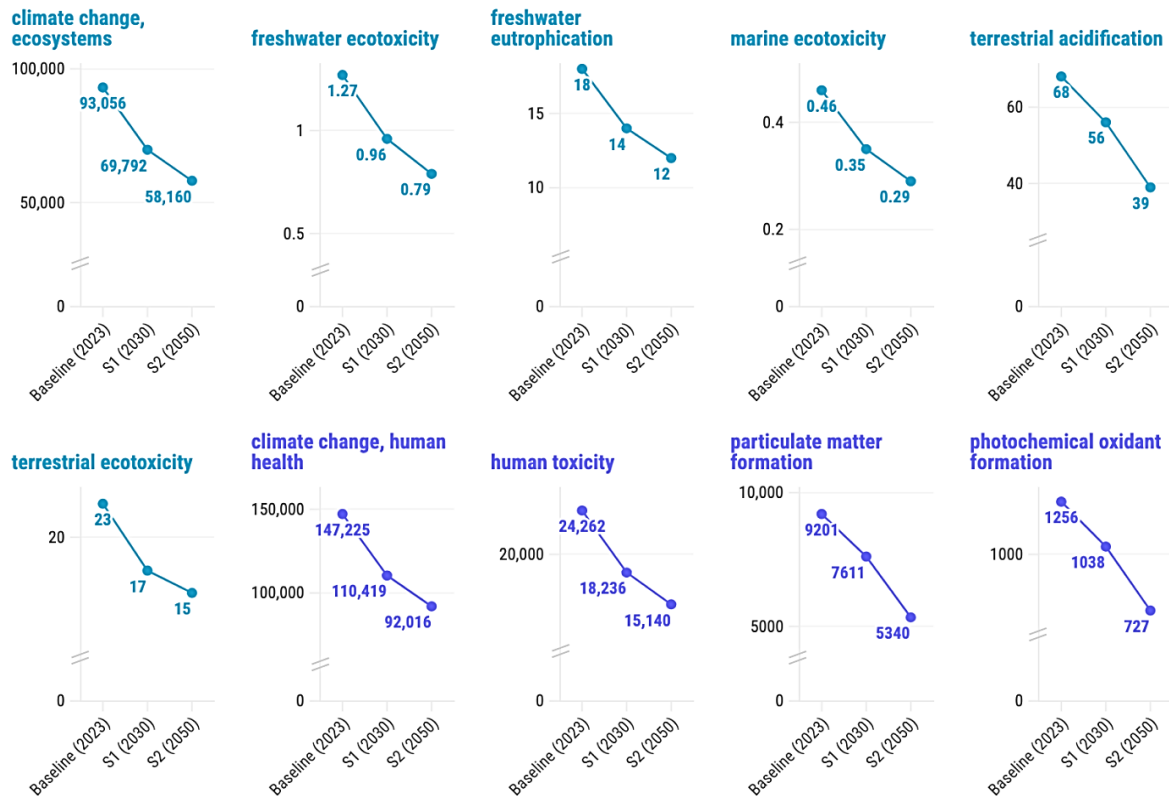
Next, we examined the changes in environmental impact between the baseline scenario and the S3 and S4 scenarios. These scenarios assume an increased diversion of compostable waste from landfill sites and WtE plants towards WtC facilities. Compared to the baseline, the S3 and S4 scenarios reduce the environmental footprint between 25% and 42% across various impact categories (Figure 36). The most significant impact reduction of over 40% is seen in ecosystem quality (terrestrial acidification) and human health (particulate matter formation and photochemical oxidant formation). This demonstrates that implementing policies and strategies to reduce landfill and WtE plant waste while increasing composting can lead to significant environmental improvements.

Figure 36: Impacts of S3 and S4 compared to baseline

Midpoint Results

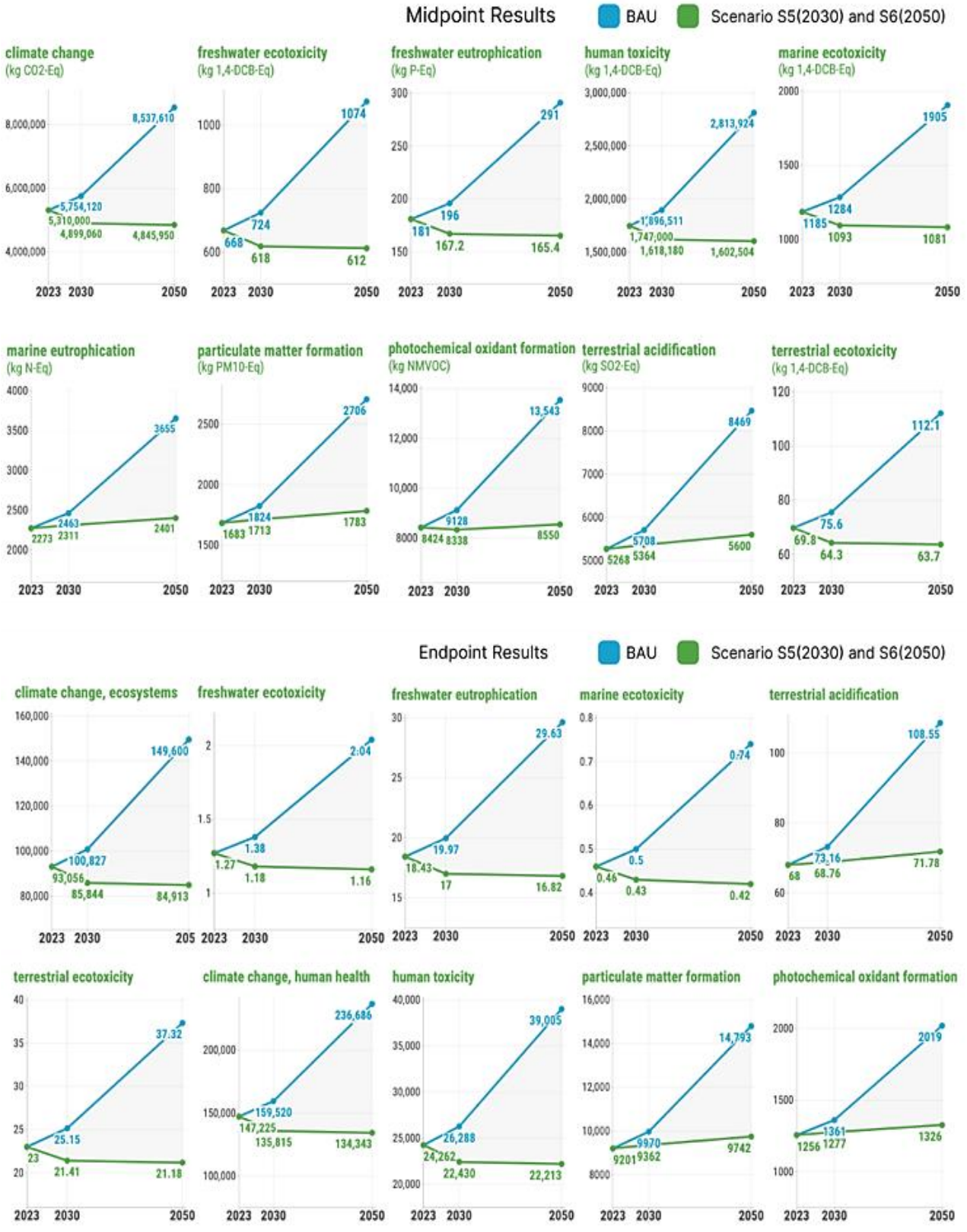


Endpoint Results



The last analysis combines the previous scenarios. It shows the result of reducing biodegradable waste going to landfills, WtE and WtC plants under the population projections for 2030 and 2050. The findings indicate that these strategies can help reduce the impact on indicators by an average of 14% in 2030 compared to the baseline. Further efficiency in diverting waste can further reduce the impact to around 42% in 2050, compared to the baseline (Figure 37). The results show an improvement in all indicators, especially in the long run, with a substantial decrease in human toxicity and terrestrial ecotoxicity (43%). More importantly, most indicators show a reduction from their initial baseline values. This suggests that the above strategy can potentially improve the current situation and successfully restore the ecosystem's capacity in various domains.

Figure 37: Impact reductions in S5 (2030) and S6 (2050), compared to 2030 and 2050 baselines



4.3.4.2 Financial costs associated with a landfill

In addition to the extensive environmental impacts and effects on human health due to leachate and LFG emissions, dumpsites pose a more physical feasibility challenge in the form of land availability and capital investment. The landfills in New Delhi are spread across 70 acres, on average. As per the circulars by the Land and Development Office (L&DO), the institutional rate of one acre of land in Delhi is valued between R\$ 3.85 – 8.8 million (US\$ 46,444 – US\$ 106,158)¹⁶ depending on the city area (L&DO, 2000). Per the latest circle rates for residential and commercial land in Delhi, an acre of land is worth between R\$ 18 – 114.5 million (US\$ 217,000 – US\$ 1.38 million) (L&DO, 2017a, 2017b). These values represent lower-bound estimates in less prestigious zones. High-end residential and commercial zones can be valued up to R\$ 168 million (US\$ 2 million) and R\$ 412 million (US\$ 4.97 million) per acre, respectively. Thus, while the average institutional rate for one acre of land in NCT Delhi amounts to R\$ 6.3 million (US\$ 76,300), the opportunity cost of land, as signalled by the market and commercial rates, is much higher than that.

The land acquisition constitutes a large share of the total investment cost. However, landfills require considerable additional costs throughout their life cycle, including operation and post-operational maintenance. Table 61 indicates the average costs associated with a 70-acre landfill.

Table 61: Average financial costs of a 70-acre landfill in NCT Delhi

no.	Item	Cost (R\$ million, 2024)	Cost (US\$ 2024)
1.	Site selection and site characterisation cost	11.3	136,778
2.	Design and detailed engineering cost	7.4	89,048
3.	Site development cost (incl. land acquisition)	569.1	6,865,863
4.	Development cost (annual)	180.2	2,174,046
5.	Operation and maintenance cost (annual)	69.4	838,324
6.	Closure cost (annual)	74.2	895,315
7.	Post-closure maintenance cost (annual)	15.6	188,273

Source: Own estimate from data of Ministry of Urban Development (2018), CPHEEO (2000, 2016)

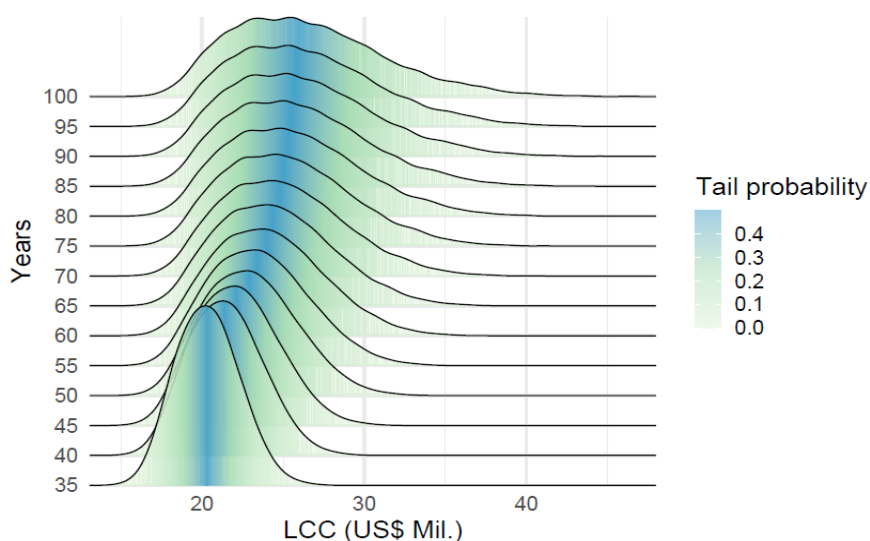
¹⁶ Conversion rate of 1 INR = 0.01206 USD as of March 4, 2024, Source: xe.com

The operational life of a landfill in India should last a minimum of 20-25 years, while its post-closure should last a minimum of 15 years (CPHEEO, 2016). The designated landfills of Ghazipur, Bhalswa and Okhla, established between 1984 and 1994, have been operational well beyond their stipulated timeline and capacities. Today, they receive three times the amount of waste initially designated (Khan *et al.*, 2022). The half-life of MSW in a landfill could last from several years – such as 10.6 years estimated for the Okhla landfill site – to several decades, while LFG emission could continue for a very long time in its post-closure phase (Manuja, Ahmad and Haq, 2018). In other words, a landfill site could take several decades (30-50 years) before it can be rehabilitated upon closure.

We used the formula shown in section 2.3 to estimate the landfill LCC over several years and performed a Monte Carlo simulation analysis on the results to control for uncertainty and variation in cost estimates and discount rates. In our simulation, we utilised detailed cost data to model the probability distribution functions for key variables. Through bootstrapping and fitting, we determined that investment operation and maintenance cost variables followed normal and log-normal distribution functions. In contrast, the discount rate variable was associated with the triangular distribution function.

The results shown in Figure 38 indicate that, on average, the LCC of a 70-acre landfill ranges between US\$ 18-27 mil., depending on its lifespan. This demonstrates that apart from the significant environmental externalities, landfills are also associated with substantial financial outlays.

Figure 38: Monte Carlo simulation results on landfill LCC



4.3.5 Discussion

4.3.5.1 Environmental emissions and financial costs of Landfills

The landfill sites in Delhi are not equipped with leachate and LFG collection systems (Gupta and Arora, 2016; Manuja, Ahmad and Haq, 2018; Talyan, Dahiya and Sreekrishnan, 2008). The results of LCIA in this study demonstrate that landfills are responsible for high GHG emission levels and significant negative impacts on human health. Photochemical oxidation of volatile organic compounds and carbon monoxide in the presence of nitrogen oxides and sunlight results in the creation of photochemical ozone often associated with human respiratory hazards (Chaudhary, Nain and Kumar, 2021).

Our study indicates that changing SWM practices can substantially reduce hazards to human health and the environment. Furthermore, the high life cycle costs and the high cost of land in Delhi challenge the financial viability of landfill sites. Despite the high environmental and financial costs, another landfill site has been identified, and its development is underway (DPCC, 2023a).

4.3.5.2 Technical, environmental and economic issues of WtE plants

Without any segregation at source, incineration-based WtE plants seem to provide an easy solution to eliminate the bulk of the waste. Public authorities, both locally and nationally, are keen on exploring the potentials of WtE plants (DPCC, 2023b) despite the significantly high environmental and health issues they bring – as shown in our results. WtE technology can undoubtedly be advantageous over alternative options like open dumping (Fernández-González *et al.*, 2017). However, they still represent a highly inefficient and polluting energy source (Tangri, 2023). Results from WtE plants in the EU report a 25% efficiency as opposed to 35% and 55% for coal fire energy plants and combined cycle plants turbine plants, respectively (Hogg, 2023).

Moreover, WtE plants require substantial economic investment and the right feedstock quality to run properly (UNEP, 2019). India's waste is not a suitable feedstock for energy production due to its high moisture level and low combustible content (ISWA, 2022). The lower calorific waste feeding into the WtE plants results in higher toxicity of emissions due to dioxins and furans release and even lower technical-economic efficiency, which in turn hampers operations' sustainability (Devraj and Sambyal, 2019; UNEP, 2019). In the past, WtE plants across India have been shut down for the same issues still affecting New Delhi today, namely: lack of waste segregation, improper

collection methods, unsuitable waste quality, litigation issues, pollution, inadequate financial support and lack of technology (Chand Malav *et al.*, 2020; Kalyani and Pandey, 2014; Pujara *et al.*, 2019).

Thus, the modernisation approach of the MSW sector in Delhi needs to be rethought, focusing less on WtE technological solutions and more on addressing social dynamics and behavioural change (Bercegol and Gowda, 2019; Randhawa *et al.*, 2020). If local authorities are willing to seize the opportunities offered by WtE technologies, they must first ensure that waste is segregated, collected and transported correctly and that efficient composting and recycling facilities are in place.

4.3.5.3 Underutilised potential of composting

We have shown that compost plants have the least environmental impact (per tonne of MSW) compared to WtE and landfills. However, compost plants receive only 3% of the total waste collected, even though biodegradables generally constitute more than 50% of MSW in New Delhi (Figure 33). This is mainly due to the lack of segregation at the source, which leads to the mixed waste composition. In addition, compost plants' capacities are considerably low compared to the share of compostable material in MSW.

Currently, only two of the three composting plants in Delhi operate, and they process about 400 TPD, running at 60% of their operational capacity (Khan *et al.*, 2022). It is evident that more compost plants need to be established to manage the 7,000 tonnes of biodegradable material that the Delhi population produces each day. The latter is a sensible strategy to help process a significant share of MSW and reduce the burden of capital investment and environmental externalities associated with landfills and WtE plants.

However, this alone would not be enough if it is not coupled with a 'stick and carrot' approach to ensure compliance with waste segregation measures through awareness campaigns and by-laws (Manuja, Pandey and Gulati, 2019). Increasing segregation, especially of biodegradable and recyclable material, should become the social norm in how society disposes of its waste.

Thus, the efforts already made in promoting behavioural change should be scaled up to ensure that citizens and decision-makers understand the relevance of their participation in resolving the SWM issue (Mani and Singh, 2016). Building upon best practices already implemented in other

Indian cities (Atin *et al.*, 2021), an increase in waste segregation can be obtained through doorstep collection carried out by local bodies, NGOs or even associations of informal waste pickers (Agarwal *et al.*, 2005; Mani and Singh, 2016). The formalisation of waste pickers has been consistently reported in the literature as a win-win strategy whereby the 200,000 waste pickers currently working in New Delhi (Chaturvedi and Gidwani, 2011) could see their income and livelihoods secured while helping segregate waste and reduce the load of waste collection and transportation (Atin *et al.*, 2021; Bercegol and Gowda, 2019; Joshi and Ahmed, 2016; Randhawa *et al.*, 2020).

4.3.5.4 Potentials of Circularity in SWM

At the COP26 summit held in Glasgow, India pledged to achieve net-zero emissions by 2070 and to fulfil 50% of its energy needs through renewable sources by 2030 (WEF, Kearney and ORF, 2021). Similarly, in the joint declaration from the latest G20 held in Delhi, world leaders renewed their commitment to reduce waste by 2030 substantially, highlighting CE's critical role in decoupling economic growth from environmental pressure (G20 Secretariat, 2023).

Nevertheless, India has been increasingly looking into the potential of WtE technology to reduce its waste. As a result, several cities in the country followed the dominant narrative that posited WtE technology as a total solution to waste issues and a sustainable energy source (Randhawa *et al.*, 2020). However, this narrative undermines and contradicts India's carbon neutrality strategy as it ignores the high environmental externalities produced by WtE plants (Tangri, 2023) – as demonstrated in our analysis (see S1 and S2 results).

To achieve its net-zero targets, India needs to take rigorous steps towards achieving circularity in material flows and reducing waste. The idea of circularity is based on a closed-loop system, and MSW must be treated as a resource to achieve a closed loop. At present, neither are there policies in place that embed value in waste, nor does there exist a robust market where waste could be effectively seen as a resource in India.

The SWM approach in Delhi is still based on a linear system, where almost all the waste goes to the designated landfill sites for its end-of-life processing. Given the large population of Delhi, the current four dumpsites – three of which have already exceeded their operational life and capacity – are insufficient to handle the amount of waste generated. As also shown in our results, the current SWM approach brings significant financial, environmental, and health damage.

Moreover, Delhi's landfills lack basic collection systems to control emissions and leakages of harmful substances. It is estimated that LFG from landfills in Delhi has grown exponentially in recent years, increasing health hazards and environmental contamination (Chaudhary, Nain and Kumar, 2021; Ghosh, Gupta and Thakur, 2015). Collecting LFG and recovering its methane content can simultaneously reduce air and soil emissions and provide clean fuel that supplements the low-calorific value of refuse-derived fuel at WtE plants and in many other industries requiring methane (Ghosh *et al.*, 2019). The same could happen for fly ash, a hazardous residue of WtE plants that must be disposed of in appropriate landfill sites (Lausselet *et al.*, 2016). Upon its inertisation, fly ash could be used as input in the construction sector, for instance, as an additive to enhance cement strength and quality (Bontempi *et al.*, 2010).

With the emergence of WtE technology, MSW is certainly a valuable input in generating refuse-derived fuel (ISWA, 2022). However, as already discussed, efficiency remains low, and without standardised quality control and proper waste segregation, low calorific values of MSW often result in higher plant emissions and residues, bringing about substantial environmental and health hazards.

Our results suggest that a more efficient strategy is expanding the capacity and outreach of WtC plants to support the entire waste value chain. After all, our suggestions largely align with the recommendations made by the WtE task force of the Planning Commission (2014). The latter advocated for an integrated approach to SWM, including awareness campaigns on segregation and waste reduction, the integration of waste picker organisations to enhance their capacity and contribution to the informal value chain of recyclable waste, the use of appropriate technologies, and decentralised or centralised approaches to waste management. The Task Force also strongly backed the adherence to the 5R principle of Reduce, Reuse, Recover, Recycle and Remanufacture, which aligns with the core concept of CE.

4.3.6 Conclusions

This study provides an innovative and comprehensive analysis, along with practical solutions, for waste management in Delhi. It fills critical gaps in the current literature by assessing the economic and environmental impacts of different waste management scenarios and offering policy recommendations to optimise waste management processes. By promoting circular economy

principles and integrating the informal sector, the study paves the way for a more sustainable and efficient waste management system in megacities.

The urban local bodies in Delhi's NCT face the arduous challenge of managing 11,352 TPD of MSW, and the current approach is largely ineffective in metabolising the generated waste. The existing facilities are oversaturated, and while the authorities are looking for a new landfill site, there is a dire need to readjust and optimise the system. Although about 70% of MSW comprises recyclable or biodegradable materials, the most common approach is sending all waste to landfills or incineration without segregation at the source. This method receives 97% of all collected waste, while only a meagre 3% is sent to compost plants. However, we have demonstrated that the MSW processing facilities are associated with high financial costs and environmental impact.

Moreover, two of the four Delhi landfills have neither a leachate collection system nor an LFG collection system. We have also indicated how landfills' leachate and LFG (containing methane, carbon dioxide, nitrous oxides, etc.) contribute to freshwater eutrophication, climate change and human health damage. Emissions and hazardous residues from WtE plants are associated with staggering environmental impact, as shown by the indicators of particulate matter, acidification, human toxicity, ecotoxicity and freshwater eutrophication. This study is the first of its kind to offer a comprehensive comparison of emissions from various waste disposal methods, including recycling, incineration, and upcycling for infrastructure construction. By utilising the latest data on waste generation, population, and cost estimates, the study accurately represents the current situation and identifies areas for improvement.

Despite numerous campaigns promoting segregation at source, little progress has been achieved. This could be seen as a social challenge that could be tackled only by changing the mind-set of how Delhi citizens generate and manage their waste. Our results suggest that environmental impacts can be substantially reduced by increasing the share of waste sent to compost plants. Similarly, the total MSW load is reduced through segregation at the source and the recovery of recyclables. This underscores the importance of waste segregation, increasing compost plant capacity, and exploring decentralised and environmentally viable solutions.

The literature review also highlighted the need to integrate the informal sector of waste picker organisations, intermediaries, and dealers to reduce the burden of recyclables ending up in

landfills. Further research is needed to obtain and integrate data concerning the nature of recyclables collected and the amount of waste recovered by the informal recycling sector in Delhi.

In summary, adopting circularity in the waste management sector can significantly reduce the financial and environmental challenges associated with the current SWM practices. Furthermore, this approach improves future results and can potentially restore the quality of the urban environment to a level above and beyond its original state. Evidently, adopting a CE approach would help India to pursue, more consistently, its long-term carbon neutrality targets. However, the latter is a collective effort where households, municipalities, and private and informal sectors must collaborate to develop policies that promote responsible production, consumption, segregation at source, efficient recovery, and recycling.

5 Thesis conclusions and policy implications

The three case studies in this thesis address critical environmental challenges in different contexts: forestry and carbon neutrality in Cambodia, coastal ecosystems in SIDS, and waste management in Delhi. From the perspective of green growth, these studies offer valuable insights into how economic development can be harmonised with environmental sustainability. By examining theoretical and methodological contributions and the implications for the science-policy interface, we can draw comprehensive lessons to inform future green growth policies and strategies.

In detail, the research findings presented in this thesis and the extensive literature review suggest that current green growth efforts in developing countries fail to achieve the decarbonisation targets of the Paris Agreement. This is primarily attributed to a lack of decoupling effect between economic growth and carbon intensity. Moreover, technological progress in more advanced economies has yet to take off in developing economies. It also appears that around the world, growth is still a necessity that largely prevails over environmental conservation, particularly in developing countries, where the priority is still to lift a large share of the population out of poverty.

From a theoretical perspective, the results highlight the non-linear relationship between economic growth and emissions, emphasising that intricate backward and forward linkages between sectors influence sectoral growth and GHG emissions. This complexity challenges the traditional linear models of economic-environmental interactions and underscores the need for more sophisticated approaches in policy formulation and analysis.

Moreover, the lack of a comprehensive policy mix, over-reliance on technology, and disregard for behavioural insights in policy-making hinder our progress towards carbon neutrality. In fact, the research suggests that solely focusing on technological change is insufficient for rapid decarbonisation. In addition to technological advancements, there must also be a shift in behaviour towards adopting low-carbon energy technologies and sustainable energy practices. Incorporating behavioural insights into green growth policy-making is not just a suggestion but a necessity to enhance the current approach that addresses market failures. Green growth policies can better address climate externality and other market failures when socio-demographic and psychological factors are considered.

From a methodological perspective, this thesis reaffirms the value of quantitative tools in supporting evidence-based policy-making. Combining economic evaluation methods like those proposed here provides a novel and more robust approach to estimating the relevance and monetary value of environmental services while increasing the precision and reliability of environmental assessment of economic development policy. Presenting precise data and projections can also bridge the gap between scientific research and practical policy formulation. However, the case studies jointly concluded that standardisation in data collection and integration of environmental data is needed to ensure global policy coherence and enhance the accuracy, effectiveness and comparability of policy interventions. Meanwhile, policymakers can leverage the results from studies like those presented here to prepare more effective and better-informed policies.

Finally, this thesis also contributes to our understanding of green growth, specifically in the context of developing countries. The research suggests that the current approach to separating CO₂ emissions from GDP may not be feasible. Credible efforts to meet Paris targets may need radical changes in how modern societies work, produce and consume. Hence, green policies should start harmonising some of the principles of green growth with ideas from a post-growth society. Ultimately, this may support the efforts made thus far and accelerate progress towards truly achieving Paris targets before it becomes too late.

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