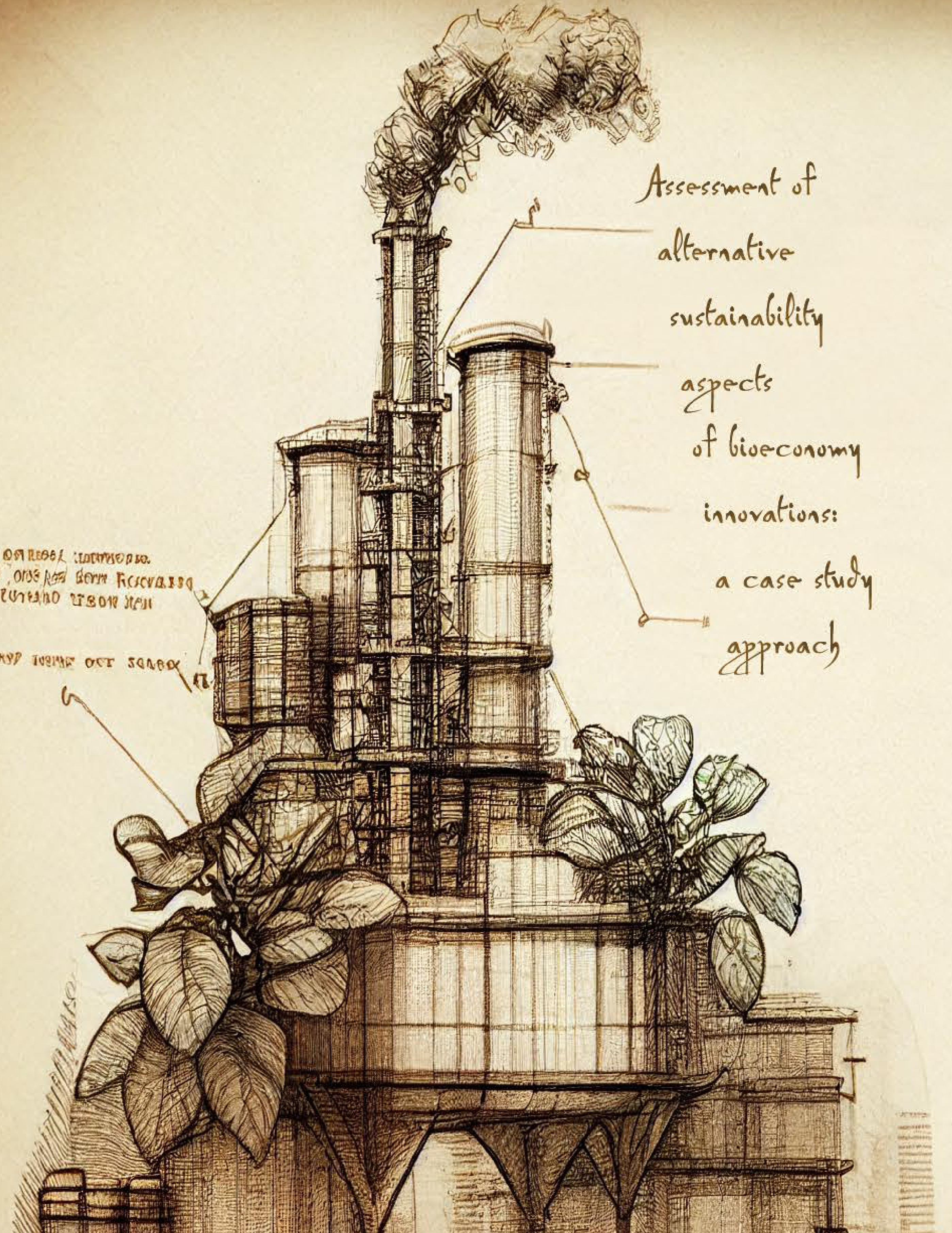


Assessment of alternative sustainability aspects of bioeconomy innovations: a case study approach

Gianmaria Tassinari

Propositions

1. Without systemizing, case studies contribute to the bioeconomy literature in an incoherent way.
(this thesis)

2. Feedstock availability is the most influential factor in the profitability of bio-based packaging film.
(this thesis)

3. The transition to a sustainable and circular economy requires a new official national accounting system.

4. The Green Revolution made food systems worse.

5. Resuming food commodity imports from Russia and Ukraine to avoid food insecurity is incorrect.

6. Precautionary limits of acceptable risk are determined politically.

Propositions belonging to the thesis, entitled

Assessment of Alternative Sustainability Aspects of Bioeconomy Innovations: A Case Study Approach

Gianmaria Tassinari
Wageningen, 12 April 2023

Assessment of Alternative Sustainability Aspects of Bioeconomy Innovations: A Case Study Approach

Gianmaria Tassinari

Thesis committee

Promotor:
Prof. Dr J.H.H. Wesseler
Professor of Agricultural Economics and Rural Policy
Wageningen University & Research

Co-promotors:
Dr D. Drabik
Associate professor, Agricultural Economics and Rural Policy Group
Wageningen University & Research

Dr. C. Soregaroli
Associate professor, Department of Agricultural and Food Economics
Università Cattolica del Sacro Cuore

Other members:
Prof.Dr A.G.J.M. Oude Lansink, Wageningen University & Research
Prof.Dr V. Beckmann, Universität Greifswald, Germany
Dr G. Philippidis, CITA, Aragon, Spain
Dr R. M'Barek, European Commission, Joint Research Centre, Seville, Spain

This research was conducted under the auspices of Wageningen School of Social Sciences (WASS).

Assessment of Alternative Sustainability Aspects of Bioeconomy Innovations: A Case Study Approach

Gianmaria Tassinari

Thesis
submitted in fulfilment of the requirements for the degree of doctor at
Wageningen University
by the authority of the Rector Magnificus
Prof. Dr A.P.J. Mol,
in the presence of the
Thesis Committee appointed by the Academic Board
to be defended in public
on Wednesday 12 April 2023 at
at 1.30 p.m. in the Omnia Auditorium.

Gianmaria Tassinari
Assessment of Alternative Sustainability Aspects of Bioeconomy Innovations:
A Case Study Approach
118 pages.

PhD thesis, Wageningen University, Wageningen, the Netherlands (2023)
With references, with summary in English

ISBN 978-94-6447-611-8
DOI https://doi.org/10.18174/588938

"After all, tomorrow is another day"

Acknowledgements

I am grateful for the intense, and exciting journey that has brought me to this point. This journey has been made possible by the support and contributions of numerous individuals whom I wish to thank sincerely.

Firstly, I express my gratitude to the esteemed members of the thesis committee, namely Prof. Dr. A.G.J.M. Oude Lansink, Prof. Dr. V. Beckmann, Dr. G. Philippidis, and Dr. R. M'Barek. Your expertise and willingness to undertake this duty are highly appreciated. I hope that I have made your task as least tedious as possible.

I would also like to express my deepest admiration and gratitude to the promoter of this journey, Prof. Dr. Justsus Wesseler. His magical bibliographic memory, which he shared on every handy occasion, has been the most useful medicine for my doubts and uncertainties. Together, I would also like to express my sincere appreciation to the BioMonitor project. I hope that my own contributions have been meaningful and valuable to its consortium.

Next, I would like to express my sincere gratitude to Dr. Claudio Soregaroli and Dr. Dusan Drabik for their exceptional supervision during this journey. I acknowledge that their roles were very complicated and demanding with someone like me, and I am truly grateful for their unwavering support and guidance. Dr. Soregaroli, I am immensely grateful for the valuable insights and behind-the-scenes secrets you have shared with me during this journey. Your courage and gestures during our trip to Naples have inspired me and are woven into this thesis. I am deeply honoured to have had the opportunity to work with you. Dr. Drabik, you have been a trusted advisor and friend, and I cannot thank you enough for all that you have done for me. Like the Talking Cricket to Pinocchio, your guidance has been invaluable in shaping my professional and personal growth. I feel incredibly fortunate to have had you as my supervisor, and I am grateful for the hundreds of fulfilled wishes you have made possible. To you, my deepest gratitude and most loyal friendship.

I would like to express my gratitude to Max Kardung, partner in unspoken (dis)adventures, those that will remain just legendary tales. Max, I am grateful for your company. Without you, the bitter parts of this journey may have been harder to digest.

I would also like to acknowledge the others who shared similar experiences with me, Yan, Evert, Anastasia, Ema, Vineta, Tévézia, Patricia, Mohamed, Eko, Kasia, Muyinatu, Carlos, Melody, Frank, Liang and Clare; I express my sincere gratitude also to Alessandro and Paolo, whose exceptional culinary skills and dedication have enabled them to promote and preserve the rich Italian culinary tradition beyond the Alps. To those who still have a part of their journey ahead, I wish you good luck.

I would like to extend my heartfelt gratitude to the AEP Group family, including esteemed professors Dr. Liesbeth Dries, Dr. Esther Gehrke, Dr. Koos Gardebroek, Prof. Dr. Wim Heijman, Dr. Rico Ihle, Prof. Dr. Hans van Meijl, Dr. Jack Peerlings, postdocs Insa and Kutay, and supporting team members Dineke, Karen and Frank. I would like to express my special gratitude to Rico, whose ping-pong sessions have had a remarkable impact on maintaining my mental and physical health at sustainable levels.

On the other side of the continent, I express my gratitude to the SMEA group, Dr. Linda Arata, Dr. Elena Maria Bianco, Dr. Stefano Boccaletti, Dr. Gabriele Canali, Dr. Elena Castellari, Dr. Anwesha Chakrabarti, Dr. Stefano Gonano, Dr. Claudia Lanciotti, Marina Maggi, Prof. Dr. Daniele Moro, Prof. Dr. Daniele Rama, and Prof. Dr. Paolo Sckokai for providing me with the invaluable opportunities that have enabled my professional growth. I am particularly grateful to Maria Grazia, Emanuele, Ronny and Gonano for their outstanding efforts in creating a professional environment that fosters a sense of warmth and familiarity.

To my dear friends, Mirta and Alessandro, I am immensely grateful. Mirta is a brilliant mind and her courage has been a source of inspiration, while Alessandro, the Nobel Price Varacca, has been a constant support. They are the only ones who truly understand how this journey was, *Bello eh, Bellissimo, Però*. I am also thankful to my friends, Andreas, Lorena, Ryan, Laura, Valerio, Andrea, and Guido. Despite the distance between us, their mere thought has been instrumental in helping me overcome numerous obstacles.

Lastly, I would like to express my heartfelt gratitude to my Family, who made this journey possible by instilling in me the most fundamental values of life. To Anastasia, for the love and patience that she has shown me throughout these years, without which I would have lost the essence of my sacrifices. To Papà, for the cultural and creative foundation that contributed to my achievements. To Alice, for the boundless joy and bravery with which she taught me to confront life's challenges. And to Mamma, for dedicating her entire life to her family without ever expecting anything in return. This is all for you; I love you all so much!

Summary

The Earth's climate system continues to change at an unprecedented rate and ongoing research underscores the urgent need for rapid progress in global and national adaptation and mitigation efforts. The European Union (EU) aims to become the first climate-neutral bloc of countries by 2050 and aspires to achieve a modern and resource-efficient economy. With that goal, on 11 December 2019, the European Commission (EC) launched the ambitious "European Green Deal". To implement it, on 22 June 2020, the EC established the EU Taxonomy Regulation to ensure the efficient movement of capital toward truly green investments (based on the degree of sustainability of an activity). However, there is currently no consensus on what should be assessed and how, which lead to different interpretations and meanings of sustainability performance as well as information inefficiencies that complicate and impair investment into facilitating green transactions. This thesis aims to contribute, through innovative approaches and methods, to a more detailed and careful evaluation of alternative paths to sustainable innovation that apply technologies and social practices based on bio-based materials to transform or re-design conventional production systems (namely, circular bioeconomy systems).

Given the complexity and heterogeneity of bio-based systems, the literature contains several coexisting narratives on bioeconomy. After the introduction, the thesis begins by systematically reviewing the versatility of empirical studies—notably, case studies—along with the narratives surrounding the bioeconomy concept (i.e., an ecological economy, a science-based economy, and a biomass-based economy). The results of Chapter 1 provide an overview of how the narratives of the concept of bioeconomy affect the versatility of the case study research. Based on the low density of the illustrated semantic networks, we conclude that future empirical research on bio-based phenomena should be more transdisciplinary and rely more on cross-sectoral approaches. Further work is also required in developing common research protocols that support transparency and replicability of case studies in the bioeconomy.

Building on the knowledge gained from this literature review, the thesis discusses three concrete cases, with both scientific and practical implications, of sustainability performance assessments. These studies capture the heterogeneity and complexity of bio-based systems, which differ in terms of feedstock, level of technological maturity, and area of application.

Chapter 2 covers the application of biopolymers in food packaging. Bio-based polymers are increasingly attracting attention as a solution to reducing the consumption of non-renewable resources and curbing the accumulation of fossil-based plastic waste. This study analyzes the economics of a new packaging film based on a polylactic acid–polyhydroxybutyrate blend (PLA-PHB), with PHB obtained from agro-industrial residues (potato peels). We model various sizes of biorefineries using the new biotechnology in Europe. For a four-year payback period, which is generally accepted in the industry, the calculated minimum product selling price ranges from 9.7 euros per kilogram to 37.2 euros per kilogram, depending, among other factors, on the production capacity of the biorefinery. We have incorporated the uncertainty over the model parameters in a Monte Carlo simulation and investigated the relative impact of individual factors on the minimum product selling price. Overall, the results indicate that the bio-based feedstock availability is the most influential factor on the profitability of the new biotechnology.

Chapter 3 examines another bio-based product of contemporary relevance, bio-based fertilizers, whose production, unlike that of biopolymers, has a higher TRL. Using a hybrid input-output approach, we systematically traced sustainability footprints of a nutrient recovery strategy from sewage sludge. The results obtained were then compared with the most common landfilling practice. Overall, accounting for infinite upstream spill over effects, using sewage sludge for organic fertiliser production generates more jobs and reduces more greenhouse gas emissions than landfilling does. By contrast, landfilling stimulates the economy more and reduces energy carrier use more. Nevertheless, the environmental benefit is certain at a high-level of confidence, as estimated by a Monte Carlo simulation.

Chapter 4 reports an ex ante evaluation of different policy scenarios—investment cost subsidies and operating cost subsidies—that support bio-based drop-in solutions. In the absence of historical data for econometric analyses of innovative, emerging bio-based products, we build a stylized partial equilibrium model to analyze and understand the inner workings of mutually linked markets for fossil-based and innovative bio-based products. This study provides qualitative and quantitative insights that can support policymakers on their journey through the complexities of the transformation toward a low-carbon bioeconomy. The effect of various potential policies on the market share in a competitive equilibrium is of particular interest as this indicator is closely related to the bioeconomy transition and the EU greenhouse gas emission reduction target. We calibrate the theoretical model using data for representative innovative biotechnology to produce 1,4-Butanediol to provide numerical estimates of the modeled policy effects. Then, we perform a Monte Carlo simulation to relax the model assumptions and capture the uncertainty over the estimated efficiency of policy instruments.

Finally, I briefly discuss the implications beyond the results of individual studies based on the combined results presented in the thesis, making recommendations for further research work and highlighting how we contributed to the academic literature.

Contents

Introduction

Context and problem statement

The Earth's climate system continues to change at an unprecedented rate. The concentration of CO_2 is rising rapidly, creating a climate state that has never been experienced in human evolutionary history (Kiehl, 2011). The rising concentration of CO_2 continues to have severe effects on agriculture, forests, and human health (Smith and Myers, 2018). Ongoing research underscores the urgent need for rapid progress in global and national adaptation and mitigation efforts (Jones et al., 2012). In response to this challenge, more than thirty years ago, the United Nations Framework Convention on Climate Change (UNFCCC) was opened for signature at the 1992 Earth Summit in Rio de Janeiro (Kuyper et al., 2018). The UNFCCC recognizes climate change as a grave threat and reflects the will to mitigate and adapt to it and generate adequate funds for that purpose. In November 2022, the twenty-seventh Conference of the Parties (COP27)—the UNFCCC's authoritative body—hosted decision-makers from 196 countries, including those from the European Union (EU), to reiterate the urgent need for accelerating actions toward a more sustainable trajectory (European Commission, 2022b). In the same month, the global population was estimated to have exceeded eight billion (UN, 2022).

The EU aims to become the first climate-neutral bloc of countries by 2050 and aspires to achieve a modern and resource-efficient economy (EC, 2022a). With that goal, on 11 December 2019, the European Commission (EC) launched the ambitious "European Green Deal" (EC, 2019). At least one trillion euros of private and public funds are to be invested over the next decade to achieve the key 2030 targets, including reducing greenhouse gas emissions by at least 55% compared to 1990[1] levels , doubling the 2019 share (19.7%) of renewable energy sources, and improving energy efficiency by at least 32.5% (compared to the projections of the expected energy use in 2030). However, a common classification system (or taxonomy) that could be used to identify whether an economic activity can be considered sustainable was not established/created.

[1]According to the UNFCCC, 1990 is the adopted baseline due to the availability of GHG inventories in many countries (Kuyper et al., 2018).

Therefore, on 22 June 2020, the EC established the EU Taxonomy Regulation to ensure the efficient movement of capital toward truly green investments (based on the degree of sustainability of an activity) and implement the European Green Deal (Taxonomy Regulation: Regulation (EU) 2020/852). Article 9 of the EU Taxonomy Regulation established six objectives to identify environmentally sustainable economic activities, namely (i) climate change mitigation, (ii) climate change adaptation, (iii) sustainable use and protection of water and marine resources, (iv) transition to a circular economy, (v) pollution prevention and control, and (vi) protection and restoration of biodiversity and ecosystems (which includes sustainable land use and management). By allowing for consistency of information, the adoption of this regulation is a milestone in defining legally sustainable activities (Gortsos, 2021). The problem is that there is currently no consensus on what should be assessed; there is also a lack of holistic approaches to quantify the relevant indicators for a given economic activity (Caldeira et al., 2022). The inconsistency in measuring sustainability performance leads to different interpretations and meanings as well as information inefficiencies that complicate and impair investment into facilitating green transactions (Caldeira et al., 2022).

A new role for agricultural economists

The link between a prosperous society, a competitive economy, and a healthy planet places sustainable food systems at the heart of the European Green Deal (EC, 2020). As illustrated by the Sustainable Development Goals of the United Nations, most of the grand challenges relate to food and farming (Fresco et al., 2021). In fact, agri-food systems play a key role in (i) safeguarding food and nutrition security for a growing population; (ii) resolving climate change, pollution, and biodiversity loss; (iii) achieving healthy diets; and (iv) reaching equality in wealth and welfare. But where do agricultural economists stand in this context? What is it that agricultural economists contribute to sustainable development, and how could a greater impact be achieved with regard to addressing the aforementioned grand challenges? As an answer to these questions, Fresco et al. (2021) suggested the need for more integrative approaches to support (green) transformations, which rely on (i) better collaboration with other disciplines through more multi- and transdisciplinary approaches; (ii) stakeholder engagement; and (iii) more systematic approaches to the analysis of the trade-offs of interventions.

Agricultural economists are thus evolving and expanding the scope of problems and the variety of economic approaches (Zilberman, 2019). One growing area of research in applied agricultural economics is that of the circular bioeconomy, which covers both bio-based and circular concepts (figure 1) and pertains to a complex set of industries that rely on new knowledge in biology to produce new foods, fuels, fibers, and fine chemicals (Zilberman, 2019).

Based on the principles of sustainability and circularity (EC, 2018), the bioecon-

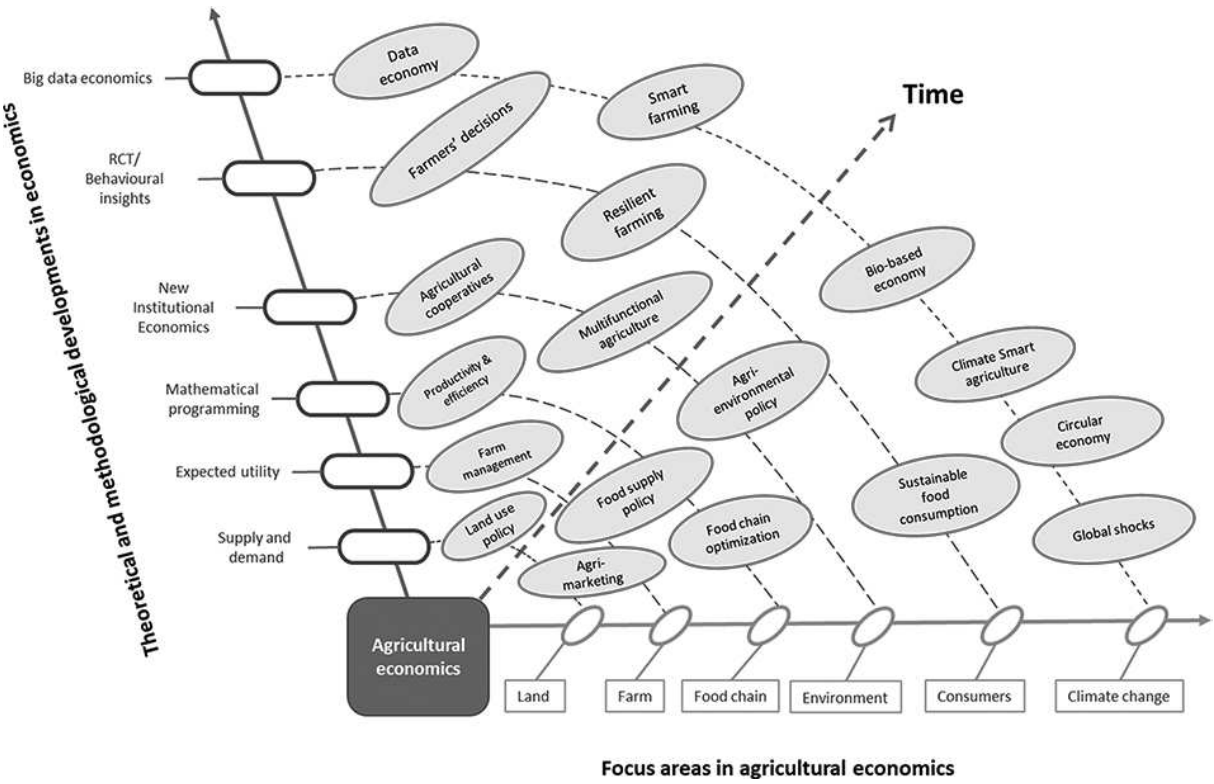


Figure 1: Conceptual framework

omy is arguably one of the main alternative economic systems to the fossil fuel-based economy. It is considered an essential driver for a sustainable and equitable future (EC, 2018; OECD, 2009). My thesis belongs to this new branch of applied economics. It aims to contribute, through innovative approaches and methods, to a more detailed and careful evaluation of alternative paths to sustainable innovation that apply technologies and social practices based on bio-based materials to transform or re-design conventional production systems. Further, it discusses concrete cases, with both scientific and practical implications, of sustainability performance assessments, providing significant insights into which aspects should be assessed, and how, and which ones should not be overlooked in order to incentivize bioeconomy development and economic actors to invest in truly sustainable activities.

Objectives, research questions, and methods

Sustainability performance assessments are not straightforward, especially when applied to bioeconomy research. A bioeconomy suffers from what could metaphorically be defined as a multiple identity disorder. Given the complexity and heterogeneity of bio-based systems, the literature contains several coexisting narratives on bioeconomy (Kardung et al., 2021). Bugge et al. (2016) and Vivien et al. (2019) propose three similar interpretations of the bioeconomy: (i) ecological economy, (ii) science-based economy, or (iii) biomass-based economy (the different interpretations will be discussed and studied in detail in Chapter 1). These narratives disperse bioeconomy research across many fields of science (Bugge et al., 2016) and lead to different, sometimes conflicting conceptions of the (scientific and policy) tools needed to support its development (Vivien et al., 2019). Thus, this thesis begins by systematically addressing the versatility of empirical studies—notably, case studies—along with the uncertainty surrounding the bioeconomy. The case study research method was chosen as it is *"the conventional way of doing process or implementation evaluations"* (Yin, 2014; p.222), especially for complex contemporary phenomena such as bioeconomy systems (Fitzgerald, 2017).

RQ.1: How do different visions of the bioeconomy influence case study approaches?

The versatility that characterizes the case study research method when studying contemporary phenomena appropriately matches the heterogeneity and complexity of modern bioeconomy systems. To answer the first research question, Chapter 1 describes a literature review based on a qualitative content analysis facilitated by systematic text coding. The ultimate goal is to formalize systematic approaches to positively influence the research agenda for bio-based systems-related case studies, the emphasis placed on elements of interest for society, and the direction of innovation and value creation in the bioeconomy research field.

Building on the knowledge gained from this literature review, the next three chapters discuss three empirical studies performed in heterogenous sectors of the bioeconomy. Chapter 2 covers the application of biopolymers derived from agro-food by-products in food packaging; Chapter 3 addresses the recycling of waste organic streams as biofertilizers for agricultural purposes; and Chapter 4 makes use, as instrumental case study, of the drop-in bio-based chemicals for an ex ante policy assessment. These case studies capture the heterogeneity and complexity of bio-based systems, which differ in terms of feedstock, level of technological maturity, and area of application.

Consumers and suppliers rely on fresh food packaged in single-use plastic containers to avoid food contamination and extend shelf life (Patrício Silva et al., 2021). Packaging, however, is also the most problematic type of plastic waste given the recalcitrant nature of fossil-based polymers and the single-use design (CIEL, 2019). Regardless of their harmful environmental effects, the COVID-19 pandemic has re-affirmed humanity's dependence on plastics (Patrício Silva et al., 2020) as well as led to changes in consumer habits, such as increased food deliveries and, consequently, plastic packaging (Filho et al., 2021). Under the new circumstances, it has thus become even more urgent to rethink and redesign plastics in a sustainable way (Patrício Silva et al., 2020). The development of bio-based polymers has attracted the attention of governments and researchers as a potentially sustainable alternative given their biodegradable and compostable nature (Kakadellis and

Harris, 2020). End-of-life alternatives, such as industrial composting or anaerobic digestion, can then reduce greenhouse gas emissions (Posen et al., 2017; Dilkes-Hoffman et al., 2018) and recover organic carbon (Vidal et al., 2007). The demand for bioplastics is growing, but their current market share is less than 1%. Investments in bioplastics applications are hampered by high capital and production costs (Lopez-Arenas et al., 2017; Shahzad et al., 2017). Chapter 2 describes the ex ante economic feasibility analysis conducted to address the second research question:

RQ.2: Do biodegradable food packaging films from agro-food waste pay off?

The case study focuses on the development of a biodegradable polymer from food waste, such as potato peels, assuming different industrial scales and logistical settings. The specific objective of this study is to assess the minimum selling price of the product for different industrial scales in Europe under conditions that would attract investors. Hence, a cost–benefit analysis was performed for the preliminary process design considering the main investment and operating costs over a selected time horizon. A major challenge was to incorporate uncertainty about scientific outcomes. Thus, a Monte Carlo simulation was applied to examine and analyze the uncertainties associated with the cost estimates and determine the relative impact of individual factors on the selling price. The method presented here not only provides new insights into the economic feasibility of investing in biodegradable and compostable polymers but also demonstrates the application of a techno-economic analysis that considers different scales of planting, logistics, and the uncertainty caused by a low technological readiness level (TRL).

The next chapter examines another bio-based product of contemporary relevance, bio-based fertilizers, whose production, unlike that of biopolymers, has a higher TRL. Disruptions in the trade of raw materials and primary products induce greater volatility for agri-food products (Behnassi and Haiba, 2022). In the fertilizer market, between the summers of 2020 and 2021, prices doubled (Smith, 2022) due to (i) market distortions related to the pandemic, (ii) rising agricultural commodity prices, and (iii) rising natural gas prices. The war in Ukraine has exacerbated this trend, causing prices to rise further, negatively impacting the EU's main fertilizer importer from Russia. Fertilizer price changes are also more volatile than those of other agricultural inputs due to demand rigidity, which is imposed by the fact that fertilizers are essential for production and have only a few substitutes (Beckman and Riche, 2015). Therefore, there are major concerns about the risk of fertilizer unavailability in the coming years. While practical substitutes exist for some inputs in agricultural production (e.g., capital for labor), chemical fertilizers tend to have few alternatives. One potential solution is the application of organic fertilizers, such as treated sewage sludge (Kumar Bhatt et al., 2019). A common end use of sewage sludge is landfilling, especially in some (new) EU member states (e.g., Malta, Croatia, and Romania) (Hudcová et al., 2019; Kelessidis and Stasinakis, 2012). However, if properly treated and processed, sewage sludge can be recycled and serve as a resource for organic fertilizer production. The third research question addresses this discussion.

RQ.3: Recycling or landfilling? What are the differences in their sustainability footprint performances?

This thesis also contributes to the strand of literature on multi-regional input–output analysis for the assessment of the spill-over effects of the economic, social, and environmental impacts generated from a change in final demand. The specific case explores sludge management and recycling for agricultural purposes, as it has become an crucial task for scientists (Kumar et al., 2017) given the global scenario and disruptions in the natural gas and fertilizer markets. The chapter contributes to the production economics literature by studying the nature of input use in a waste nutrient recovery strategy using an input–output and structural path analysis. It adds to the bioeconomy supply chain literature by comparing the recycling of treated sludge for agriculture purposes with landfilling. Finally, the input–output methodological framework is improved by combining it with a Monte Carlo simulation to address the uncertainty of model parameters. Researchers, policymakers, and practitioners can utilize this research to improve waste management in support of a more circular and sustainable strategy to ensure greater farm resilience to supply chain disruptions and plan for better supplier diversification within an upstream industry.

From the facts presented in previous chapters, it appears that the speed of the transition from fossil fuels to sustainable (bio-based) renewables depends on cost competitiveness and the readiness of markets for a green transformation (Asada and Stern, 2018). Once commercialized, a product must gain market share for the company to expand its production. Unlike the previously reported cases (referred to as dedicated bio-based products and having precisely dedicated development paths), there is a category of sustainable products that are identical to conventional products (so-called drop-in solutions; for example, 1,4-butanediol). The advantage of a drop-in solution over dedicated alternatives is that of being a homogeneous product of fossil-based equivalents, such that only price and environmental footprint are relevant to their adoption/diffusion (De Jong et al., 2012). Hence, these are more likely to be adopted more rapidly than dedicated products that need the development of new markets (De Jong et al., 2012).

A key chemical used in the production of plastics, elastic fibers, polyesters, polyurethanes, and pharmaceuticals is 1,4-butanediol (1,4-BDO) (Taylor et al., 2015). We have the technology to produce this same molecule through direct bioconversion from plant sugars (Satam et al., 2019). Bio-based production of 1,4-BDO has been shown to be interchangeable with the conventional molecule, with no change in product performance or post-production procedures (DSM Engineering Plastics, 2013); but it is still not competitive in terms of price. Given the reiterated urgency of funding climate change mitigation and adaptation activities, Chapter 4 reports an ex ante evaluation of different policy scenarios that support bio-based drop-in solutions, which serves to answer the last research question:

RQ.4: How well do policies—investment cost subsidies and operating cost subsidies—perform as incentives for stimulating a higher market share for renewable bio-based input alternatives?

This chapter aims to provide qualitative and quantitative insights that can support policymakers on their journey through the complexities of the transformation toward a low-carbon bioeconomy. Following this objective, and in the absence of historical data for econometric analysis, this study describes the development of a stylized partial equilibrium model to analyze and understand the inner workings of mutually linked markets for fossil-based and emerging renewable bio-based productions, which can be considered potentially perfect substitutes for the former. The effect of political tools on the competitive market share in equilibrium is of particular interest, as this indicator is closely related to the bioeconomy transition and the EU greenhouse-gas-emission-reduction target. In this chapter, the model is illustrated and calibrated using the instrumental empirical case of 1,4-BDO to provide a numerical analysis of the policy tool impacts. Finally, a Monte Carlo simulation is performed to relax assumptions regarding parameter calibration and capture the uncertainty on welfare effects.

After the main body of the thesis (Chapters 1 through 4)—to which these research questions and methods belong—the text discusses the implications beyond the findings of the individual studies according to the combined results presented in the thesis.

References

Asada, R., & Stern, T. (2018). Competitive Bioeconomy? Comparing Bio-based and Non-bio-based Primary Sectors of the World. Ecological Economics, 149(March), 120–128. https://doi.org/10.1016/j.ecolecon.2018.03.014

Beckman, J., & Riche, S. (2015). Changes To the Natural Gas, Corn, and Fertilizer Price Relationships From the Biofuels Era. Journal of Agricultural and Applied Economics, 47(4), 494–509. https://doi.org/10.1017/aae.2015.22

Behnassi, M., Haiba, M. El. (2022). Implications of the Russia – Ukraine war for global food security. Nature Human Behaviour. https://doi.org/10.1038/s41562-022-01391-x

Bugge, M. M., Hansen, T., Klitkou, A. (2016). What is the bioeconomy? A review of the literature. Sustainability (Switzerland), 8(7). https://doi.org/10.3390/su8070691

Caldeira, Farcal, R., Moretti, C., Mancini, L., Rauscher, H., Rasmussen, K., Riego, J., Sala, S. (2022). Safe and Sustainable chemicals by design chemicals and materials (Issue March). https://doi.org/10.2760/68587

CIEL. (2019). Plastic Health: The Hidden Costs of a Plastic Planet; Center for International Environmental Law Technical Report. https://www.ciel.org/wp-content/uploads/2019/05/Plastic-and-Climate-FINAL-2019.pdf

Daniel Posen, I., Jaramillo, P., Landis, A. E., Michael Griffin, W. (2017). Greenhouse gas mitigation for U.S. plastics production: Energy first, feedstocks later. Environmental Research Letters, 12(3). https://doi.org/10.1088/1748-9326/aa60a7

De Jong, E., Higson, A., Walsh, P., Wellisch, M. (2012). Product developments in the bio-based chemicals arena. Biofuels, Bioproducts and Biorefining, 6(6), 606–624. https://doi.org/10.1002/bbb.1360

Dilkes-Hoffman, L. S., Lane, J. L., Grant, T., Pratt, S., Lant, P. A., Laycock, B. (2018). Environmental impact of biodegradable food packaging when considering food waste. Journal of Cleaner Production, 180, 325–334. https://doi.org/10.1016/j.jclepro.2018.01.169

DSM Engineering Plastics. (2013). Achieving higher bio-based content in DSM Arnitel® copolyesters: Results from testing 1,4-butanediol (BDO) made with Genomatica's process technology.

EC. (2018). A sustainable Bioeconomy for Europe: strengthening the connection between economy, society and the environment.

EC. (2019). The European Green Deal. 53(9), 24

EC. (2022a). A European Green Deal.

EC. (2022b). EU at COP27 Climate Change Conference.

Filho, W. L., Voronova, V., Kloga, M., Paço, A., Minhas, A., Salvia, A. L., Ferreira, C. D., Sivapalan, S. (2021). COVID-19 and waste production in households: A trend analysis. Science of the Total Environment, 777. https://doi.org/10.1016/j.scitotenv.2021.145997

Fitzgerald, N. D. (2017). Chemistry challenges to enable a sustainable bioeconomy. Nature Reviews Chemistry, 1(10). https://doi.org/10.1038/s41570-017-0080

Fresco, L. O., Geerling-Eiff, F., Hoes, A. C., Van Wassenaer, L., Poppe, K. J., Van Der Vorst, J. G. A. J. (2021). Sustainable food systems: Do agricultural economists have a role. European Review of Agricultural Economics, 48(4), 694–718. https://doi.org/10.1093/erae/jbab026

Gortsos, C. V. (2021). The Taxonomy Regulation: more important than just as an element of the capital markets union. In Sustainable Finance in Europe (pp. 351–395). Palgrave Macmillan, Cham.

Hudcová, H., Vymazal, J., Rozkošný, M. (2019). Present restrictions of sewage sludge application in agriculture within the European Union. Soil and Water Research, 14(2), 104–120. https://doi.org/10.17221/36/2018-SWR

Jones, H. P., Hole, D. G., Zavaleta, E. S. (2012). Harnessing nature to help people adapt to climate change. Nature Climate Change, 2(7), 504–509. https://doi.org/10.1038/nclimate1463

Kakadellis, S., Harris, Z. M. (2020). Don't scrap the waste: The need for broader system boundaries in bioplastic food packaging life-cycle assessment – A critical review. Journal of Cleaner Production, 274, 122831. https://doi.org/10.1016/j.jclepro.2020.122831

Kardung, M., Cingiz, K., Costenoble, O., Delahaye, R., Heijman, W., Lovrić, M., ... Zhu, B. X. (2021). Development of the circular bioeconomy: Drivers and indicators. Sustainability, 13(1), 413.

Kelessidis, A., Stasinakis, A. S. (2012). Comparative study of the methods used for treatment and final disposal of sewage sludge in European countries. Waste Management, 32(6), 1186–1195. https://doi.org/10.1016/j.wasman.2012.01.012

Kiehl, J. (2011). Lessons from earth's past. Science, 331(6014), 158–159. https://doi.org/10.1126/science.1199380

Kumar Bhatt, M., Labanya, R., Joshi, H. C. (2019). Influence of Long-term Chemical fertilizers and Organic Manures on Soil Fertility - A Review. Universal Journal of Agricultural Research, 7(5), 177–188. https://doi.org/10.13189/ujar.2019.070502

Kumar, V., Chopra, A. K., Kumar, A. (2017). A Review on Sewage Sludge (Biosolids) a Resource for Sustainable Agriculture. Archives of Agriculture and Environmental Science, 2(4), 340–347. https://doi.org/10.26832/24566632.2017.020417

Kuyper, J., Schroeder, H., Linnér, B. O. (2018). The Evolution of the UNFCCC. Annual Review of Environment and Resources, 43, 343–368.

Lopez-Arenas, T., González-Contreras, M., Anaya-Reza, O., Sales-Cruz, M. (2017). Analysis of the fermentation strategy and its impact on the economics of the production process of PHB (polyhydroxybutyrate). Computers and Chemical Engineering, 107, 140–150. https://doi.org/10.1016/j.compchemeng.2017.03.009

OECD. (2009). The Bioeconomy to 2030: Designing a policy agenda. OECD Publishing.

Patrício Silva, A. L., Prata, J. C., Walker, T. R., Campos, D., Duarte, A. C., Soares, A. M. V. M., Barcelò, D., Rocha-Santos, T. (2020). Rethinking and optimising plastic waste management under COVID-19 pandemic: Policy solutions based on redesign and reduction of single-use plastics and personal protective equipment. Science of the Total Environment, 742, 140565. https://doi.org/10.1016/j.scitotenv.2020.140565

Patrício Silva, A. L., Prata, J. C., Walker, T. R., Duarte, A. C., Ouyang, W., Barcelò, D., Rocha-Santos, T. (2021). Increased plastic pollution due to COVID-19 pandemic: Challenges and recommendations. Chemical Engineering Journal, 405(August 2020), 126683. https://doi.org/10.1016/j.cej.2020.126683

Satam, C. C., Daub, M., Realff, M. J. (2019). Techno-economic analysis of 1,4-butanediol production by a single-step bioconversion process. Biofuels, Bioproducts and Biorefining, 13(5), 1261–1273. https://doi.org/10.1002/bbb.2016

Shahzad, K., Narodoslawsky, M., Sagir, M., Ali, N., Ali, S., Rashid, M. I., Ismail, I. M. I., Koller, M. (2017). Techno-economic feasibility of waste biorefinery: Using slaughtering waste streams as starting material for biopolyester production. Waste Management, 67, 73–85. https://doi.org/10.1016/j.wasman.2017.05.047

Smith, A. (2022). The Story of Rising Fertilizer Prices. In ARE Update (Vol. 25, Issue 3). https://coststudies.ucdavis.edu/en/current.

Smith, M. R., Myers, S. S. (2018). Impact of anthropogenic CO2 emissions on global human nutrition. Nature Climate Change, 8(9), 834-839.

Taylor, R., Nattrass, L., Alberts, G., Robson, P., Chudziak, C., Bauen, A., van Ree, R. (2015). From the Sugar Platform to biofuels and biochemicals: Final report for the European Commission Directorate-General Energy. https://ec.europa.eu/energy/sites/ener/files/documents/EC Sugar Platform final report.pdf

The Bioeconomy Council. (2015). Communiqué of the global bioeconomy summit - making bioeconomy work for sustainable development. 9.

UN. (2022). World population to reach 8 billion on 15 November 2022. https://www.un.org/en/desa/world-population-reach-8-billion-15-november-2022: :text=The global population is projected,today on World Population Day

Vidal, R., Martínez, P., Mulet, E., González, R., López-Mesa, B., Fowler, P., Fang, J. M. (2007). Environmental assessment of biodegradable multilayer film derived from carbohydrate polymers. Journal of Polymers and the Environment, 15(3), 159–168. https://doi.org/10.1007/s10924-007-0056-5

Vivien, F. D., Nieddu, M., Befort, N., Debref, R., Giampietro, M. (2019). The Hijacking of the Bioeconomy. Ecological Economics, 159(January), 189–197. https://doi.org/10.1016/j.ecolecon.2019.01.027

Yin, R. K. (2014). Case study research: Design and methods (fifth edit). SAGE publications. https://doi.org/10.1016/0149-7189(86)90052-2

Zilberman, D. (2019). Agricultural economics as a poster child of applied economics: Big data big issues. American Journal of Agricultural Economics, 101(2), 353–364. https://doi.org/10.1093/ajae/aay101

Chapter 1

Case Studies Research in the Bioeconomy: A Systematic Literature Review

Abstract

Case study research plays a crucial role in studying the development of the bioeconomy. The versatility of the empirical method coupled with the uncertainty surrounding the bioeconomy concept requires a consistent and comparable application of the method to obtain valid and generalizable results. To stimulate such systematization, we first need to know the state of case studies in bioeconomy research. This article reviews the recent literature with a qualitative content analysis facilitated by systematic text coding. Our results provide an overview of how the narratives of the concept of bioeconomy affect the versatility of the case study research. Based on the low density of the illustrated semantic networks, we conclude that future empirical research on bio-based phenomena should be more transdisciplinary and rely more on cross-sectoral approaches. Further work is also required in developing common research protocols that support transparency and replicability of case studies in the bioeconomy.

Keywords: circular bio-based economy; protocol; research methodology; resource management; sustainability

This chapter is based on: Tassinari, G., Drabik, D., Boccaletti, S., & Soregaroli, C. (2021). Case studies research in the bioeconomy: A systematic literature review. Agricultural Economics - CZECH, 67(7), 286-303.

1.1 Introduction

Global challenges regarding health, climate change, food security, energy security, cities, and migration urgently need solutions (Lund Declaration, 2015). The bioeconomy is a main alternative to fossil materials (OECD, 2009) to address these challenges (Lund Declaration, 2009, 2015). Several national strategies and policies for bioeconomic development worldwide confirm its key role (ACIL Tasman, 2008; BioteCanada, 2009; Organization for Economic Co-operation and Development [OECD], 2009; Staffas et al., 2013; European Commission, 2019; Kardung et al., 2021). Increasingly transformative policy initiatives, such as the European Green Deal or the target of net-zero emissions in the European Union (EU) by 2050 (European Commission, 2019), require intense efforts and coordination to exploit the bioeconomy and available synergies in all sectors and policy areas.

Given the contemporary, context-dependent nature of related phenomena, case study research [1] plays a crucial role in developing the bioeconomy. When little is known about a complex social phenomenon, a starting point is the collection and analysis of empirical evidence to gain information and key insights into real-world cases, a well-established research method (Denscombe, 2010).

Case study research is characterized by great versatility (Cavaye, 1996; Eisenhardt, 1989; Yin, 2014). A case study contributes to the knowledge of the individual, group, organizational, social, political, and related contemporary phenomena. It is preferred over other research tools, such as experiments, when the researchers have little or no control over the events studied and the boundaries between a phenomenon and its context are unclear (Yin, 2014). In support of the bioeconomy, several projects and initiatives (e.g., Annevelink et al., 2016; BERST, 2015; Charles et al., 2016; Gomez San Juan et al., 2019; Tassinari et al., 2019) have relied on case studies to advise on action plans, programs, and policies, to expand the general understanding of its sustainability, and to assess the effectiveness and maturity of possible bridging biotechnologies. Given the uncertainty surrounding the concept of the bioeconomy, however, this versatility can also threaten the method's consistent, comparable application.

Despite the emphasis placed on it, the bioeconomy concept is still a matter of debate, and a commonly accepted definition is lacking. As Bugge et al. (2016) and Vivien et al. (2019) have underlined, the literature contains several co-existing narratives of the bioeconomy concept. These narratives disperse bioeconomy research across many fields of science (Bugge et al., 2016) and lead to different, sometimes conflicting conceptions of economic policies and instruments needed to support the future bioeconomy (Vivien et al., 2019). As demonstrated later more exhaustively, case research reflects this richness

[1] In this article, we use the terms "case study research" or "case research" to refer to the scientific research method. "Case study" refers instead to a study that applies the scientific research method.

and diversity. Many disciplines use it as a research approach to support various models of sustainability with empirical evidence. To meet different needs, therefore, bioeconomy narratives apply case research differently. This complicates the understanding of case study approaches, which can seem fragmented and inconsistent in the bioeconomy domain. The problem is sharpened by the lack of common guidelines and protocols on how to conduct case study research in this field.

In the context of bioeconomy, case study methodology suffers from a lack of systematization, that is, consistent, comparable application. Without systematization, it is difficult to contribute to the literature in a clear, operationally defined way (Yin, 2014). The development of case study protocols can fill this gap by producing a comprehensive set of rules and procedures that make case study methodology more rigorous, forcing researchers to consider all the issues relevant to their research projects (Pervan & Maimbo, 2005).

To stimulate case study protocols in a specific research domain, we first need to gain knowledge on the current state of case studies. The literature provides several examples of case study reviews that discuss the variety of approaches adopted in specific research fields. For instance, Cavaye (1996) and Dubé and Paré (2003) reviewed the state of case studies in the field of information systems, Runeson and Höst (2009) in the field of software engineering, and Barratt et al. (2011) in the field of operations management. To the best of our knowledge, however, we are the first to examine case studies approaches in the field of bioeconomics.

This paper assesses the state of case studies in the bioeconomy domain. More specifically, considering the uncertainty surrounding the bioeconomy concept (Bugge et al., 2016; Vivien et al., 2019), the main research question we address is the following: How do different visions of the bioeconomy influence case study approaches? In answering this question, our goal is to provide a review of the current state of case studies in the bioeconomy by focusing on the differences among three co-existing economic narratives. In this way, we aim to provide a basis for critical discussion to develop case study protocols for bioeconomy research. The increased effectiveness of case study research should positively influence the research agenda, the political debate, the emphasis placed on elements of interest for society, and the direction of innovation and value creation.

Due to the ambiguity surrounding both bioeconomy and case study concepts, we begin by describing how the bioeconomy and case study concepts are interpreted in the literature. Based on the main features of these concepts, we present the research method and the systematic coding process applied to the selected articles. Next, we present and discuss our results. The last section summarizes the main arguments of the paper and provides final recommendations.

1.2 Literature review—conceptual framework

The scientific community interprets the concept of the bioeconomy and applies case study research in many ways. The following section describes the bioeconomy narratives in the academic literature and the key elements characterizing case study approaches. It then describes the attributes investigated and coded during the review of the selected articles.

1.2.1 Bioeconomy narratives

Despite the emphasis placed on it, the bioeconomy notion is still debated with little consensus on what it implies (Bugge et al., 2016). Divergent visions and interpretations of the bioeconomy characterize the literature.

Exploring the generic characteristics and nature of the term "bioeconomy," Bugge et al. (2016) identified three co-existing ideal visions of it: the bio-ecological vision, the biotechnology vision, and the bio-resources vision. The bio-ecological vision highlights sustainability as a central theme, including tensions and critical voices that focus on economic growth. Value creation is supported by promoting biodiversity, ecosystem conservation, ecosystem services provision, and soil-degradation prevention, emphasizing circular self-sustained productions and organic bio-ecological practices. This vision calls for more attention to transdisciplinary sustainability issues, taking the global scale as a starting point and including the negative externalities of bio-resources and biotechnologies. The biotechnology vision, in contrast, is primarily concerned with economic growth and job creation. It treats sustainability as subordinate since it a priori assumes positive effects from adopting biotechnologies. In this vision, investments in research and innovation play a central role as drivers of value creation, stimulating the applicability and commercialization of scientific knowledge in various sectors through the close interaction of universities and industries. Like the biotechnology vision, the bio-resources vision highlights cross-sectoral research involving innovation and collaboration as an important source of value creation. This third vision refers to both economic growth and sustainability, which are driven by the capitalization of bio-resources into new products and the establishment of new value chains. Issues related to the use and availability of biological resources, the cascading use of biomass, and waste management are predominant in this vision. Based on these ideal visions, it is not surprising that Bugge et al. (2016) argued that natural sciences and engineering perspectives influence bioeconomy research the most.

In accordance with these interpretations of the bioeconomy, Vivien et al. (2019) described three ideal types of bioeconomy narratives: (i) an ecological economy, (ii) a science-based economy, and (iii) a biomass-based economy. The first interpretation echoes Georgescu-Roegen's (1975) definition that the bioeconomy is meant to be a development model that simultaneously ensures economic and ecological balance by incorporating environmental variables into economic resource management solutions. This interpretation stresses the importance of preserving a limited stock of accessible resources that are disparately and unequally allocated (Georgescu-Roegen, 1975). Strict ecological constraints bind this bioeconomy type, promoting a standard of the sufficiency as a strategy for long-term development. Not surprisingly, economic policies and instruments supporting such a bioeconomy redistribute wealth equitably via ecological planning and limits. In the second bioeconomy narrative, in contrast, public policy strongly fosters a biological industrial revolution based on the establishment of biotechnologies as general-purpose technologies (Patermann & Aguilar, 2018). This narrative is known as a "knowledge-based bio-economy", as first advocated by the OECD (1998), or as a technology-driven bioeconomy, and it is often subject to social resistance, such as the case of genetically modified organisms. Because it aims to equate biology and life with biotechnology, Vivien et al. (2019) placed this narrative on the weak side of the sustainability debate. At a very early stage, as in the Cell Factory Key Action of the 5th Framework Programme (1998-2002), the European Union (EU) applied this second interpretation of the bioeconomy in research policy by encouraging the pragmatic mobilization of any research or technological development (Patermann & Aguilar, 2018). Following empirical evidence gathered on ongoing developments, such as the assessment of indirect land-use changes caused by the promotion of agrofuels, the European Commission (EC) has claimed the central role that the sustainability debate must play in the bioeconomy. Therefore, the EU started to support the bioeconomy as a circular renewable carbon economy based on biorefining—the biomass-based bioeconomy. This new bioeconomy interpretation reflects the definition used by the EC (European Commission, 2018), in which the bioeconomy includes all sectors and systems involving the economically viable use of biological resources and waste streams. Biorefining concepts also belong to this type of bioeconomy, which contributes to fossil-resource substitution via raw biomass fractionation and new bio-based value-added products. This interpretation lies between the sustainability models proposed by the bioeconomy narratives described above and seems to dominate them.

1.2.2 Case study research

As with the bioeconomy concept, there is no single interpretation of the notion of a case study. The case study method refers to an in-depth investigation of a contemporary phenomenon in its real-world context (Benbasat et al., 1987; Eisenhardt, 1989; Merriam, 1998; Robson, 2002; Yin, 2014). The method is open to several interpretations and approaches: case study research can adopt a deductive or inductive approach, can investigate one or more cases, apply different sampling strategies, use multiple data sources, and select analysis techniques that best fit both the qualitative and quantitative evidence. Regardless of this versatility, some common practices are highly recommended when conducting a case study.

Define the research question and objectives: A common agreement across prominent case study methodologies (Eisenhardt, 1989; Merriam, 1998; Stake, 1995; Yin, 2014) is to start by defining a research question and objectives since they largely determine the nature of a case study (Yin, 2014). The objective can be descriptive, exploratory, or explanatory; that is, case study research can be used for both theory building (inductive) and theory testing (deductive). A research question should have a well-defined focus and a clear objective as the groundwork for data collection and analysis (Eisenhardt, 1989).

Select case and unit of analysis: Once research questions and objectives have been defined, it is important to specify the case and the unit of analysis. Following the contributions of Grünbaum (2007) and Yin (2014), we define a case as a "phenomenon" studied in its real context, where a phenomenon is understood as any fact or event liable to be directly or indirectly observed, such as the implementation of a business model or the consequences of a production system. The analysis of a phenomenon must handle complex social systems integrated with equally complex natural systems (Boons & Wagner, 2009; Starik & Rands, 1995). Such complexity forces researchers to be selective, looking for a system boundary that allows them to develop significant insights into the studied complexity (Flood, 1999; Stewart, 2001). This system is called the "unit of analysis," the heart of the case (Grünbaum, 2007), which can be investigated in more detail using sub-units of analysis. For example, the implementation of a business model (case) could be investigated by analyzing a company (unit of analysis), which in turn can be represented by its manager or employees (sub-units).

Sampling strategies: Case study research requires a precise definition of boundaries, chosen units, and sampling strategy, as well as justifications of those choices based on the type, nature, and purpose of the study (Etikan, 2016). These choices significantly influence the feasibility and validity of data collection and analysis. Both probability and non-probability sampling techniques can be applied in case research. In probability sampling, each unit is randomly selected (Battaglia, 2008), whereas in non-probability sampling (such as convenience sampling and purposive sampling), units must meet certain criteria that justify the rationale of the sampling. For instance, in convenience sampling, the units meet practical criteria, such as easy accessibility, geographical proximity, availability at a given time, or willingness to participate (Etikan, 2016). In purposive sampling, the selection of units is based on theoretical aims dictated by the nature of the research project (Riffe et al., 2014). Units can be chosen because they express the maximum possible variation, share similar traits, or simply because they are considered typical, unusual, or critical (Etikan, 2016).

Data gathering: One strength of case study research is the opportunity to use both qualitative (e.g., interviews, observations) and quantitative (e.g., questionnaires) data-collection methods. Any finding or conclusion in a case study is much more convincing and accurate if it is based on heterogeneous sources of information (Dubé & Paré, 2003; Eisenhardt, 1989). Therefore, triangulation—the use of multiple sources aimed at corroborating the same evidence (Yin, 2014)—is highly recommended (Eisenhardt, 1989; Miles & Huberman, 1994; Yin, 2014). Triangulation allows building a richer, more complete picture of a phenomenon (Cavaye, 1996). It can be implemented using sources of the same data type (e.g., qualitative, such as survey data compared with documents from the literature) or different types (e.g., questionnaires administered by an interviewer and field observations).

Data and context analysis: Case study research allows selecting the methods that best suit the research questions (Creswell et al., 2007; Greene & Hall, 2010), making it possible to handle both qualitative and quantitative evidence. Data analysis *"consists of examining, categorizing, tabulating, testing, or otherwise recombining both quantitative and qualitative evidence to address the initial propositions of a study"* (Yin, 2014, p. 109). In this sense, case studies allow great flexibility and individual variation (Cavaye, 1996). The description of adopted data-analysis strategies and techniques should demonstrate the objectivity of the process by which the data are developed into conclusions (Barratt et al., 2011) and allow an external observer to understand those conclusions better (Dubé & Paré, 2003).

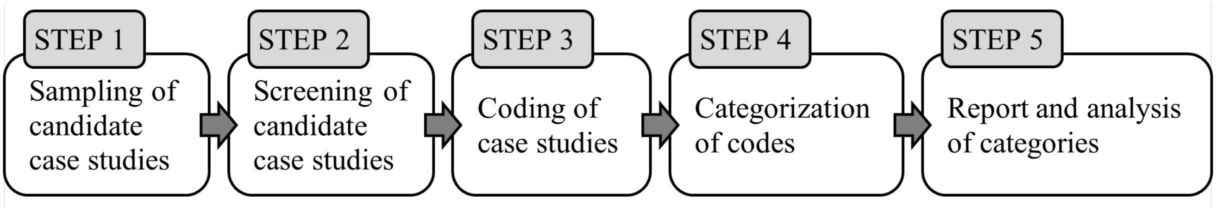
Understanding a real case involves important contextual conditions relevant to the case (Yin, 2014). Context plays a key role in the analysis. For the bioeconomy, several studies (Sheppard et al., 2011; Talavyria et al., 2015; Wesseler & von Braun, 2017) have identified the main forces driving the development of the bioeconomy and related phenomena. Kardung et al. (2021) summarized these forces by grouping them as supply drivers (technology and innovation, markets, and climate change adaption), demand drivers (consumer preferences, economic development, and demography), resource availability, and government measures. Researchers should not disregard detailed descriptions of context to ensure the robustness and generalizability of their findings.

1.3 Material and methods

We examine how various narratives of the bioeconomy affect case study approaches. Toward this end, we follow a five-step methodology (Figure 1.1). The following sections describe the details of each step.

The article-sampling strategy: Given the exploratory purpose of this study and the multidisciplinary nature of the bioeconomy, the sampling strategy focused on obtaining as many scientific journal sources as possible. For this reason, the Scopus database was selected. Due to its wide coverage of journals and articles, Scopus represents recent scientific literature well (Aghaei et al., 2013; Harzing & Alakangas, 2016), especially the social sciences (Aksnes & Sivertsen, 2019).

Figure 1.1: Methodological approach



The samples were delimited according to the following keywords and their variants: (i) "case study," "case study method," "field study," and "action research" to include all possible case studies; (ii) "bio*," "bio-*," "green," and "circular" to select possible bioeconomy case studies; and (iii) "bioeconomy," "economy," "supply chain," "value chain," "industry," and "sector" to focus on economic research. Considering the most recent complete year at the time of the sampling (2018) and extracting only English-language literature, the database provided 693 case studies in the field of bioeconomics. This significant number of articles provided a manageable, sufficiently exhaustive basis for an in-depth screening phase.

Screening phase: All the titles, abstracts, authors, journals, subject areas, citations, and keywords of the candidate case studies were tabulated. Two researchers independently read the abstract of each article to exclude articles that were not case studies and did not relate to the bioeconomy. The choices made independently by the researchers were consistent in 89% of cases. Any disagreement was discussed and eventually resolved. This process yielded 209 verified articles.

The articles from this sample were subjected to further screening to identify case studies in which the phenomena under investigation were the main objectives. In some cases, case study methodology is applied for instrumental purposes to facilitate understanding models, frameworks, or practical applications (Stake, 1995). For example, a case study whose purpose is simulating, calibrating, or demonstrating a model can be considered "instrumental" and usually has a narrow scope. This phase, which required reading the full texts of the articles, led to the exclusion of 117 articles and a final sample of 92 publications that were case study articles in the bioeconomy research field.

Coding phase: The 92 selected case studies were independently read several times by researchers. The following attributes were investigated: bioeconomy vision; research questions and objectives achieved; cases and related aspects under study; unit and subunits analyzed; economic activities involved; sampling strategy and data sources adopted; and data and context analysis conducted. Whenever possible, any evidence (text passages, phrases, and paragraphs) of the attribute studied was collected and reported in an Excel spreadsheet. This spreadsheet was used as a support tool for the coding, categorization, and analysis phases. From all the evidence gathered, codes were first extrapolated. Codes are constructs that provide an interpreted meaning for each datum for subsequent categorization and other analytical processes (Saldaña, 2013). Table 1.1 shows an example of this coding process.

Categorization phase: Based on Bugge et al. (2016) and Vivien et al. (2019) and on the sustainability model proposed by the selected case study, we defined the prevailing bioeconomy vision (or narratives). If the bioeconomy promoted in the paper was based on strict ecological constraints and environmental concerns took first priority, the case study was classified as having a bio-ecological vision (or "ecological economy" narrative). If the article emphasized a biotechnology-driven economy and promoted technological progress as a solution to all sustainability problems, it was categorized as having a biotechnology vision (or "science-based economy" narrative). Finally, if the article focused on the use of various types of biomass to replace fossil resources, promoting a biomass-based economy, it was categorized as having a bio-resources vision (or "biomass-based economy" narrative).

Once the prevailing bioeconomy visions were coded, the codes were grouped into categories to consolidate their meanings and descriptions. The basic categorization of the types of research questions concerned the series: "how," "why," "what," "where," and "whether." For the economic activities, the final wording corresponded to the categories in the International Standard Industrial Classification of All Economic Activities (ISIC) according to the procedure adopted by Food and Agriculture Organization (FAO) (2019). We used R software and WordNet for a computational strategy to group codes based on research objectives, cases and related aspects, and units of analysis. This approach revealed several groupings of synonymous words (synsets) and semantic relations that we verified and validated manually. A major advantage of this method is that it increases the objectivity and reliability of the qualitative analysis. Finally, the codes of the sampling strategies were categorized as either probabilistic or non-probabilistic sampling techniques, data sources and analysis as either qualitative or quantitative, and the context as either clearly or not clearly stated.

Reporting phase: The following section reports the results of the coding and categorization phases for each attribute according to the prevailing bioeconomy visions of the articles. As previously argued, the perception that research questions and objectives have the greatest influence on case study methodological versatility is shared by most involved scholars (Benbasat et al., 1987; Eisenhardt, 1989; Merriam, 1998; Robson, 2002; Yin, 2014). Given that bioeconomy visions differ in aims and objectives (Bugge et al. 2016), one can conclude that different bioeconomy narratives justify different case study approaches. Moreover, other criteria might highlight methodological differences across case studies. However, different interpretations of the bioeconomy also lead to different conceptions of sustainability models, economic policies, and instruments needed to support them (Vivien et al., 2019). Therefore, using bioeconomy visions as a criterion, we

Table 1.1: Example of coding process

Raw data for the attributes based on Mengistu et al. (2018)	Attributes	Codes	Categories
This challenge stimulates the need to move from an economy based on fossil fuels to a biomass-based economy (. . .). The definition of biomass-based economy, also known as 'bioeconomy', remains a matter of debate.	bioeconomy vision	biomass-based economy	bio-resources vision
For what purposes do farmers use maize biomass? How important are these decisions for household food security?	research questions	what, how	what, how
The maize sector (. . .) indicated some potential 'demand sinks' and new opportunities in the livestock and food processing sectors that would help to overcome market related constraints. We identified several challenges in realizing these opportunities.	sector	maize sector, livestock sector, food-processing sector	agricultural sectors; food; beverages; tobacco
The aim was to quantify maize biomass production and utilization, and thereby to examine the implications of biomass use on household food security... This paper examines the uses of maize biomass as a bioeconomy crop, and its implications and challenges for household food security.	research objective	quantify, examine	determine; examine
	related aspects	production, utilization, implications, challenges	status; implications; challenges
	case	maize biomass	Biomass
Data were collected from a household survey covering 325 randomly selected farmers, interviews with key informants, and focus group discussions in two maize-belt districts, Mecha and Bako, in Ethiopia.	unit of analysis	(two) districts	administrative district
Our key informants included maize growers, experts at the district agriculture office, researchers at the national maize research center, experts with the food and feed processing industries and poultry farm managers and owners.	sub-units	households, farmers, experts, researchers, managers, owners	stakeholders groups
	data sources	survey, interview, focus group discussion	qualitative; quantitative
A multi-stage random sampling technique was employed to draw sample households. Firstly, Mecha district in the Amhara region and Bako district in [the] Oromia region were selected purposively. (. . .) Secondly, three peasant associations fromBako district and four from Mecha were randomly selected. Finally, a total of 325 maize farmers, 188 from Mecha and 137 from Bako, were selected randomly.	sampling strategies	multi-stage random sampling technique, purposive, random	non-probability; probability sampling; techniques
Data were analyzed using content analysis, descriptive statistics and an endogenous switching regression model.	data analysis	content analysis, descriptive statistics, endogenous switching regression model	qualitative; quantitative
The first section of the results gives a description of the sociodemographic and socio-economic characteristics.	context analysis	descriptive	clearly stated

can highlight the environmental, economic, and social implications boosted by different case studies.

Based on the Scopus All Science Journal Classification (ASJC) and the sectors that emerged from the categorization process, the cross-disciplinarity and cross-sectoral approaches of the case studies were investigated. The results were reported using circle semantic network graphs. Similarly, the emerging research objectives, cases and related aspects, units of analysis, and their links were visualized with edge-bundling graphs. Finally, we tabulated the results in terms of research questions, sampling strategies, data sources, data analysis, and context analysis.

1.4 Results and discussion

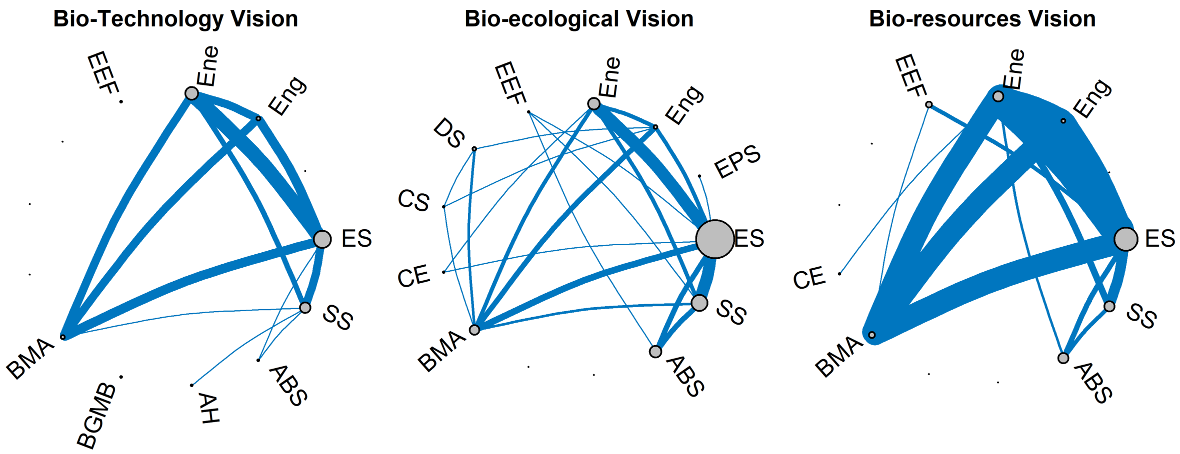
The selected articles covered 47 scientific journals and collected a total of 522 citations over two complete years (2018–2019), with an average of nearly three citations per year for each article. Nevertheless, more than 80% of total citations belonged to ten journals, which published around 60% of the selected case studies (see Supplementary Table S1 for the complete list). Among the case studies, the bio-resources vision of the bioeconomy was the most widespread (40 out of 92), followed by the bio-ecological vision (31 out of 92) and the biotechnology vision (21 out of 92).

Subject areas and cross-disciplinarity: Based on Scopus ASJC, the journals from the sample covered 13 subject areas. Figure 1.2 reports these (as nodes) and illustrates the cross-disciplinary[2] nature of the case studies according to the bioeconomy vision. The size of each node depends on the number of different journals in that discipline. Each link between nodes represents the number of case studies published in journals that covered the subject areas connected by the link. In this paper, the number of nodes in the networks is used as an indicator of the interdisciplinarity degree of the case studies according to the bioeconomic vision. On the other hand, the number of links is used as an indicator of the transdisciplinarity of the case studies.

Figure 2 is quite revealing in several ways. First, a comparison of the three networks reveals that case studies with a bio-ecological vision adopt more cross-interdisciplinary approaches than case studies with biotechnology and bio-resources visions. The bio-ecological vision case study network covered 11 subject areas and 22 interactions among them. In contrast, the biotechnology vision covered nine disciplines, two of which were not interlinked with any other discipline, and 12 links. The bio-resources vision network

[2]Following Aagaard-Hansen (2007), we use the term cross-disciplinarity as a general designation for all research forms involving different disciplinary backgrounds; interdisciplinarity refers instead to the engagement of different disciplines to address common issues but still with a discipline-specific approach; transdisciplinarity identifies research that entails more integration across disciplines than interdisciplinarity.

Figure 1.2: Cross-disciplinarity among bioeconomy case studies under different bioeconomy visions



Note: ABS – agricultural and biological sciences; AH – arts and humanities; BGMB – biochemistry, genetics, and molecular biology; BMA – business, management, and accounting; CE – chemical engineering; CS – computer science; DS – decision sciences; EPS – Earth and planetary sciences; EEF – economics, econometrics, and finance; Ene – energy; Eng – engineering; ES – environmental science; SS – social sciences
Source: Own elaboration based on the reviewed literature published in 2018

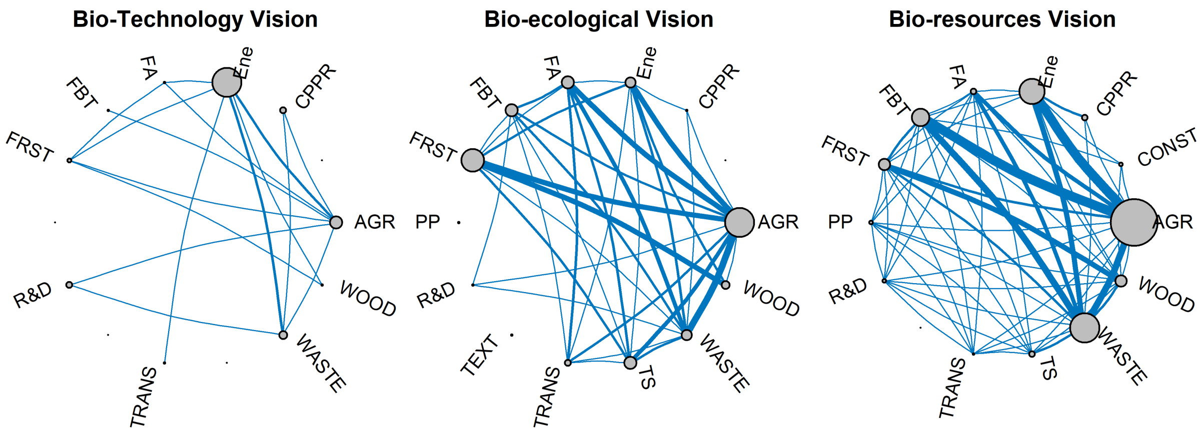
included 14 links among eight disciplines.

Second, the figure shows that the selected case studies were more frequently related to environmental sciences regardless of the bioeconomy interpretation. The journals in the environmental thematic area were 19 (of 25 journals) in the bio-ecological vision, 11 (of 18) in the bio-resources vision, and 8 (of 13) in the biotechnology vision. Based on the number of links that each node had with others (degree of centrality), environmental sciences occupied the most central position in transdisciplinary approaches. Similarly, the subject area "energy" was central in the bio-resources vision and "social sciences" in the biotechnology vision.

Finally, it is apparent from this figure that most potential transdisciplinary approaches were not concretized in the case studies. The density of a network is a measure of the ratio of the number of existing connections to the number of total potential connections. Considering 13 nodes, 78 potential connections between disciplines were possible in each vision. In the bio-ecological vision, the network of case studies had the highest density of 28%; the bio-resources and biotechnology visions had densities of only 18% and 15%, respectively.

Sectors and cross-sectoral approaches: The sample of case studies covered 14 economic activities. Figure 1.3 illustrates these sectors (as nodes) and the cross-sectoral approaches captured by the sample case study according to the bioeconomy vision. The size of the node and the width of each link between nodes depends on the number of case studies related to those sectors.

Figure 1.3: Cross-sectoral approaches among case studies under different bioeconomy visions



Note: AGR – agricultural sectors; CONST – bio-based construction material; CPPR – bio-based chemicals, pharmaceuticals, plastics and rubber (excluding biofuels); Ene – bioenergy; FA – fishing and aquaculture; FBT – food, beverages and tobacco; FRST – forestry; PP – pulp and paper; RD – research and development; TEXT – bio-based textiles; TRANS – transportation and storage; TS – recreation associated with ecotourism; WASTE – waste management; WOOD – wood products and furniture

Source: Own elaboration based on the reviewed literature published in 2018

A closer inspection of Figure 3 shows how the case studies focused on different economic activities according to each bioeconomy vision. Case studies with bio-ecological or bio-resources visions had similar profiles, involving 13 different branches of economic activities, while case studies with a biotechnology vision involved 10 sectors in total. The case studies with biotechnology visions focused on bioenergy-related economic activities (14 out of 21). The case studies with bio-ecological visions instead involved primarily agricultural sectors (14 out of 31) and forestry (11 out of 31). Finally, in the bio-resources vision group, the case studies referred most frequently to agriculture (24 out of 40), waste management (15 out of 40), and energy (13 out of 40). Regardless of the bioeconomy vision, agricultural sectors represented the most central node according to the degree of centrality of the network.

Regarding the density of the networks, the selected case studies showed more concrete cross-sectoral approaches than the cross-disciplinarity. Considering 14 nodes and a total of 91 potential connections, the case studies with bio-resources visions covered 63% of the potential interactions between sectors. Lower densities characterized the case study networks of the bio-ecological (38%) and biotechnology visions (17%).

Research questions: The selected bioeconomy case studies focused on the "what"

Table 1.2: Type of research questions

Research Questions	Visions			Total (n = 92)
	biotechnology (n = 21)	bio-ecological (n = 31)	bio-resources (n = 40)	
Not clearly stated	4	9	10	23
Clearly stated				
What	9	16	19	44
How	10	11	13	34
Why	0	0	2	2
Where	0	0	1	1
Whether	1	2	1	4

Note: A case study could have more than one research question, so the sum of the columns can exceed the number of case studies (n)
Source: Own elaboration.

and "how" questions (Table **??**). The "what" questions (e.g., "What are the main challenges related to the emergence of novel bio-based value chains?" [Carraresi et al., 2018]) were predominant in the bio-ecological and bio-resources visions. This type of question generally defines an exploratory study (Yin, 2014), with the aim of developing relevant hypotheses and propositions for further investigation. In contrast, the "how" question (e.g., "How do the economic costs of acquiring a biotechnology compare to the costs saved and additional benefits accrued?" [Kabyanga et al., 2018]) was the most common in the biotechnology vision. This question, as well as "why" (e.g., "why is the use of biomass for energy different among countries" [Bentsen et al., 2018]), is usually more explanatory (Yin, 2014), providing grounds for modifications of a theoretical framework (Grünbaum, 2007).

Research objectives, cases, and units of analysis: For all the selected case studies, it was possible to codify the research objectives, the phenomena studied and related aspects, and the units analyzed (Figure 1.4). For illustrative purposes, the figure shows only nodes (categories) and edges (links between categories) shared by at least one pair of case studies within the same bioeconomy vision group. Each idiosyncratic form, specific to a case study and not shared by articles, is grouped into a single node called "others." The size of each node and link is based on the number of case studies.

In terms of research objectives, less than half of the selected case studies (42 out of 92) had multiple research objectives. Regardless of the bioeconomy vision, case study research was adopted mainly to observe or inspect carefully or critically ("examine", 33 out of 92), estimate values ("evaluate", 26 out of 92), and ascertain facts or information ("investigate", 15 out of 92). For instance, case studies were used to examine the implications of biomass use (Mengistu et al., 2018), to evaluate the opportunities and barriers of bio-

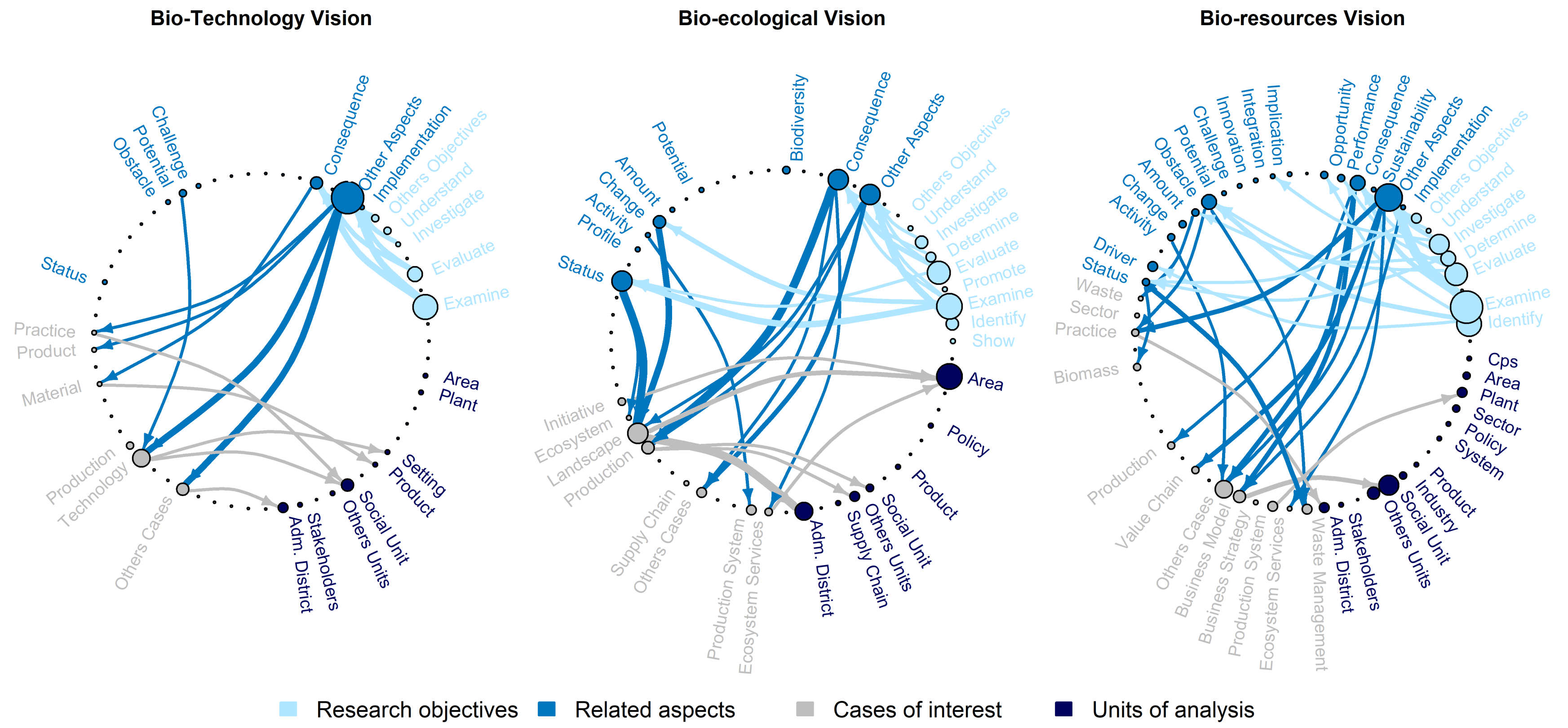


Figure 1.4: Research objectives, cases, and units of analysis

Note: Adm. – administrative; Cps – consumption production system

Source: Own elaboration based on the reviewed literature published in 2018 based production (Singlitico et al., 2018), or to investigate the integration of innovative technologies into bio-based production (Skvortsova et al., 2018). The bio-ecological and bio-resource visions also frequently used case studies to establish identities ("identify", 15 out of 92) and set limits ("determine", 10 out of 92). For instance, case studies were used to identify the contributions to the global environmental impact of bio-based production (Newton & Little, 2018) and to determine the direct and indirect value of economic losses to ecosystem services (Toledo et al. 2018).

Compared to the research objectives, the studied phenomena and related aspects showed more heterogeneity across bioeconomy visions. In the biotechnology vision, the case studies focused on the case-specific aspects of mature biotechnologies (7 out of 21), such as alternative bioenergy digesters. Case-specific aspects were generally related to the opportunity to increase production and profitability. In the bio-ecological vision, case studies focused on the sustainability and circularity of agricultural and forest-land use (8 out of 31) and on the environmental impact of bio-based productions (5 out of 31), generally providing reference standards for ecological compensations. Finally, in the bio-resources vision, the case studies exhibited a predominantly process-oriented nature, focusing on several aspects related to business models and strategies (7 out of 40), production and production systems (7 out of 40), waste-management systems (6 out of 40), business practices (3 out of 40), and value chains (3 out of 40). As opposed to the biotechnology vision, these case studies focused on new and emerging phenomena to gather empirical evidence for potential future scenarios. To this end, the case studies also emphasize reducing uncertainty about the properties and availability of biological raw materials, such as biomass and waste.

Regarding units of analysis, half the total sample relied on three types of units: administrative districts (16 out of 92), geographical areas (15 out of 92), and social units (16 out of 92), including companies and households. Administrative districts and geographical areas were mainly approached by the bio-ecological vision, while social units were more frequently analyzed by the other two bioeconomy visions. The units were then split into sub-units involving different categories of stakeholders and individuals, including experts, technicians, policymakers, business members, managers, employees, unemployed, producers, farmers, out-takers, out-growers, end-users, residents, retirees, and students.

Sampling strategies: Forty-seven percent of case studies did not describe their adopted sampling strategies (Table 1.6). The rest described different non-probabilistic sampling (45 out of 49) and probability sampling (9 out of 49) strategies or a combination of the two (5 out of 49).

For all the bioeconomy visions, the case studies mainly used purposive sampling strategies. In the bio-ecological vision, this strategy was primarily based on the maximum-variation sampling technique (5 out of 12) to collect samples with the most heterogeneous characteristics possible, such as a sample of different geographical areas chosen for their

Table 1.3: Sampling strategies

Sampling Strategy	Visions			Total (n = 92)
	biotechnology (n = 21)	bio-ecological (n = 31)	bio-resources (n = 40)	
No logic offered	9	16	18	43
Non-probability				
Purposive	11	12	21	44
Convenience	1	1	2	4
Probability	2	3	4	9

Note: A case study could have more than one sampling strategy, so the sum of the columns can exceed the number of case studies (n)
Source: Own elaboration.

different soil, climate, socio-economic, and legislative conditions. In the bio-resources vision, the most common purposive sampling techniques included critical and extreme case sampling (10 out of 21), such as the choice of industries with large quantities of waste streams or a company selected for its economic results and market position. In the biotechnology vision, case studies mainly relied on critical and maximum-variation sampling criteria (8 out of 11 purposively chosen units).

The probabilistic random sampling strategy involved four case studies, three of which shared a bio-ecological vision of the bioeconomy and adopted probabilistic sampling strategies to examine the status and use of ecosystems. The fourth, with a biotechnology vision, used a random sample to review the efficiency and implementation of a public-private program related to the bioenergy sector. If both non-probabilistic and random sampling techniques were adopted, the selection generally involved multiple phases, such as applying a stratified sampling or multi-stage random sampling technique.

Data collection: All the selected case studies described their data-collection methods. Most classified as having bio-resources or biotechnology visions relied on qualitative data sources, while in the bio-ecological vision, case studies relied more frequently on quantitative data sources.

Data triangulation was a common practice as well (Table 1.4). The case studies with bio-resources visions triangulated various data sources less than the other groups, however. Case studies that developed convergent evidence mainly combined qualitative and quantitative lines of evidence (37 out of 70) or triangulated different qualitative evidence (28 out of 70).

Data and context analysis: The trends observed for data collection reflected those of the data analysis. Most case studies with biotechnology or bio-resources visions adopted qualitative analyses, while quantitative analyses were the most common in the

Table 1.4: Number of bioeconomy case study articles (n) by data sources and triangulation

Data sources	Visions			Total
	biotechnology (n = 21)	bio-ecological (n = 31)	bio-resources (n = 40)	(n = 92)
Qualitative data sources				
Documents (literature)	14	14	23	51
Field visits	3	2	3	8
Focus groups	2	5	5	12
Interviews	15	15	24	54
Observations	4	5	4	13
Web and social networks	0	1	1	2
Workshops	2	0	2	4
Quantitative data sources				
Censuses	0	3	1	4
Databases	3	6	4	13
Maps	1	5	0	6
Questionnaires	4	11	10	25
Records	3	4	5	12
Reports	4	4	1	9
Data triangulation:				
Yes	17	26	27	70
No	4	5	13	22

bio-ecological vision group (Table 1.5).

When adopting a qualitative analysis, case studies frequently indicated a generic qualitative analysis (19 out of 64); when a specific qualitative analysis was mentioned, content and thematic analysis were the most common. Among the quantitative analyses, lifecycle analysis was the most widely used (14 out of 48). Over 20% of the case studies combined qualitative and quantitative analysis.

Turning to context analysis, of the 92 case studies, 23 (25%) did not provide any information on the real-world contextual conditions pertinent to their cases. This trend was similar in each bioeconomy vision. The case studies that conducted context analyses described the geographical, economic, social, and legislative contexts of the analyzed units.

1.4.1 The effect of bioeconomy narratives on case study approaches

The case study methodology can provide important empirical evidence for a better understanding of the bioeconomy and its components. The exploratory results of this work demonstrate the versatility of case research in the bioeconomy domain. This flexibility varies according to different narratives of the bioeconomy concept. Table 6 summarizes the key categories that emerged for each attribute investigated.

Table 1.5: Data and context analysis

Data analysis	Visions			Total
	biotechnology (n = 21)	bio-ecological (n = 31)	bio-resources (n = 40)	(n = 92)
Data analysis				
Qualitative	16	15	33	64
Quantitative	10	20	18	48
Context analysis				
Clearly stated	16	23	30	69
Not clearly stated	5	8	10	23

Source: Own elaboration.

The selected case studies highlight common patterns across the bioeconomy narratives. Among the scientific disciplines, the environmental science perspective was confirmed as the most central. Relationships with the environment and natural resources are integral, essential parts of the reality of any bio-economic system. Similarly, among the branches of economic activity, primary production and agri-food systems play prominent roles in most bioeconomy strategies given their dependence on biological resources. Other common aspects concern research objectives and sampling choices. Regardless of a study's vision, the bioeconomy is an emerging field and is thus unexplored in many respects that must be examined carefully and critically. For the same reason, the purposive sampling strategy is the most widely adopted, as it relies on intentionally chosen units of analysis to clarify doubts inherent in the emerging bioeconomy. Finally, the many case studies describing context-specific features reinforce the high context-dependence of bioeconomic success. In addition to these common traits, the case studies exhibited several specificities related to their own visions of the bioeconomy.

The contemporary literature on the bioeconomy draws from case study research to gather new evidence of the spreading use of biotechnologies. Operational biotechnologies are examined and evaluated in detail primarily for explanatory purposes to answer "how" questions. To this end, case studies are specific and narrowly focused and, therefore, less cross-disciplinary and multisectoral than in other research fields. Empirical evidence is collected primarily qualitatively from companies that are key players in the biotechnology revolution. Critical sampling or maximum-variation techniques facilitate the analytical generalization of such evidence. The results obtained by these case studies could bolster the development of policies to support bioeconomic expansion, such as funds and subsidies to cover the initial capital costs of technological modernization (Fuldauer et al., 2018).

In the ecology-focused narrative, the case studies confirm the criticisms of the bio-ecological vision reported by Bugge et al. (2016). Through cross-disciplinary global-sustainability judgments, case studies primarily question environmental impact (Corcelli

Table 1.6: Summary comparison of bioeconomy case studies by methodological attributes

	Case studies		
	biotechnology	bio-ecological	bio-resources
Disciplines:	2+7 (12 links)	11 (22 links)	8 (14 links)
Highest degree of centrality	Environmental sciences; social sciences	Environmental sciences	Environmental sciences; energy
Sectors:	10 (15 links)	13 (34 links)	13 (57 links)
Highest degree of centrality	Agriculture	Agriculture	Agriculture
Research questions	How (explanatory)	What (exploratory)	What (exploratory)
Research objectives	Examine; evaluate	Examine; evaluate	Examine; identify; evaluate; investigate
Phenomena (case)	Technology-oriented	Environment-oriented	Process-oriented
Unit of analysis	Social unit	Administrative district; geographic area	Social unit
Sampling strategy	Purposive (critical or maximum-variation sampling techniques)	Purposive (maximum sampling techniques)	Purposive (critical or extreme sampling techniques)
Data sources	Qualitative	Quantitative	Qualitative
Data analysis	Qualitative	Quantitative	Qualitative
Clearly stated context	76%	74%	75%

Source: Own elaboration.

et al., 2018) and local land-use planning (Angelstam et al., 2018; Naumov et al., 2018) stemming from bio-economic developments. In contrast to the other narratives, the ecological economy narrative steers case studies toward mainly quantitative exploratory assessments based on cases expressing the largest variability. Case studies thus succeed, for example, in promoting sustainable economic development and integrated landscape planning (Naumov et al., 2018) and in providing important reference standards for ecological compensation, which is useful to regional environmental policymakers (Wang et al., 2017).

Finally, consistent with Vivien et al. (2019), the bioeconomy narrative explaining fossil resource replacement through bio-resource capitalization appears to be the most common. Case studies in this area play important roles in anticipating future scenarios. By answering exploratory research questions, case studies shed light on the potential properties, opportunities, and obstacles of bio-based raw materials and innovative processes, focusing on circularity and recycling. For these aspects, case studies should not disregard multisectoral approaches based mainly on the qualitative analysis of critical or extreme cases. Evidence gathered in this way emphasizes, for example, knowledge creation, entrepreneurial experimentation, and market formation (Binz et al., 2014; Dautzenberg & Hanf, 2008) and aids in the design of appropriate policies to support innovative systems and sustainable transitions (Purkus et al., 2018).

1.4.2 Practical implications and recommendation for future case studies

The existing literature is important when formulating new case studies. Researchers often draw from the most recent published research to choose key methodological elements for their case study analyses. However, as our findings imply, adapting the methodology to the research questions is a crucial phase, entailing that a full comprehension of the bioeconomy case study literature is necessary. To this end, we provide a key to understanding and properly coding case studies based on their methodological and context features, therefore enabling a more systematic detection of their bioeconomy visions.

Improved systematization across the described key attributes would facilitate corrective actions toward a more common logic of case studies in bioeconomy research. In essence, this article aims to encourage the development of case study protocols by highlighting all relevant issues that researchers should consider for their case research in the bioeconomy. A reasonable approach to tackle this need could be developing different research protocols specific to the bioeconomy vision. By developing case study protocols, research and innovation efforts can be directed to "the systematic approaches needed to achieve the aims of the Green Deal" (European Commission, 2019, p. 18). The Commission, for example, may foster the use of case study protocols in research projects for bioeconomic development. Such protocols must incentivize two aspects that are lacking in the case studies reviewed in this article.

The first aspect concerns greater methodological transparency. Several methodological gaps frequently occur without significant differences across bioeconomy narratives. First, authors should pay more attention to their descriptions of their research questions. One-quarter of the selected case studies underestimated the role that clearly stating the research questions plays in full comprehension of the focus of the study (Dubé & Paré, 2003). The next step would be to apply sufficiently generalizable interpretations of cases and units of analysis. Less idiosyncratic forms of these elements are recommended to facilitate better integrations and comparisons of case studies. Furthermore, our findings recommend more attention to the motivation and logic of sampling strategies to improve the validity, generalizability, and comparability of case studies. For the same purpose, context analysis should be more systematic in the elements discussed (e.g., those suggested in Section 2.2). Finally, given the frequent use of generic qualitative analysis, the level of rigor that characterizes quantitative analytical procedures should also be extended to qualitative approaches.

The second aspect concerns the need for more cross-disciplinary, multisectoral efforts in bioeconomy case studies. Cross-disciplinary studies in which different research areas work jointly on a specific problem have great potential for creativity and innovation (Borge & Bröring, 2017), which is also supported by the identification of cross-sectoral collaborations for value creation (Bauer et al., 2018). In general, any narrative can better leverage case research as a cross-disciplinary, cross-sector platform: strengthening networks of companies and research institutions; fostering the development of localized bio-based technology clusters (Golembiewski et al., 2015); promoting the development of sustainable production systems (Binz et al., 2014; Markard et al., 2012); exploring the potential for further convergence between agricultural activities and less-explored sectors (Carraresi et al., 2018), such as chemicals; and helping to tackle the challenges associated with biotechnology transfer, particularly between academia and industries (Borge & Bröring, 2017).

These two aspects—better methodological transparency and greater cross-disciplinary, multisectoral efforts—would facilitate the integration of bio-economic narratives. Instead of competing with each other (Vivien et al., 2019), such as promoting conflicting governmental policies (e.g., the intensification of biomass production and biodiversity conservation) (Naumov et al., 2018), the different bioeconomy visions can complement each other through integrated planning of applied empirical methods. Such synergies are supported and facilitated by the methodological standard that we propose.

In the absence of clear case research protocols for the bioeconomy, the present study provides an initial set of recommendations for future analysis, summarized in the following list: (1) Begin by clearly defining the research questions and objectives, bearing in mind that case research in the bioeconomy domain is primarily used for exploratory with "What" and "How" questions, (2) Define the bioeconomic narrative to refer to, without assuming a common interpretation of the bioeconomy concept, thus providing a means for a better understanding of the case study, (3) Select cases and units of analysis carefully, leveraging cross-disciplinary and cross-sectoral designs, knowing that environmental science and agricultural sectors will likely play prominent roles, (4) Apply sufficiently generalizable interpretations of cases and units of analysis, avoiding idiosyncratic forms using a proper coding system from the previous literature, (5) Adopt a suitable sampling strategy based on the type, nature, and purpose of the study and justify it to improve the validity, generalizability, and comparability of the case study, (6) Collect data emphasizing data triangulation, that is, use multiple sources to corroborate the same evidence, (7) Demonstrate the objectivity of the process by which the data are developed into conclusions with a proper level of rigor for both qualitative and quantitative analytical procedures, (8) Describe context-specific features extensively, recognizing the high context dependence of any bioeconomy-related phenomena.

1.5 Conclusions

The primary purpose of this study was to provide a review of the current state of case studies in the bioeconomy by focusing on the differences among three co-existing economic narratives. Our findings have practical implications that should fuel the debate on the systematization of case study analysis in bioeconomy research. Often, dissimilarities among case studies with different bioeconomy visions are speculative. By looking at the literature, this paper provides evidence of common traits and differences in case study approaches across different bioeconomic narratives.

Overall, there is a need for developing common research protocols that support transparency and replicability of case studies in the bioeconomy domain. Such protocols can compensate for the methodological gaps that occur in the bioeconomy literature by incentivizing common research standards. In the same way, greater attention should be placed on transdisciplinary and multi-sectoral efforts as a way to accelerate progress toward the bioeconomy.

The results of this study are subject to some limitations. The main limitation concerns the selected sample of case studies. The choice of a single year was necessary to reduce the high number of case studies without limiting variety. A second limitation relates to a potential subjectivity bias. To mitigate this problem, we selected the articles independently and coded them using the systematic coding procedures described in the methodology section.

Further work regarding the current state of case study in the bioeconomy would be worthwhile. We believe that future studies could take advantage of the list of journals that emerged from our analysis to select a multi-year sample of articles and conduct quantitative analysis of the state of case studies in the bioeconomy. A multi-year sample would allow an examination of the level of interaction and integration across case studies. This would be a significant area of improvement in providing a conceptual framework of the sustainability and circularity of the phenomena studied. In addition, for a full discussion of the role of case research in the bioeconomy, a better understanding of instrumental case studies must be developed.

References

Aagaard-Hansen J. (2007): The challenges of cross-disciplinary research. Social Epistemology, 21: 425–438.

Aghaei C.A., Salehi H., Yunus M.M., Farhadi H., Fooladi M., Farhadi M., Ale Ebrahim N. (2013): A comparison between two main academic literature collections: Web of science and Scopus databases. Asian Social Science, 9: 18–26.

Aksnes D.W., Sivertsen G. (2019): A criteria-based assessment of the coverage of Scopus and Web of Science. Journal of Data and Information Science, 4: 1–21.

Angelstam P., Naumov V., Elbakidze M., Manton M., Priednieks J., Rendenieks Z. (2018): Wood production and biodiversity conservation are rival forestry objectives in Europe's Baltic Sea region. Ecosphere, 9.

Barratt M., Choi T.Y., Li M. (2011): Qualitative case studies in operations management: Trends, research outcomes, and future research implications. Journal of Operations Management, 29: 329–342.

Battaglia M.P. (2008): Non-Probability Sampling. Encyclopedia of Survey Research Methods. California, SAGE Publications: 1–4.

Bauer F., Hansen T., Hellsmark H. (2018): Innovation in the bioeconomy – Dynamics of biorefinery innovation networks. Technology Analysis and Strategic Management, 30: 935–947.

Benbasat I., Goldstein D.K., Mead M. (1987): The case research strategy in studies of information systems. MIS Quarterly, 11: 369–386.

Bentsen N.S., Nilsson D., Larsen S. (2018): Agricultural residues for energy – A case study on the influence of resource availability, economy and policy on the use of straw for energy in Denmark and Sweden. Biomass and Bioenergy, 108: 278–288.

Binz C., Truffer B., Coenen L. (2014): Why space matters in technological innovation systems – Mapping global knowledge dynamics of membrane bioreactor technology. Research Policy, 1: 138–155.

Boons F., Wagner M. (2009): Assessing the relationship between economic and ecological performance: Distinguishing system levels and the role of innovation. Ecological Economics, 68: 1908–1914.

Borge L., Bröring S. (2017): Exploring effectiveness of technology transfer in interdisciplinary settings: The case of the bioeconomy. Creativity and Innovation Management, 26: 311–322.

Bugge M.M., Hansen T., Klitkou A. (2016): What is the bioeconomy? A review of the literature. Sustainability, 8: 691.

Carraresi L., Berg S., Bröring S. (2018): Emerging value chains within the bioeconomy: Structural changes in the case of phosphate recovery. Journal of Cleaner Production, 183: 87–101.

Cavaye A.L.M. (1996): Case study research: A multi-faceted research approach for IS. Information Systems Journal, 6: 227–242.

Corcelli F., Fiorentino G., Vehmas J., Ulgiati S. (2018): Energy efficiency and environmental assessment of papermaking from chemical pulp – A Finland case study. Journal of Cleaner Production, 198: 96–111.

Creswell J.W., Hanson W.E., Clark Plano V.L., Morales A. (2007): Qualitative research designs: Selection and implementation. The Counseling Psychologist, 35: 236–264.

Dautzenberg K., Hanf J. (2008): Biofuel chain development in Germany: Organisation, opportunities, and challenges. Energy Policy, 36: 485–489.

Denscombe M. (2014): The Good Research Guide for Small-Scale Social Research Projects. 5th Ed. UK, McGraw-Hill Education: 36.

Dubé L., Paré G. (2003): Rigor in information systems positivist case research: Current practices, trends, and recommendations. MIS Quarterly, 27: 597–636.

Eisenhardt K.M. (1989): Building theories from case study research. Academy of Management Review, 14: 532–550.

Etikan I. (2016): Comparison of convenience sampling and purposive sampling. American Journal of Theoretical and Applied Statistics, 5: 1.

European Commission (2018): A Sustainable Bioeconomy for Europe: Strengthening the Connection between Economy, Society and the Environment. Publications Office of the EU: 1–103. Available at https://doi.org/10.2777/478385 (accessed Jan 3, 2020).

European Commission (2019): The European Green Deal, Communication from the Commission to the European Parliament, the European Council, the Council, the European Economic and Social Committee and the Committee of the Regions. Eu¬ropean Commission. Available at https://eur-lex.europa.eu/ legal-content/EN/TXT/?uri=COM

FAO (2019): Towards Sustainable Bioeconomy Guidelines. Food and Agricultural Organization of the United Nations. Available at http://www.fao.org/3/a-bs923e.pdf (accessed Jan 10, 2020).

Flood R.L. (1999): Rethinking the Fifth Discipline: Learning within the Unknowable. London, New York, Psychology Press: 92.

Fuldauer L.I., Parker B.M., Yaman R., Borrion A. (2018): Managing anaerobic digestate from food waste in the urban environment: Evaluating the feasibility from an interdisciplinary perspective. Journal of Cleaner Production, 185: 929–940.

Georgescu-Roegen N. (1977): Inequality, limits and growth from a bioeconomic viewpoint. Review of Social Economy, 35: 361–375.

Golembiewski B., Sick N., Bröring S. (2015): The emerging research landscape on bioeconomy: What has been done so far and what is essential from a technology and innovation management perspective? Innovative Food Science Emerging Technologies, 29: 308–317.

Greene J., Hall J. (2010): Dialectics and pragmatism: Being of consequence. In: Tashakkori A.M., Teddlie C.B. (eds): Handbook of Mixed Methods in Social Behavioral Research. United Kingdom, SAGE: 119–144.

Grünbaum N.N. (2007): Identification of ambiguity in the case study research typology: What is a unit of analysis? Qualitative Market Research, 10: 78–97.

Harzing A.W., Alakangas S. (2016): Google Scholar, Scopus and the Web of Science: A longitudinal and cross-disciplinary comparison. Scientometrics, 106: 787–804.

Kabyanga M., Balana B.B., Mugisha J., Walekhwa P.N., Smith J., Glenk K. (2018): Economic potential of flexible balloon biogas digester among smallholder farmers: A case study from Uganda. Renewable Energy, 120: 392–400.

Kardung M., Cingiz K., Costenoble O., Delahaye R., Heijman W., Lovrić M., van Leeuwen M., M'Barek R., van Meijl H., Piotrowski S., Ronzon T., Sauer J., Verhoog D., Verkerk P.J., Vrachioli M., Wesseler J.H.H., Zhu B.X. (2021): Development of the circular bioeconomy: Drivers and indicators. Sustainability, 13: 413.

Maimbo H., Pervan G. (2005): Designing a case study protocol for application in IS research. In: Proceedings Pacific Asia Conference on Information Systems, PACIS 2005, Bangkok, Thailand, July 7–10, 2005: 106.

Markard J., Raven R., Truffer B. (2012): Sustainability transitions: An emerging field of research and its prospects. Research Policy, 41: 955–967.

Mengistu T.W., Gupta S., Birner R. (2018): Analysis of maize biomass use in Ethiopia and its implications for food security and the bioeconomy. Food Security, 10: 1631–1648.

Merriam S.B. (1998): Qualitative Research and Case Study Applications in Education. San Francisco, Jossey-Bass: 26–43.

Miles M.B., Huberman A.M. (1994): Qualitative Data Analysis: An Expanded Sourcebook. London, Sage Publications: 117.

Naumov V., Manton M., Elbakidze M., Rendenieks Z., Priednieks J., Uhlianets S., Yamelynets T., Zhivotov A., Angelstam P. (2018): How to reconcile wood production and biodiversity conservation? The Pan-European boreal forest history gradient as an "experiment". Journal of Environmental Management, 218: 1–13.

Newton R.W., Little D.C. (2018): Mapping the impacts of farmed Scottish salmon from a life cycle perspective. The International Journal of Life Cycle Assessment, 23: 1018–1029. OECD (1998): 21st Century Technologies: Promises and Perils of a Dynamic Future. Paris, OECD Publishing: 30. Patermann C., Aguilar A. (2018): The origins of the bioeconomy in the European Union. New Biotechnology, 40: 20–24.

Purkus A., Hagemann N., Bedtke N., Gawel E. (2018): Towards a sustainable innovation system for the German wood-based bioeconomy: Implications for policy design. Journal of Cleaner Production, 172: 3955–3968.

Riffe D., Lacy S., Fico F., Watson B. (2014): Analyzing Media Messages: Using Quantitative Content Analysis in Research. 3rd Ed. New York, London, Routledge: 101.

Robson C., McCartan K. (2016): Real-World Research. 4th Ed. Chichester, West Sussex, Wiley: 45–66. Runeson P., Höst M. (2009): Guidelines for conducting and reporting case study research in software engineering. Empirical Software Engineering, 14: 131–164.

Saldaña J. (2013): The Coding Manual for Qualitative Researchers. 2nd Ed. Los Angeles, London, New Delhi, Singapore, Washington DC, SAGE: 1–41.

Sheppard A.W., Gillespie I., Hirsch M., Begley C. (2011): Biosecurity and sustainability within the growing global bioeconomy. Current Opinion in Environmental Sustainability, 3: 4–10.

Singlitico A., Goggins J., Monaghan R.F.D. (2018): Evaluation of the potential and geospatial distribution of waste and residues for bio-SNG production: A case study for the Republic of Ireland. Renewable and Sustainable Energy Reviews, 98: 288–301.

Skvortsova T.A., Denisova I.P., Romanenko N.G., Sukhovenko A.V. (2018): Innovations and support for quality in agriculture: A case study. European Research Studies Journal, 21: 423–431.

Chapter 2

Do Biodegradable Food Packaging Films from Agro-food Waste Pay off? A Cost-Benefit Analysis in the Context of Europe

Abstract

Bio-based polymers are increasingly attracting attention as a solution to reducing the consumption of non-renewable resources and curbing the accumulation of fossil-based plastic waste. In this study, we analyze the economics of a new packaging film based on a polylactic acid–polyhydroxybutyrate blend (PLA-PHB), with PHB obtained from agro-industrial residues (potato peels). We model various sizes of biorefineries using the new biotechnology in Europe. For a four-year payback period, which is generally accepted in the industry, the calculated minimum product selling price ranges from 9.7 euros per kilogram to 37.2 euros per kilogram, depending, among other factors, on the production capacity of the biorefinery. We have incorporated the uncertainty over the model parameters in a Monte Carlo simulation and investigated the relative impact of individual factors on the minimum product selling price. Overall, the results indicate that the bio-based feedstock availability is the most influential factor on the profitability of the new biotechnology.

Keywords: circular bioeconomy; bioplastics; sustainability; resource management; techno-economic analysis

This chapter is based on: Tassinari, G., Bassani, A., Spigno, G., Soregaroli, C., & Drabik, D. (2023). Do biodegradable food packaging films from agro-food waste pay off? A cost-benefit analysis in the context of Europe. Science of the Total Environment, 856, 159101.

2.1 Introduction

Plastics facilitate our daily lives, and they are particularly useful in packaging applications (Asgher et al., 2020). With the global population rising, the production of plastics continues to increase (Shahzad et al., 2017; Kim et al., 2020) and has already exceeded 360 million metric tons (Mt) per year (PlasticEurope, 2020). Europe contributes to 16% of the global plastic production, and this mainly (39.6%) covers the requirements of European plastic packaging converters (PlasticEurope, 2020). Packaging is one of the most problematic types of plastic waste, given its usual single short-use design (CIEL, 2019). In fact, over 8 Mt of plastics end up in the oceans every year (Ellen MacArthur Foundation, 2016). Due to the fossil-based nature of polymers, the so-called conventional plastics are highly recalcitrant to natural degradation, making them a growing environmental threat. Several initiatives have been developed to curb the problem of petrochemical-based plastic waste accumulation, including fees, environmental taxes, and legislative bans on certain single-use plastics (Patrício Silva et al., 2020).

Irrespective of their detrimental environmental effects, plastics are essential for food storage and distribution (Shahzad et al., 2017). Consumers and suppliers currently rely on fresh foods packaged in single-use plastic containers to avoid food contamination and extend shelf life (Patrício Silva et al., 2021). Recently, the COVID-19 pandemic has changed the perceived role of plastics in modern society and re-affirmed humanity's dependence on plastic materials (Patrício Silva et al., 2020). The lockdowns have led to changes in consumer habits as a result of the increase in food delivery and plastic packaging (Filho et al., 2021). Furthermore, safety concerns related to cross-contamination when reusing containers or bags led to several jurisdictions to reverse or temporarily postpone plastic reduction policies (Prata et al., 2020). Thus, under the new circumstances, it has become ever more compelling to rethink and redesign plastics in a sustainable way (Patrício Silva et al., 2020).

The development of bio-based, biodegradable polymers has attracted the attention of governments, industries, and academia as a potentially sustainable solution to the currently used fossil-based polymers (Koller et al., 2013; Arrieta et al., 2015; Kim et al., 2020). Recycling long-lasting plastic polymers as packaging materials is often impractical due to food contamination (Siracusa et al., 2008). In contrast, most commercialized biopolymers are biodegradable. Thus, bioplastics can reduce the landfilling of food waste and packaging while preventing plastic leakage into the environment (Kakadellis and Harris, 2020). The end-of-life alternatives, such as industrial composting or anaerobic digestion, can reduce greenhouse gas emissions (Posen et al., 2017; Dilkes-Hoffman et al., 2018) and recover organic carbon, which can be further recycled, for example, in restoring the fertility of depleted soils (Vidal et al., 2007).

The demand for bioplastics is growing, but their current market share is less than one percent. The annual global production of bioplastics is estimated at 2.11 Mt, of which 26.3% is used for flexible packaging application and 21% for rigid packaging (European Bioplastics, 2020). More than 24% of bioplastics are currently produced in Europe (European Bioplastics, 2020). Investments in bioplastics applications are typically hampered by the high capital and production costs involved (Lopez-Arenas et al., 2017; Shahzad et al., 2017). The higher costs entailed in the production of bioplastics as compared to conventional fossil-based polymers are associated with the chemical complexity of bio-based feedstocks, the fermentation or extraction conditions, and the downstream separation process for product recovery (Naranjo et al., 2013; Lopez-Arenas et al., 2017). Among the biodegradable polymers, polylactic acid (PLA) is by far the most commercially developed (Hatti-Kaul et al., 2020) due to its relatively low production cost (Battegazzore et al., 2014; Ioannidou et al., 2022). However, the use of PLA in food packaging is limited due to its stiffness, brittleness, and poor thermo-mechanical properties (Asgher et al., 2020). Therefore, current research efforts are focused on overcoming these inherent limitations through the development of bio-based plastic films that combine PLA with other biopolymers, such as polyhydroxybutyrate (PHB) (Marra et al., 2016; Saravanan et al., 2016; Asgher et al., 2020).

In this context, recent research activities in industrial settings supported by the European Union and the Bio-based Industries Consortium validated the technical feasibility (with regard to the specific final requirements for food packaging) of an innovative product based on a PLA-PHB blend plasticized with oligomeric lactic acid (OLA) (Carolis, 2021; BBI JU, 2022). All the plastic blend components of the film come from renewable resources. That is, PHB production utilizes potato peel residues from the agriculture and food industries, while both PLA and OLA can be produced from starch (Kwan et al., 2015; Asgher et al., 2020). However, the economic feasibility of the investment is just as critical to the sustainability and technological feasibility of the product. There are studies that have investigated the techno-economic feasibility of PLA (e.g., Dornburg et al., 2006; Ioannidou et al., 2022) and PHB (e.g., Lopez-Arenas et al., 2017) separately, but no one has, as of yet, evaluated the profitability of a PLA-PHB blend plasticized with OLA.

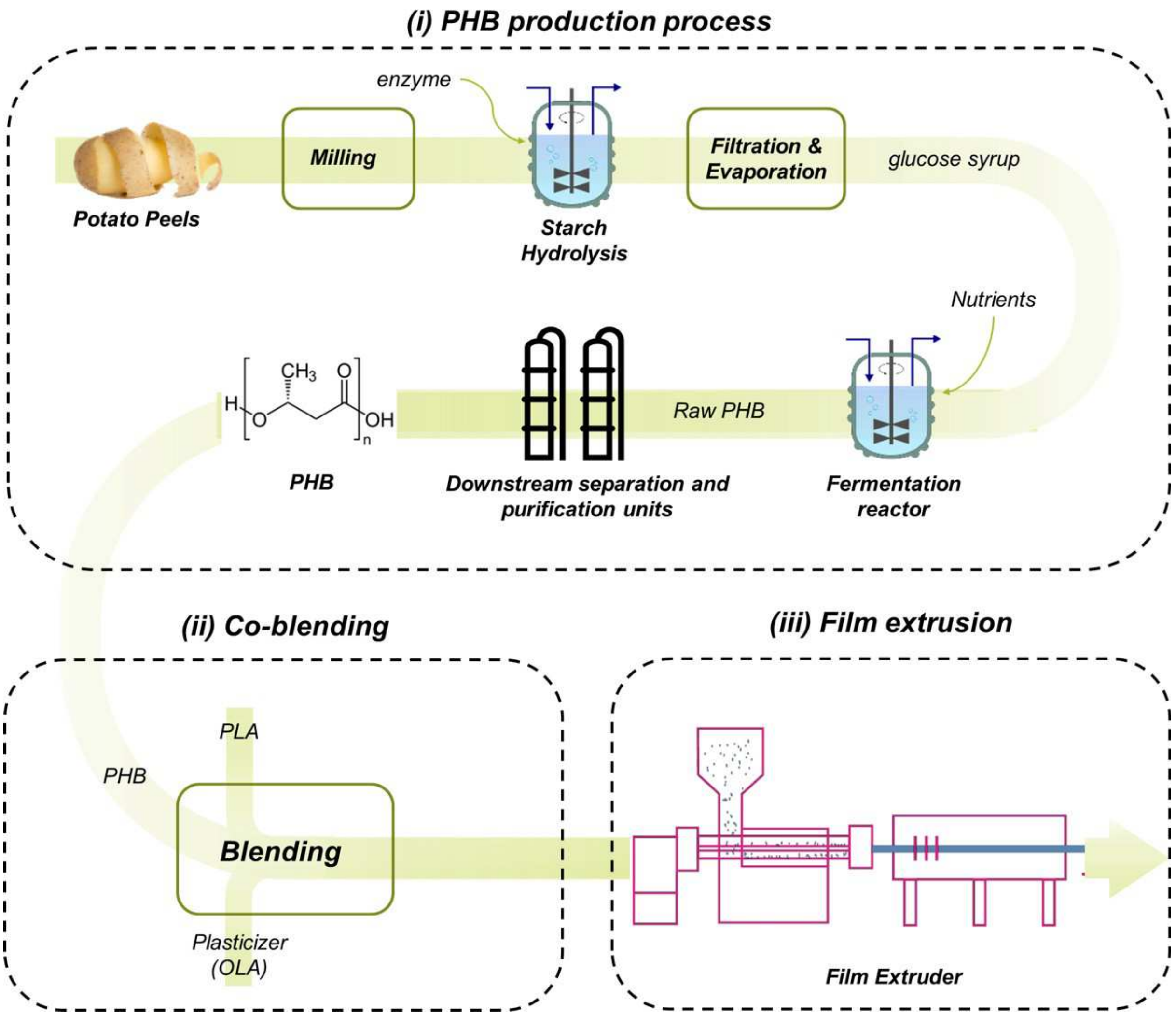
This paper explores ex ante the economic feasibility of this novel bio-based packaging film, which fulfils all biodegradability and compostability standards as verified by biodegradability, disintegration, and plant growth tests (CORDIS, 2022). The specific objective of this study is to evaluate the minimum product selling price (MPSP) for different industrial scales in Europe under conditions that would attract investors, such as a low payback period. We performed a cost-benefit analysis of the preliminary process design by considering major investment and operating costs over a selected time horizon. Finally, we reviewed and analyzed the uncertainties involved in the cost estimation using Monte Carlo simulations. The method presented here not only provides new insights into the economic feasibility of investing in biodegradable and compostable bio-based plastics, but can also be applied for the economic evaluation of other bio-based productions, even those characterized by a low technology readiness level (TRL).

2.2 Material and methods

2.2.1 Process description

The production of the bio-based plastic film under study involves a multi-step process. The biorefinery concept includes (i) the production of PHB from potato peels, (ii) co-blending of PHB with PLA and OLA (as the plasticizer), and (iii) film extrusion (Figure 2.1).

Figure 2.1: The production process of the NEWPACK bio-based plastic film and biorefinery concept economically assessed in the article.



The production of PHB from potato peels involves six main processing steps: milling, enzymatic hydrolysis, biomass removal, evaporation, fermentation, and PHB purification. Feedstock is initially collected and stored in the biorefinery plant under refrigerated conditions for a short period until processing. Potato peels were preliminarily selected as feedstock for PHB production, given the importance of the potato crop as long as the rich starch content. Potatoes are the third most important staple food crop in the world, produced on all continents and with increasing global production (Birch et al., 2012). Unlike other staple crops (such as wheat, corn, rice, and soybeans), its global trade is negligible because it is mostly produced and consumed locally, making it more resilient to financial, commercial, and political factors that influence price, availability, and relative scarcity (Campos & Ortiz, 2019). In addition, the potato is an easy plant to grow, providing nutrients quickly, on less soil than any other food crop, and in almost any habitat (Mullins et al., 2006). Finally, while fresh potato consumption is declining, potatoes are being processed more to meet the demand for convenience and fast food, consequently generating large amounts of peels that pose a serious disposal problem for the industry (Schieber & Saldaña, 2009).

At the start of the production line, the potato peels are pre-treated by milling and enzymatic hydrolysis to convert starch to glucose. The potato peel slurry formed is then filtered to eliminate fibers. The sugar liquid (glucose syrup) is concentrated via evaporation. After these pre-treatment steps, the concentrate is fermented with the addition of nutrients (i.e., ammonium sulfate, citric acid, and potassium sulfate) and a gram-negative microorganism, which stores PHB granules in the cytosol in response to a phosphate limitation of the environment when an external carbon source is present in excess. Next, PHB is extracted from the fermentation broth. Different agents (including NaOH, H2SO4, and NH4OH) are used for pH adjustments in these steps.

The obtained purified PHB is blended with PLA (¿50% by weight of PLA). The exact value cannot be declared because it is the Intellectual Property of the NEWPACK project. Next, 4 phr (parts per hundred resin) of OLA is added as a plasticizer to the PHB-PLA blend. OLA is a lactic acid additive with an oligomeric structure that can be used as a plasticizer for PLA/PHB blends to achieve biofilms with increased flexibility without loss of transparency. OLA, developed and patented internationally by Condensia Quimica (Spain), was chosen as it is completely biodegradable and bioavailable being extracted from bio-renewable raw materials. In this study, we assumed that both PLA and OLA are purchased from external suppliers. The final stage involves film extrusion and preparation, with an expected 10% loss of the initial processed bio-plastic blend. The valorization pathway, therefore, is based on eight significant steps (i.e., functional units) (Gerrard, 2000) in total, including the six steps for PHB production and the next two stages of blending and extrusion.

2.2.2 Cost-benefit assessment

Due to the low TRL of the project (TRL 5-6), we do not have information on the exact industrial configuration of the process. Furthermore, data as sensitive as historical costs of similar commercial plants are rarely shared by companies or made publicly available. Therefore, we relied on the early capital cost estimation and conceptual framework of Cristóbal et al. (2018), who evaluated several biorefinery concepts based on food waste availability, including the waste streams of potatoes and potato-based products. Adopting this approach, we assessed the economics of the valorization pathway for the PLA-PHB bio-based plastic film at the European level, assuming different industrial scales and logistics settings.

Food waste quantification and plant processing capacity: Cristóbal et al. (2018) quantified food waste streams by triangulating different data sources, including EUROSTAT, PRODCOM, and various pieces of literature on industrial production efficiency coefficients. Given the need for high compositional homogeneity of the feedstock, the authors only considered food waste from the food processing stage that was to be further processed by biorefineries. Based on data from different industrial productions[1], Cristóbal et al. (2018) estimated that 2.34 Mt of potato peels was available in Europe in 2015. Compared to 2015, potato production increased by 6% in 2019 (Eurostat, 2021d). In this study, therefore, we considered that a total of 2.48 Mt of potato peels was available. Based on this, we derived different plant processing capacities by modeling a variable number of biorefineries (from 7 to 140) that we assumed would operate in the European Union (EU28). According to this computation, each scenario is assigned a different number of plants of the same size.

Investment costs: The costs of purchasing and installing all equipment, i.e., inside battery limits (ISBL), were estimated using a step-counting method. Cristóbal et al. (2018) used Bridgwater's correlation (Eq.1) to represent ISBL as a function of the number of functional units (i.e., significant steps involved in the process, N_f) and plant processing capacity (i.e., the amount of material passing through the process per year, Q/s).

$$ISBL = \left[401,600 + 1.304\left(\frac{Q}{s}\right)\right] N_f \tag{1}$$

Bridgwater developed several correlations based on historical plant cost data that are suitable for fermentation processes. Unlike many other step-counting techniques (for a review, see Tsagkari et al., 2016), the advantage of Eq.1 is that it uses the amount of inflow instead of outflow, since the latter is not a reliable metric for estimating plant size given the low conversion factor that generally characterizes waste-based biorefineries.

[1] Productions of potato starch and (frozen) potatoes, whether raw or cooked, prepared or preserved, cut or sliced, in the form of flour, meal, flakes, granules, pellets, or crisps.

Eq.1 estimates ISBL in UK 1976 pounds. The value was, therefore, adjusted to 2019 euros based on the difference in the chemical engineering plant cost index for 1976 (192.1) and 2019 (607.5) (Jenkins, 2020), and the exchange rate of 1.14 euros per pound (Exchange Rate, 2021) (Eq.2).

$$ISBL_{2019} = ISBL_{1976} \frac{CEPCI_{2019}}{CEPCI_{1976}} \left(1.12 \frac{euros}{pound}\right) \tag{2}$$

Finally, the total capital investment (TCI) was derived via factorial estimation. Factors reported by Cristóbal et al., 2018 were applied to convert delivered equipment costs into fixed-capital investment components, start-up expenses, and working capital. A single factor (so-called Lang Factor) of 2.9 was derived from these expenditure items and multiplied with ISBL to determine the TCI.

Biomass transportation cost: For determining the feedstock transportation cost, the average cost[2] of 0.14 euros per metric ton-kilometer (tkm) (Schade et al., 2006) was multiplied by the road distance estimated using Eq. 3 below. The function derived by Cristóbal et al. (2018) estimates international and intranational transport distances covered by road to acquire potato peels. The function was calibrated by taking into account inter- and intra-national transportation distances under the assumption that the estimation included seven plants located in the main potato peel-producing countries (i.e., the Netherlands, Germany, Italy, and the United Kingdom) and 70 plants distributed across each waste-producing country in the EU. In this study, we extended the upper bound to 140 facilities, thus simulating an extreme scenario in which the distance (and, hence, also the transportation cost) for waste collection is zero (which implies that the plant is located in the same industry that processes potatoes).

$$\text{Distance (tkm)} = -5 \cdot 10^6 \cdot N_{plants} + 7 \cdot 10^8 \tag{3}$$

Operating costs: The operating costs we modeled include raw material cost, labor cost, and utility cost. Water and waste treatment costs were considered negligible[3] since wastewater and solid waste can be recovered from different process stages and recycled within the system.

The raw material expense was calculated based on the linear scaled-up process mass streams validated at a pilot level (BBI JU, 2022). The prices of feedstocks and chemicals were collected from several sources (see Supplementary Information Table S1). In terms of

[2]It represents the average transport operating cost per tkm in the EU for heavy duty vehicles, including non-refrigerated transport. Depending on the scenario, this limitation may lead to an underestimation of transport costs.

[3]While the operating costs of waste treatment can be considered negligible, fixed costs refer to the processing efficiency of the plant and thus include costs related to the purchase of waste management equipment.

labor costs, according to the method proposed by Cristóbal et al. (2018), we multiplied the hourly labor cost in the EU (Eurostat, 2021b) by the total working hours of the biorefinery operators. We assumed that the plant is operated for 24 hours a day, 365 days a year. Given that, in Europe, employees work an average of 1,960 hours a year (Cristóbal et al., 2018), at least 4.5 operators are needed to ensure the continuity of the operation. Furthermore, the plant needs a certain number of operators at any given time to ensure the operation of the plant. Using the correlation (Eq. 4), the plant requires the presence of 11.6 operators per shift (N_{OL}) for the operation of eight machine units (N_{np}) (i.e., refrigerator, grinding, filters, evaporators, extruders, heaters, fermentation, and purification units) and for two particulate (P) handling steps. Therefore, the total operational labor is 53 (4.5×11.6) operators per plant (excluding support or supervision staff).

$$N_{OL} = (6.29 + 31.7 \cdot P + 0.23 \cdot N_{np})^{0.5} \tag{4}$$

Utilities refer to the energy consumption for storing and pre-treating raw materials, fermenting glucose syrup, extruding, heating plasticizers, cutting, and blowing the blend. At the pilot-scale level, the process requires 7.06 kWh per kg of packaging film. With regard to the electricity cost, the average European rate for non-household consumers of 0.12 euros per kWh was considered (Eurostat, 2021a). With regard to the heating requirements, the fuel was assumed to be natural gas and the price was set to 0.0315 euros per kWh (Eurostat, 2021c), based on the reference year 2019.

Net present value and threshold price for the bio-based packaging film: Net present value (NPV) is an important indicator for analyzing the profitability of a project and represents the sum of future cash flows in their present value equivalents. A positive NPV means that the value of discounted cash inflows is greater than the discounted cash outflows over a considered period. We calculated NPV using the model set-up in Golberg et al. (2021). In this model, in the first year ($t = 0$), the project's total capital investment (I) is paid. A fraction (α) of it is covered with a bank loan at the interest rate ρ; the rest is covered by the investor's own sources. The loan is paid back in N years in fixed annuities (A) starting from $t = 1$, after the interest has been charged. The annuity of the loan is calculated as

$$A = \frac{\alpha I \rho}{1 - (1 + \rho)^{-N}} \tag{5}$$

At the end of the first year, we assume that the plant will receive the first net annual benefit (NB). The NB is assumed to be constant over the modeled horizon and is equal (in nominal terms) to the gross profit per year estimated in the base year. In addition to the operating costs, the facility incurs ongoing maintenance costs, which are set as a fraction (β) of the initial investment. The maintenance cost is assumed to grow exponentially at an annual rate (g). Finally, a salvage (scrap) value is calculated at the end of the discounting period. It was set as a fraction (γ) of the initial investment. To convert all future cash flows into their present value equivalents, we adopted the real discount rate (r) (Eq. 6), which was calculated using a 4-percent nominal discount rate (i) (European Commission, 2014) and 1.9-percent inflation rate (m) (Knoema, 2021). The real discount rate would be equal to 2.1 percent at the baseline.

$$r = \frac{i - m}{1 + m} \tag{6}$$

Based on the above set-up, the NPV can be calculated as follows.

$$NPV = -(1-\alpha)I - A\left[\frac{(1+r)^N - 1}{r(1+r)^N}\right] + NB\left[\frac{(1+r)^{\tau+1} - 1}{r(1+r)^{\tau}}\right] +$$

$$-\beta I\left(\frac{1+g}{r-g}\right)\left[1 - \left(\frac{1+g}{r-g}\right)^{\tau}\right] + \frac{\gamma I}{(1+r)^{\tau}} \tag{7}$$

If the NPV is set to 0, the MPSP of the bio-based packaging film for the payback period τ (four years at the baseline) can be calculated. A quick return on investment is required to attract investors who are generally discouraged by high upfront costs and market uncertainty (Heck et al., 2014). Table 1 presents the baseline values of the key parameters we used in the model, as well as their domain (i.e., the lower and upper bounds) used in the uncertainty analysis below.

2.2.3 Uncertainty analysis

Because we did not have access to all the necessary information, we made several assumptions about the project's economics that inevitably entail uncertainty. To address part of this uncertainty, we performed Monte Carlo simulations. The PERT distribution (or three-point estimation technique) is used to characterize the uncertainty of each variable (Table 2.1 and S1). The PERT distribution is commonly adopted by decision makers in project management because of its ease of use and practicality (Peters, 2016). The distribution is defined by three parameters: min, mode, and max. We used the baseline values of the parameters as the mode of the PERT distribution and their extreme values as min and max. In cases where the domain was unknown, we reduced or increased the baseline value by 30%, except for the number of years for repayment of the bank loan, which was held constant at 10 years.

Once the distributions were defined, we performed 10,000 iterations (for each biorefinery size) by randomly and independently selecting a value from each distribution. We then analyzed how the MPSP for each iteration changes as only the number of operating plants in Europe varies. We visualized and compared the results with a jitter plot in the R software (R-4.0.0, package ggplot2) (Wickham, 2016). Finally, we investigated the relative impact of individual factors on MPSP by running an ordinary least square (OLS) regression of the MPSP on all model parameters, including the varying number

Table 2.1: Type of research questions

Item	Baseline	Min	Max	Unit	Source
Potato peel availability	2.48	-	-	Mt/year	(Cristóbal et al., 2018)
ISBL	(Eq. 1)	-	-	Euros	Estimated
TCI	2.9 ISBL	-	-	Euros	Estimated
Transport cost	0.14	-	-	Euros/tkm	(Schade et al., 2016)
Labor cost	28.2	27.7	31.4	Euros/h	(Eurostat, 2021b)
Electricity price	0.122	0.119	0.125	Euros/kWh	(Eurostat, 2021a)
Heat (by natural gas)	0.03	0.027	0.033	Euros/kWh	(Eurostat, 2021c)
Length of the loan (N)	10	-	-	Years	Assumed
Share of the investment from a loan ($alfa$)	0.5	-	-	Percent	(Golberg et al., 2021)
Annual interest rate on the loan rate (rho)	0.05	-	-	Percent	(Golberg et al., 2021)
Maintenance cost fraction ($beta$)	0.005	-	-	Percent	(Golberg et al., 2021)
Annual growth rate of the maintenance cost (g)	0.005	-	-	Percent	(Golberg et al., 2021)
Salvage value fraction ($gamma$)	0.05	-	-	Percent	(Golberg et al., 2021)
Nominal discount rate (i)	0.04	-	-	Percent	(European Commission, 2014)
Inflation (m)	0.019	0.013	0.02	Percent	(Knoema, 2021)
Payback (tau)	4	2	7	Years	(Heck et al., 2014)

of biorefineries (7–140). We converted all regression variables into log values to facilitate comparison of the impact of individual factors (the OLS regression coefficients are interpreted as elasticities of the MPSP over changes in each model parameter).

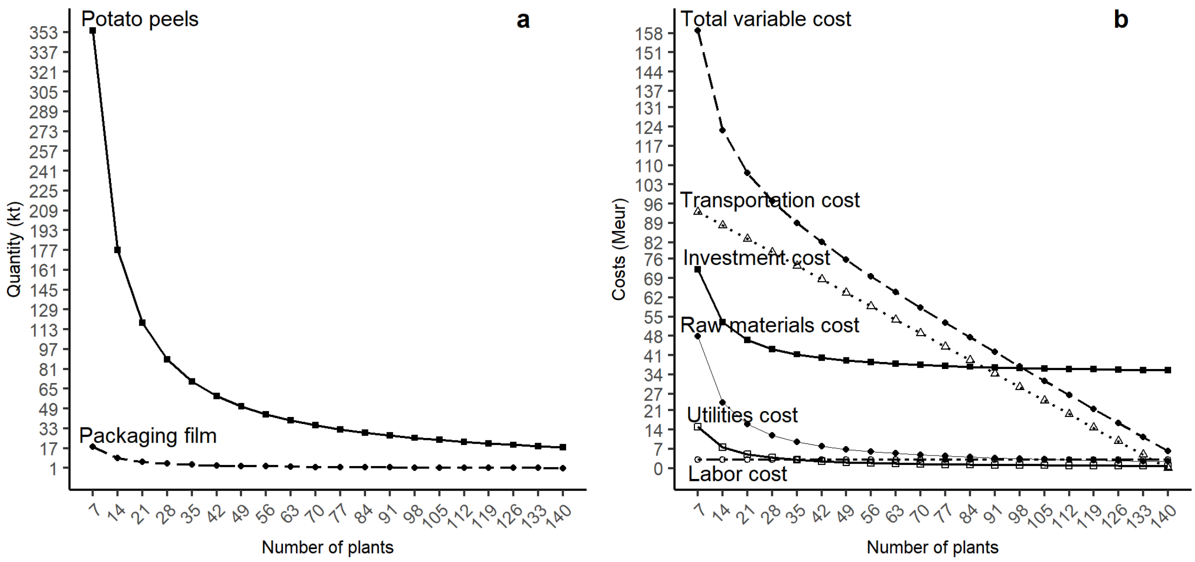
2.3 Results

2.3.1 Ex-ante cost estimation

We evaluated the economics of the valorization pathway for a different number of homogenous plants. By keeping the total feedstock amount constant and increasing the number of biorefineries from 7 to 140, we observed that the annual processing capacity per plant decreases from 354.28 kilotons (kt) to 17.71 kt (Figure 2a) and the annual production of packaging film per plant decreases from 18 kt to 0.9 kt. Initially, the plant processing capacity decreases rapidly, but the decline (i.e., the rate of change) slows down as the number of plants increases. Next, we estimated the (fixed and variable) costs for each plant size.

As Figure 2.2b shows, plant size influences (in absolute terms) all cost items except for labor cost, which is constant. According to the model, the investment cost, the raw material cost, and the utility cost depend directly on mass inflows and, therefore, also show an inverse relationship with the number of plants, as observed for processing capacity. On the other hand, transportation cost depends directly on the number of plants and, therefore, the relationship between the two is linear, with a constant decrease of 0.7 million euros as the number of plants increases.

Figure 2.2: Trends in plant processing/production capacity (a) and costs (b) as a function of the number of plants.



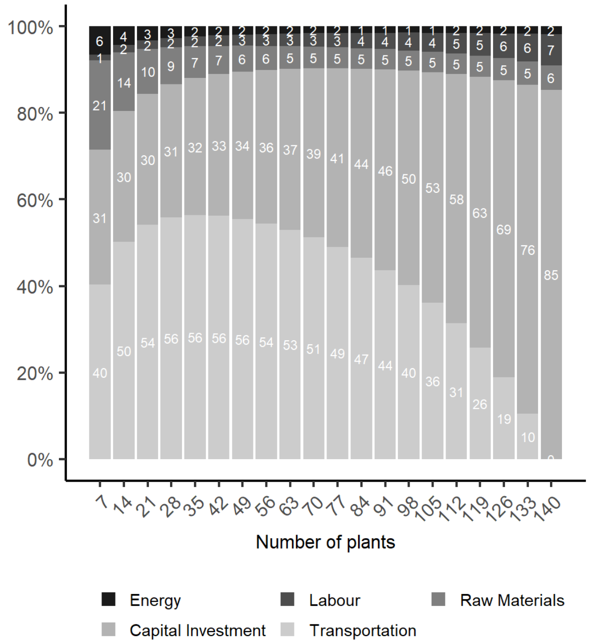
Based on the above trends, the modeled costs contribute to the total expenditures differently (Figure 2.3). For up to 140 plants, the capital cost and transportation cost are the most decisive factors. The investment cost alone dominates a plant's cost structure when there are 98 or more plants in the market, whereas the transportation cost is dominant when the total number of plants varies between 14 and 70. The contribution of labor cost is minimal but increases as plant size decreases.

2.3.2 Profitability analysis

In this section, we present the MPSP for the bio-based packaging film that can achieve the investment break-even point in four years (in the baseline). The results are summarized in Table 2.2.

The estimated MPSP in the last column of Table 2 shows an inverted U-shape as the number of plants increases. MPSP reaches a maximum value of 37.2 euros per kg in the 77-plant scenario and equals 9.7 euros per kg and 15.4 euros per kg for plants with the largest and smallest processing capacities, respectively. The inverted U-shape is a result of the combined effect of the average fixed cost and average transportation costs, which influenced the unit cost the most.

Figure 2.3: The proportion of each cost item in the total cost structure according to the modelled number of plants.



As Table 2 shows, the average fixed cost decreases when the level of output (i.e., plant size) increases. This is because the constant term in the fixed cost function (Eq. 1) is distributed over a larger number of units of output in the model. The average transportation cost—defined as the ratio of the total transportation cost to a plant's capacity—is not monotonic (see Supplementary Information Figure S1). This is because the numerator and denominator of the ratio both decrease with a higher number of plants but at different rates. As a result, the average transportation cost increases with plant size until it reaches 1.8 kt per year (in the 70-plant scenario), after which it decreases.

2.3.3 Uncertainty analysis

We conducted a Monte Carlo simulation to capture the uncertainty surrounding our assumptions. The key parameters were assumed to follow a PERT distribution defined by the minimum, most likely, and maximum parameter values (Table 1 and S1). For each plant processing capacity, we randomly and independently drew model parameters 10,000 times from their respective distributions and calculated the corresponding MPSP values (Figure **??**a). To prevent overplotting, we used the jittering method, whereby the MPSP values on the y-axis were randomly shifted along the x-axis.

Figure 4a shows the highest point density around the baseline results (illustrated as white triangles). However, as the MPSP increases, so does its variation for a given number of biorefineries (this is most visible when there are 77 biorefineries in the market). The

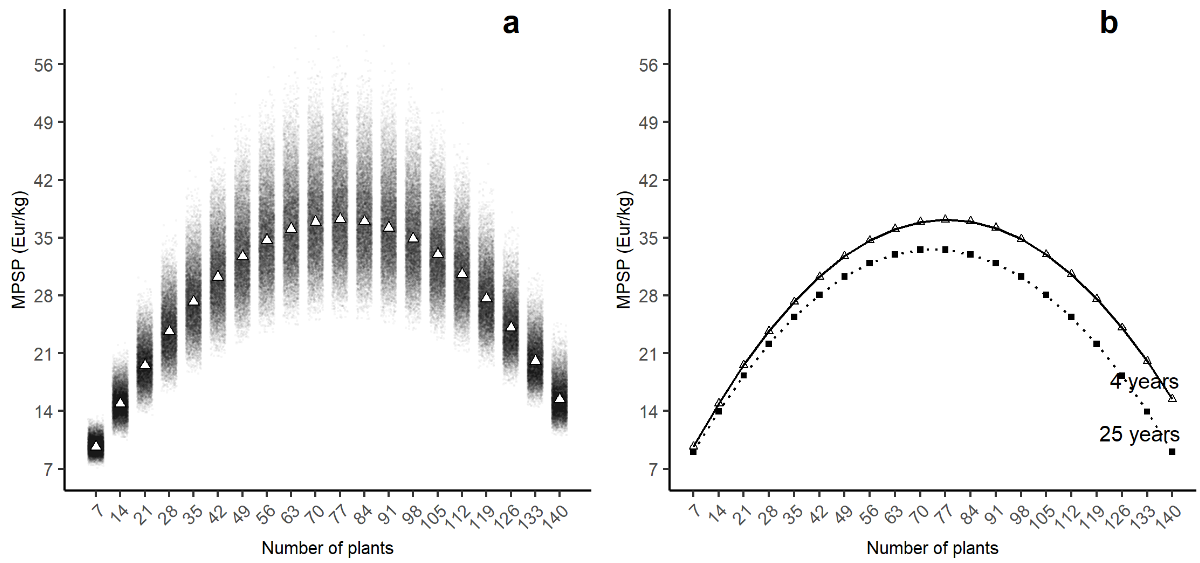
Table 2.2: Average costs and calculation of MPSP at the baseline (with a payback time of four years) for different industrial scales.

Number of plants	Processing capacity (kt/year)	Production (kt/year)	Average Total Capital Investment (Euros/kg)	Average Labor cost (Euros/kg)	Average Transportation cost (Euros/kg)	Average Energy cost (Euros/kg)	Average Raw material cost (Euros/kg)	MPSP (Euros/kg)
7	354.3	18	4	0.2	5.2	0.8	2.7	9.7
14	177.1	9	5.9	0.3	9.8	0.8	2.7	14.9
21	118.1	6	7.7	0.5	13.9	0.8	2.7	19.6
28	88.6	4.5	9.6	0.7	17.4	0.8	2.7	23.7
35	70.9	3.6	11.5	0.8	20.4	0.8	2.7	27.2
42	59.1	3	13.3	1	22.9	0.8	2.7	30.3
49	50.6	2.6	15.2	1.2	24.8	0.8	2.7	32.7
56	44.3	2.3	17.1	1.3	26.1	0.8	2.7	34.7
63	39.4	2	18.9	1.5	27	0.8	2.7	36.1
70	35.4	1.8	20.8	1.7	27.2	0.8	2.7	36.9
77	32.2	1.6	22.6	1.8	26.9	0.8	2.7	37.2
84	29.5	1.5	24.5	2	26.1	0.8	2.7	37
91	27.3	1.4	26.3	2.2	24.7	0.8	2.7	36.2
98	25.3	1.3	28.2	2.3	22.8	0.8	2.7	34.9
105	23.6	1.2	30.1	2.5	20.4	0.8	2.7	33
112	22.1	1.1	31.9	2.6	17.3	0.8	2.7	30.6
119	20.8	1.1	33.8	2.8	13.9	0.8	2.7	27.6
126	19.7	1	35.7	3	9.8	0.8	2.7	24.1
133	18.7	1	37.5	3.1	5.2	0.8	2.7	20.1
140	17.7	0.9	39.5	3.3	0	0.8	2.7	15.5

standard deviation values (Table 2.3) indicate that the largest industrial scale showed the least variation in MPSP. Furthermore, Table 3 shows that the means are greater than the medians in all conditional (based on the number of biorefineries) distributions. This means that the distributions are positively skewed; that is, they have a longer tail toward higher selling prices.

With regard to the payback period, we chose the interval of two to seven years in the Monte Carlo simulations to reflect the need for the investment to be attractive to industries. This domain should, however, be adjusted if supportive policies are available that result in investors accepting longer payback periods (e.g., grant aid). As illustrated in Figure 4b, in a scenario with a longer payback time that is equal to the plant lifetime (i.e., 25 years), the mean MPSP would decrease, especially for smaller plants (i.e., when there are more plants). This is because the amortization of the fixed cost has a greater impact on smaller plants. Larger facilities, on the other hand, would be less affected by this change as they are more constrained by variable costs.

Figure 2.4: Estimations of MPSP according to the number of plants: (a) Probability distributions of MPSP from Monte Carlo simulations and (b) comparison of mean MPSP at different payback times.



2.3.4 Factors that affect the minimum selling price

Understanding the relative impact of individual factors on the MPSP would be helpful for investors considering the construction of a new production plant or policymakers debating policies to support the new industry. One way of determining the relative impact of exogenous parameters used in our model on MPSP is to run a Monte Carlo simulation with a high number of runs (we performed 100,000 runs). Each run contains a unique random combination of parameters that results in a value of MPSP. In the next step, we tease out the effect of the variation in the exogenous parameters on the variation in MPSP by running an OLS regression.

The column titled "untransformed regression" in Table 4 presents the coefficient estimates for a model with untransformed values of the exogenous factors. We included not only the number of plants but also the square of those numbers to capture a possible non-linear relation between MPSP and n, as suggested by Figure 4. Some of the parameters, but not all of them, were found to have a significant effect (at least at the five-percent level of significance) on MPSP.

While the above results are informative, it is not possible to rank the parameters according to their influence on MPSP because each of them has a different unit. Two typical approaches to fixing this problem are (1) to run a regression based on standardized variables, where each parameter value is transformed into a corresponding z-score (i.e., the number of standard deviations an observation is away from the population mean), and

Table 2.3: Results of the Monte Carlo simulations (N = 10,000) for MPSP (euros/kg)

Number of plants	Min	1st Qu.	Median	Mean	3rd Qu.	Max	Standard deviation
7	7.08	9.13	9.73	9.81	10.41	13.53	0.94
14	10.47	13.81	14.92	15.08	16.18	22.28	1.74
21	13.52	18.01	19.57	19.8	21.36	30.13	2.46
28	16.21	21.72	23.68	23.97	25.92	37.06	3.1
35	18.56	24.93	27.25	27.59	29.88	43.07	3.65
42	20.56	27.67	30.27	30.65	33.24	48.17	4.11
49	22.21	29.93	32.76	33.17	35.99	52.35	4.48
56	23.51	31.68	34.7	35.13	38.12	55.63	4.76
63	24.46	32.95	36.09	36.55	39.64	57.98	4.96
70	25.03	33.74	36.93	37.41	40.57	59.42	5.07
77	25.24	34.04	37.23	37.72	40.88	59.95	5.09
84	25.11	33.85	36.99	37.48	40.61	59.56	5.02
91	24.63	33.17	36.21	36.68	39.73	58.26	4.88
98	23.81	31.98	34.89	35.34	38.24	56.05	4.64
105	22.64	30.31	32.99	33.45	36.14	52.92	4.33
112	21.12	28.13	30.58	31	33.48	48.87	3.94
119	19.26	25.45	27.61	28	30.2	43.91	3.49
126	17.05	22.26	24.12	24.45	26.34	38.04	3
133	14.49	18.51	20.08	20.35	21.88	31.25	2.49
140	10.99	14.17	15.46	15.7	16.96	24.67	2.07

(2) to transform the dependent and independent variables into log values, in which case the estimated parameters are interpreted as elasticities. Unfortunately, the applicability of both these approaches is limited in our context. The z-score approach is problematic because the z-score of n^2 does not equal the square of the z-score of n. This renders the interpretation of the marginal effect of n on MPSP meaningless. With regard to the second approach, converting all variables into log values creates the problem of perfect collinearity between ln$n$ and ln n^2 because ln $n^2 = 2\ln n$. To overcome these issues, we ran a mixed model in which the values of all variables are converted into log values, except for the number of plants and its square (the last two columns in Table 4). The estimates of the coefficients corresponding to the log values of the variables represent elasticities, that is, the percentage change in MPSP for a one-percent change in an exogenous parameter. The absolute value of these elasticities, thus, determines the impact of a factor on MPSP.

To ensure that the change in the number of production plants is comparable to other parameters, we need to determine the implicit elasticity of MPSP with respect to n. Everything else held constant, the estimated relationship between MPSP and n in the transformed model in Table 4 is

$$\ln MPSP = \alpha_1 n + \alpha_2 n^2 \tag{8}$$

where α_1 and α_2 are the estimated coefficients. Strictly speaking, because n is an integer, the derivative of MPSP with respect to n cannot be defined. However, to be able to proceed with our calculations, we assume that the plants are perfectly divisible (i.e., their number can be represented by a positive real number). Then, by totally differentiating equation (8) and rearranging the terms, we obtain

$$\frac{dMPDP}{dn} = (\alpha_1 d + 2\alpha_2 n)\, MPSP \tag{9}$$

The elasticity of MPSP with respect to n can then be defined as follows.

$$\varepsilon = \frac{dMPSP}{dn}\frac{n}{MPSP} \tag{10}$$

By inputting (9) into (10), we obtain the explicit elasticity formula

$$\varepsilon = (\alpha_1 + 2\alpha_1 n)\, n \tag{11}$$

Finally, applying the estimated coefficients, the elasticity formula becomes $\varepsilon = (0.0328 - 0.000384n)\, n$. A close inspection of this formula reveals that when the number of production plants is low, the elasticity is initially positive but entails an unambiguous decrease in the number of plants. After a certain break-even point, it will turn negative. This is not surprising, as a concave relationship was found between the number of plants and MPSP (Figure 4).

The break-even point occurs when $0.0328 - 0.000384n = 0$, that is, for n $\approx$ 85. Based on the estimated coefficients of the untransformed model, MPSP reaches its maximum when $0.947 - -2x0.0056n = 0$, that is, when n $\approx$ 84. This is consistent with the previous result. It should be noted that this number is a bit higher than that shown in Figure 4, where the (average) maximum value of MPSP was reached at n=77. This is because the coefficients presented in Table 2.4 are estimated based on a greater variation in the value of n (any integer between 1 and 140) than in Figure 4 (where the n values are in multiples of 7), and there are more random interactions with other parameters in Table 4 (the same set of 10,000 parameter combinations was used for each plant size in Figure 4, while all model parameters were randomly drawn for each model run in Table 4).

Finally, we applied Eq. 11 to each of the 100,000 model runs to obtain the desired elasticities. Their values range between -2.94 and 0.70, with the mode equal to 0.54. The mode of the negative values is -0.65, while that of the positive values is 0.54. When the absolute values of the modes of the elasticities of MPSM with respect to the number of plants are compared with other elasticities estimated for the remaining parameters in the "transformed" column in Table 4, the number of production plants is found to be the third most influential factor for the MPSP, after the availability of potato peels and transportation cost.

Table 2.4: Relative impact of exogenous parameters on the MPSP

Variables	Untransformed regression		’Transformed regression	
	Estimate	p-value	Estimate	p-value
Potato peel price	13.3	0.048	0	0.792
NaOH price	0.143*	0.196	-0.001	0.439
H2SO4 price	0.6	0.585	-0.001	0.705
NH4OH price	0.035	0.692	0.003	0.106
Amylase price	0.054 ***	0	0.017 ***	0
MgSO4 price	0.099	0.5	-0.003 *	0.042
NH4SO4 price	-0.208 *	0.018	0.001	0.454
C6H8O7 price	0.01	0.78	-0.003	0.105
K2SO4 price	-0.046	0.434	0.003	0.115
PLA price	0.853 ***	0	0.063 ***	0
OLA price	0.013	0.504	0.003	0.066
Electricity price	7.51	0.089	0.037	0.077
Gas price	-1.77	0.688	0.002	0.683
Labor cost	0.042 ***	0	0.053 ***	0
Transport cost (tkm)	136.000 ***	0	0.530 ***	0
Potato peel availability (Q)	0.000 ***	0	-0.854 ***	0
Number of plants (n)	0.947 ***	0	0.033 ***	0
Square value of the number of plants (n2)	-0.006 ***	0	0.000 ***	0
Share of the investment from a loan ($alfa$)	-22.000 ***	0	-0.306 ***	0
Annual interest rate on the loan (rho)	-0.202	0.819	-0.002	0.371
Nominal discount rate (i)	0	0.165	-0.002	0.161
Inflation (m)	3.37	0.431	-0.003	0.298
Salvage value fraction ($gamma$)	1.56	0.075	-0.001	0.732
Maintenance cost fraction ($beta$)	-8.24	0.351	0.002	0.326
Annual growth rate of the maintenance cost (g)	-2.23	0.8	-0.003 *	0.046
Payback period (tau)	0.002	0.649	0.001	0.094
Intercept	23	0	15.600 ***	0
Adjusted R2	0.975	-	0.968	-

Number of observations = 100,000
’All variables in the log-log model are log transformed except for n and n2.

2.4 Discussion

This research evaluated ex ante the economic feasibility of a novel bio-based packaging film (PLA-PHB blend). On reviewing the literature, we found that most studies have evaluated the economic performance of a bioeconomy investment for a given production scale and a long payback period. Instead, the present study was designed to determine the MPSP of a PLA-PHB blend at the European level for a reasonable payback period that is acceptable to investors.

Considering the amount of potato peels produced in Europe, the potential biofilm production capacity for PLA/PHB packaging is 126 kt annually. Referring to the 2019 data, this amount could potentially replace 0.22% of total European fossil-based plastic, increase European bioplastic production by 24.8% and global flexible packaging bioplastic production by 22.7%. Thus, we can consider this biofilm as an attractive alternative to the development of bioplastics and, due to its biodegradability and compostability, a potential solution to reducing landfilling of food waste and packaging and greenhouse gas emissions. The conditions for concrete market development, however, remain complicated.

The results showed that there was a significant variation in the MPSP of bioplastics as the number of homogeneous plants operating in Europe changed. The MPSP peaked at 37.2 euros per kg (at the baseline), with 77 plants producing 1.6 kt of bioplastic a year. As the number of plants with the smallest scale increased (i.e., 140 plants with an annual production capacity of 0.9 kt), the price decreased to 15.4 euros per kg. However, in the scenario of seven large-scale plants (with an annual production capacity of 18 kt), the MPSP reached a minimum value of 9.7 euros per kg. These trends were primarily driven by the combined effect of fixed and transportation costs, which were highly dependent on the amount of processed biomass.

As mentioned in the introduction, the production of bioplastics is typically hampered by high capital and production costs. In this specific case, as well, the low conversion coefficient due to the use of by-products/wastes (with low carbohydrate content) leads to high upfront investments. As a result, the technology is unlikely to attract investors in the current packaging market. In fact, even if the payback time was longer, the estimated prices were far from being competitive with those of fossil-based plastics (such as PET, PP, and PS), which cost 1–1.5 euros per kg (Halonen et al. 2020). It should be noted, however, that the technology studied is still in its experimental stage, and material flows were scaled up linearly from a pilot plant. Thus, further improvements are possible for the estimates implemented in this study.

According to the results of the uncertainty analysis, the availability of biobased feedstock is the most influential factor on the profitability of the new biotechnology. In this regard, the designed biotechnology can be improved and extended to other starch-rich foods (such as corn, barley or cassava), since the processes for extracting starch from their residues are similar (Dziedzic and Kearsley, 1995). In fact, all these recovery processes rely on wet milling followed by separation using a centrifuge or cyclone, given the insolubility of starch in water (Sánchez et al., 2017). Cellulose-rich residues (such as rice by-products or wheat straw) are another alternative. Cellulose can be recovered through a fractionation process (Bassani et al., 2020) and hydrolyzed to produce glucose (Gupta and Verma, 2015), which can then potentially be processed to obtain PHB. With the ability to use different residues, the feedstock availability increases and with it the profitability of biotechnology, as demonstrated in this study. In addition, the biorefinery would be less affected by the seasonality of the by-product. By-product production may not be uniformly distributed throughout the year, forcing the plant to collect and store large amounts of material for stable production. Biotechnology based on different feedstock also characterized by different seasonality is an important technological development to make biofilm competitive, due to the resulting reduction in operating costs and selling price.

The present study confirms the trade-off between reducing transportation costs and economizing on fixed costs as a core element for a biorefinery's production organization. Establishing fewer plants for the production of this bioplastic film pays off only when large plant scales are achieved, in the order of 9–18 kt per year. Alternatively, minimizing transport distances by developing many small plants is more cost-efficient. Another important finding in our model is the trend in the average transport costs with the variation in the number of plants. That is, minimizing transport distances assumes that small-scale plants are present close to a single potato processor. Although numerous, there will be no competition among plants for resources given their low productivity. Similarly, there will be no competition if there is only one large plant collecting all the resources within an area, thus creating mono/oligopoly regimes. In the case of intermediate plant scales, on the contrary, competition for resources increases, and this will affect the cost of biomass acquisition and increase collection distances. Considering the results of this study, it is unlikely that such an intermediate scenario would achieve better economic performance than other plant sizes. However, this should be better investigated by future research that focuses on the biorefinery's decision process for optimal plant size and location and allows for heterogeneous conditions. Moreover, further work is required to promote circular bio-based solutions for the systematic, international monitoring of waste streams.

2.4.1 Barriers and suggestions for future reseach

Along with paying attention to the availability (and seasonality) of waste, capital and transportation costs, as discussed above, future research should be undertaken to investigate and overcome other relevant obstacles to the adoption of bio-based plastic films.

The first aspect concerns the mechanical and gas barrier properties of biofilms for food packaging. In this respect, fossil-based films perform significantly better than biofilms. Examples of solutions to these limitations include adding cellulose to the blend to improve mechanical and barrier properties or natural extracts to improve antioxidant and antimicrobial properties. Further work is needed to determine the feasibility of these solutions concerning increased production costs and potential reduction in film transparency and food compatibility.

Another obstacle concerns legislation and its ambiguity. In 2021, the European Commission published the guidelines on the Single-Use Plastic Directive (EU) 2019/904. According to these guidelines, biopolymers such as PLA and PHB cannot be used for single-use food packaging in Europe. The directive, however, points out that packaging for ready-to-eat food (e.g., pastry, single-dose fruit) is considered single-use plastics (including bottles below 3 liters), whereas if the food is not ready-to-eat, the packaging is not considered for single-use. The PLA-PHB blend in this study, for instance, was tested to package mushrooms and vegetables for cooking and thus falls outside the scope of the directive. The ambiguity of the legislation regulating plastic and food packaging production generates high uncertainties in the market, reducing investments in the short run. Future research should take these issues into account by evaluating biotechnological development as the type of food and the conditions of use of the biopolymer change.

Finally, consumers also play an essential role in the development of biotechnology. Research shows that consumers accept higher prices for packaging perceived as sustainable. Quantitative assessment of this willingness to pay should accompany future bioplastics developments to establish a benchmark selling price. This, however, requires experts, companies, and governments to agree and share standards (based on consistent criteria, metrics, and methods) that ensure a correct and effective sustainability assessment.

2.5 Conclusions

This work presented the economic evaluation of a biorefinery concept encompassing a PHB-PLA blend production from agro-industrial waste. The baseline scenario showed that the threshold price of the studied bio-based packaging film ranges from 9.7 to 37.2 euros per kg and can lead to the investment break-even point in four years. Given the lack of resource competition, both larger and smaller plants performed better economically than intermediate-sized plants. However, considering the current market price for fossil-based plastics of about 1–1.5 euros per kg, the current production state is unlikely to be competitive or attractive for industries. Given the urgent need to redesign plastics for more sustainable production processes, further research is needed, particularly for increasing conversion efficiency at low fixed costs, such that the strategy of recycling agro-industrial waste as a substrate is economically feasible. The early-stage cost estimation method we used here can be applied to other emerging bio-based productions in the future.

Our results are not without limitations. The first is the assumption that the total mass of waste is processed by a diverse number of biorefineries. In addition to not accounting for the heterogeneity of plants, this assumption implies that all produced waste is processed. The second limitation relates to the lack of information due to the low TRL of the studied technology. In terms of cost items, transportation distances represent the most uncertain element. We used the method of Cristóbal et al. (2018), as they are the only researchers to develop a logistic correlation for potato peel-based biorefineries so far. To resolve the uncertainty, we characterized the uncertainty surrounding our estimations using Monte Carlo simulations. Finally, an additional uncontrolled factor is the environmental impact of production according to industrial scale. Further research that takes environmental externalities and transportation distances into account is needed for a more complete sustainability assessment of PLA-PHB blends.

Acknowledgments

This research was financially supported by the Bio-Based Industries Joint Undertaking under the European Union's Horizon 2020 Research and Innovation programme under grant agreement No 792261 (NewPack project).

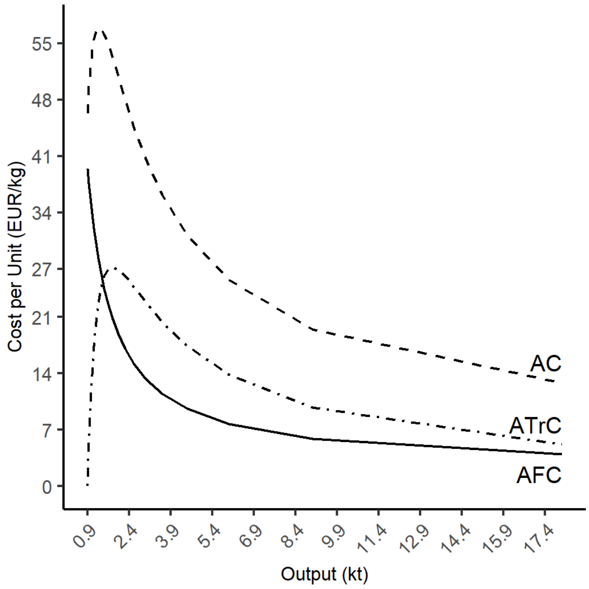
Supplementary information

Table S1: Prices used to calculate the raw material cost

Item	Baseline	Min	Max	Unit	Source
Potato peels	2.5	0	4	Euros/tons	Personal communication with company
NaOH	0.4	-	-	Euros/kg	(Fernández-Rodríguez et al., 2021)
H2SO4	0.04	-	-	Euros/kg	(Fernández-Rodríguez et al., 2021)
NH4OH	0.5	-	-	Euros/kg	(Shahzad et al., 2017)
Gluco-amylase	8.9	5.6	13.3	Euros*/kg	(Manandhar and Shah, 2020)
Alpha-amylase	8.9	5.6	13.3	Euros*/kg	(Manandhar and Shah, 2020)
MgSO4	0.3	-	-	Euros*/kg	(Sano Coelho, 2017)
(NH4)2SO4	0.5	-	-	Euros*/kg	(Diep et al., 2012)
Citric acid	1.2	-	-	Euros*/kg	(Bondancia et al., 2020)
KH2SO4	0.75	-	-	Euros/kg	EUROSTAT
PLA	2.3	1.7	3	Euros*/kg	(Levett et al., 2016)
OLA	2.2	-	-	Euros*/kg	(Kwan et al., 2015)

*indicates US dollars converted at an exchange rate of 0.89 euros/dollar; In cases where the domain was unknown (-), we reduced or increased the baseline value by 30%.

Figure S1: Cost estimations for each unit: Average fixed cost (AFC), average transportation cost (ATrC), and average cost (AC) for NEWPACK production



References

Abelha, P. and Kiel, J. 2020. “Techno-economic assessment of biomass upgrading by washing and torrefaction,” Biomass Bioenergy 142: 105751. doi: 10.1016/j.biombioe.2020.105751.

Arrieta, M. P. et al. 2015. “Bionanocomposite films based on plasticized PLA-PHB/cellulose nanocrystal blends,” Carbohydrate Polymers 121: 265–275. doi: 10.1016/j.carbpol.2014.12.056.

Asgher, M. et al. 2020. “Bio-based active food packaging materials: Sustainable alternative to conventional petrochemical-based packaging materials,” Food Research International 137(June). doi: 10.1016/j.foodres.2020.109625.

de Assis, C. A. et al. 2017. “Conversion Economics of Forest Biomaterials: Risk and Financial Analysis of CNC Manufacturing,” Biofuels, Bioproducts and Biorefining 11(4): 682–700. doi: 10.1002/bbb.1782.

Bassani, A., Fiorentini, C., Vadivel, V., Moncalvo, A., Spigno, G. (2020). Implementation of auto-hydrolysis process for the recovery of antioxidants and cellulose from wheat straw. Applied Sciences (Switzerland), 10(17). https://doi.org/10.3390/app10176112Battegazzore, D., Alongi, J. and Frache, A. 2014. “Poly(lactic acid)-Based Composites Containing Natural Fillers: Thermal, Mechanical and Barrier Properties,” Journal of Polymers and the Environment 22(1): 88–98. doi: 10.1007/s10924-013-0616-9.

BBI JU (2022) Development of new Competitive and Sustainable Bio-Based Plastics. Available at: http://www.bbeu.org/pilotplant/newpack/ (accessed May, 2022)

Birch, P. R. J., Bryan, G., Fenton, B., Gilroy, E. M., Hein, I., Jones, J. T., Prashar, A., Taylor, M. A., Torrance, L., Toth, I. K. (2012). Crops that feed the world 8: Potato: Are the trends of increased global production sustainable? In Food Security (Vol. 4, Issue 4). https://doi.org/10.1007/s12571-012-0220-1

Bondancia, T. J. et al. 2020. "Production of Nanocellulose Using Citric Acid in a Biorefinery Concept: Effect of the Hydrolysis Reaction Time and Techno-Economic Analysis," Industrial and Engineering Chemistry Research 59(25): 11505–11516. doi: 10.1021/acs.iecr.0c01359.

Campos, H., Ortiz, O. (2019). The potato crop: Its agricultural, nutritional and social contribution to humankind. In The Potato Crop: Its Agricultural, Nutritional and Social Contribution to Humankind. https://doi.org/10.1007/978-3-030-28683-5

Carolis, C. De (2021) 'Life Cycle Assessment', in Biobased Products from Food Sector Waste. Springer International Publishing, pp. 365–377.

CIEL. 2019. Plastic Health: the hidden costs of a plastic planet; Center for International Environmental Law Technical Report. Available at https://www.ciel.org/wp-content/uploads/2019/05/Plastic-and-Climate-FINAL-2019.pdf. (accessed September, 2021)

CORDIS. 2020. Public final report. Available at https://cordis.europa.eu/project/id/792261/results (accessed August, 2022)

Coelho, R. 2017. "Biodiesel production from heterotrophic microalgae". https://doi.org/10.4233/uuid:adc2054d-a2cd-451e-b2f6-1e2f87b3409b

Cristóbal, J. et al. 2018. "Techno-economic and profitability analysis of food waste biorefineries at European level," Bioresource Technology 259(January): 244–252. doi: 10.1016/j.biortech.2018.03.016.

Dávila, J. A., Rosenberg, M. and Cardona, C. A. 2016. "A biorefinery approach for the production of xylitol, ethanol and polyhydroxybutyrate from brewer's spent grain," AIMS Agriculture and Food 1(1): 52–66. doi: 10.3934/agrfood.2016.1.52.

Diep, N. Q. et al. 2012. "Comparison of the potential for ethanol production from rice straw in Vietnam and Japan via techno-economic evaluation," International Energy Journal 13(3): 113–122.

Dilkes-Hoffman, L. S. et al. (2018) 'Environmental impact of biodegradable food packaging when considering food waste', Journal of Cleaner Production. Elsevier Ltd, 180, pp. 325–334. doi: 10.1016/j.jclepro.2018.01.169.

Dornburg, V. et al. (2006) 'Economics and GHG emission reduction of a PLA bio-refinery system - Combining bottom-up analysis with price elasticity effects', Resources, Conservation and Recycling, 46(4), pp. 377–409. doi: 10.1016/j.resconrec.2005.08.006.

Dziedzic, S. Z., Kearsley, M. W. (1995). The technology of starch production. In Handbook of starch hydrolysis products and their derivatives (pp. 1–25). Springer.Ellen MacArthur

Foundation. 2016. The New Plastics Economy: Rethinking the Future of Plastics. (accessed September, 2021)

European Bioplastics. 2020. Bioplastics market development update 2019.(accessed September, 2021)

European Commission. 2014. Guide to Cost-Benefit Analysis of Investment Projects. (accessed September, 2021)

Eurostat. 2021a. Electricity price statistics. (accessed September, 2021)

Eurostat. 2021b. Hourly labor cost in 2020. (accessed September, 2021)

Eurostat. 2021c. Natural gas prices for non-household consumers. (accessed September, 2021)

Eurostat. 2021d. Crop production in EU standard humidity. (accessed September, 2021)

Exchange Rate. 2021. British Pound to Euro Spot Exchange Rates for 2019. (accessed September, 2021)

Fernández-Rodríguez, J. et al. 2021. "Techno-economic analysis of different integrated biorefinery scenarios using lignocellulosic waste streams as source for phenolic alcohols production," Journal of Cleaner Production 285. doi: 10.1016/j.jclepro.2020.124829.

Filho, W. L. et al. 2021. "COVID-19 and waste production in households: A trend analysis," Science of the Total Environment 777. https://doi.org/10.1016/j.scitotenv.2021.145997

Gerrard. 2000. Guide to capital cost estimating. IChemE, United Kingdom

Golberg, A., Polikovsky, M., Epstein, M., Slegers, P. M., Drabik, D., and Kribus, A. 2021. "Hybrid solar-seaweed biorefinery for co-production of biochemicals, biofuels, electricity, and water: Thermodynamics, life cycle assessment, and cost-benefit analysis," Energy Conversion and Management 246: 114679. https://doi.org/10.1016/j.enconman.2021.114679

Gupta, A., Verma, J. P. (2015). Sustainable bio-ethanol production from agro-residues: A review. Renewable and Sustainable Energy Reviews, 41, 550–567. https://doi.org/10.1016/j.rser.2014.08.032Halonen, N. et al. 2020. "Bio-based smart materials for food packaging and sensors–a review," Frontiers in Materials 7: 82.

Hatti-Kaul, R., J. Nilsson, L., Zhang, B., Rehnberg, N., and Lundmark, S. 2020. "Designing Biobased Recyclable Polymers for Plastics," Trends in Biotechnology 38(1): 50–67.

Heck, S., Rogers, M., and Carroll, P. 2014. Resource revolution: how to capture the biggest business opportunity in a century. McKinsey Company.

Hickenbottom, K. L. et al. 2017. "Techno-economic assessment of a closed-loop osmotic heat engine," Journal of Membrane Science 535(February): 178–187. doi: 10.1016/j.memsci.2017.04.034.

Ioannidou, S. M. et al. (2022) 'Techno-economic risk assessment, life cycle analysis and life cycle costing for poly(butylene succinate) and poly(lactic acid) production using renewable resources', Science of the Total Environment, 806. doi: 10.1016/j.scitotenv.2021.150594.

Jenkins, Scott. 2020. Chemical Engineering Plant Cost Index. Available at https://www.chemengonline.com/2019-chemical-engineering-plant-cost-index-annual-average/. (accessed September, 2021)

Kakadellis, S. and Harris, Z. M. (2020) 'Don't scrap the waste: The need for broader system boundaries in bioplastic food packaging life-cycle assessment – A critical review', Journal of Cleaner Production. Elsevier Ltd, 274, p. 122831. doi: 10.1016/j.jclepro.2020.122831.

Kim, H. et al. 2020. "Sustainable Production of Bioplastics from Lignocellulosic Biomass: Technoeconomic Analysis and Life-Cycle Assessment," ACS Sustainable Chemistry and Engineering 8(33): 12419–12429. doi: 10.1021/acssuschemeng.0c02872.

Knoema. 2021. Euro Area Inflation Forecast 2019-2024 and up to 2060. Available at https://knoema.com/zobdrl/euro-area-inflation-forecast-2019-2024-and-up-to-2060-data-and-charts. (accessed September, 2021)

Koller, M. et al. 2013. "Polyhydroxyalkanoates: Biodegradable polymers and plastics from renewable resources," Materials and Technology. 47(1): 5–12.

Kwan, T. H. et al. 2015. "Techno-economic analysis of a food waste valorization process via microalgae cultivation and co-production of plasticizer, lactic acid and animal feed from algal biomass and food waste," Bioresource Technology 198(2015): 292–299. doi: 10.1016/j.biortech.2015.09.003.

Levett, I. et al. 2016. "Techno-economic assessment of poly-3-hydroxybutyrate (PHB) production from methane - The case for thermophilic bioprocessing," Journal of Environmental Chemical Engineering 4(4): 3724–3733. doi: 10.1016/j.jece.2016.07.033.

Lopez-Arenas, T. et al. 2017. "Analysis of the fermentation strategy and its impact on the economics of the production process of PHB (polyhydroxybutyrate)," Computers and Chemical Engineering, 107, 140–150. doi: 10.1016/j.compchemeng.2017.03.009.

Manandhar, A. and Shah, A. 2020. "Techno-Economic Analysis of Bio-Based Lactic Acid," Processes 199: 8.

Marra, A. et al. 2016. "Polylactic acid/zinc oxide biocomposite films for food packaging application," International Journal of Biological Macromolecules 88: 254–262. doi: 10.1016/j.ijbiomac.2016.03.039.

Moncada, J., El-Halwagi, M. M. and Cardona, C. A. 2013. "Techno-economic analysis for a sugarcane biorefinery: Colombian case," Bioresource Technology 135: 533–543. doi: 10.1016/j.biortech.2012.08.137.

Mullins, E., Milbourne, D., Petti, C., Doyle-Prestwich, B. M., Meade, C. (2006). Potato in the age of biotechnology. Trends in Plant Science, 11(5), 254–260. https://doi.org/10.1016/j.tplants.2006.03.002

Nampoothiri, K. M., Nair, N. R., and John, R. P. 2010. "An overview of the recent developments in polylactide (PLA) research," Bioresource Technology 101(22): 8493–8501.

Naranjo, J. M. et al. 2013. “Valorization of glycerol through the production of biopolymers: The PHB case using Bacillus megaterium,” Bioresource Technology 133: 38–44. doi: 10.1016/j.biortech.2013.01.129.

Naranjo, J. M., Cardona, C. A. and Higuita, J. C. 2014. “Use of residual banana for polyhydroxybutyrate (PHB) production: Case of study in an integrated biorefinery,” Waste Management 34(12): 2634–2640. doi: 10.1016/j.wasman.2014.09.007.

Patrício Silva, A. L. et al. 2020. “Rethinking and optimising plastic waste management under COVID-19 pandemic: Policy solutions based on redesign and reduction of single-use plastics and personal protective equipment,” Science of the Total Environment 742: 140565. https://doi.org/10.1016/j.scitotenv.2020.140565

Patrício Silva, A. L. et al. 2020 “Increased plastic pollution due to COVID-19 pandemic: Challenges and recommendations,” Chemical Engineering Journal 405: 126683. https://doi.org/10.1016/j.cej.2020.126683

Peters, L. 2016. “Impact of Probability Distributions on Real Options Valuation,” Journal of Infrastructure Systems 22(3): 04016005. https://doi.org/10.1061/(asce)is.1943-555x.0000289

PlasticEurope. 2020. Plastics–the Facts 2020. Available at https://www.plasticseurope.org/en/resources/market-data. (accessed September, 2021)

Posada, J. A. et al. 2011. “Design and analysis of poly-3-hydroxybutyrate production processes from crude glycerol,” Process Biochemistry 46(1): 310–317. Doi: 10.1016/j.procbio.2010.09.003.

Posen, I. D. et al. (2017) ‘Greenhouse gas mitigation for U.S. plastics production: Energy first, feedstocks later’, Environmental Research Letters, 12(3). doi: 10.1088/1748-9326/aa60a7.

Prata, J.C., Patrício Silva, A.L., Walker, T.R., Duarte, A.C., and Rocha Santos, T. 2020. “COVID-19 pandemic repercussions on the use and management of plastics,” Environmental Science Technology 54(13): 7760-7765. doi.org/10.1021/acs.est.0c02178.

Sánchez, A. S., Silva, Y. L., Kalid, R. A., Cohim, E., Torres, E. A. (2017). Waste bio-refineries for the cassava starch industry: New trends and review of alternatives. Renewable and Sustainable Energy Reviews, 73(February), 1265–1275. https://doi.org/10.1016/j.rser.2017.02.007Saravanan, S., Leena, R. S. and Selvamurugan, N. 2016. “Chitosan based biocomposite scaffolds for bone tissue engineering,” International Journal of Biological Macromolecules 93: 1354–1365. doi: 10.1016/j.ijbiomac.2016.01.112.

Schieber, A., Saldaña, M. (2009). Potato Peels: A Source of Nutritionally and Pharmacologically Interesting Compounds – A Review. Food, 3, 23–29.

Shahzad, K. et al. 2017. “Techno-economic feasibility of waste biorefinery: Using slaughtering waste streams as starting material for biopolyester production,” Waste Management 67:

73–85. doi: 10.1016/j.wasman.2017.05.047.

Siracusa, V. et al. (2008) 'Biodegradable polymers for food packaging: a review', Trends in Food Science and Technology. Elsevier Ltd, 19(12), pp. 634–643. doi: 10.1016/j.tifs.2008.07.003.

Tsagkari, M. et al. 2016. "Early-Stage Capital Cost Estimation of Biorefinery Processes: A Comparative Study of Heuristic Techniques," ChemSusChem 9(17): 2284–2297. doi: 10.1002/cssc.201600309.

Vanhatalo, K. M., Parviainen, K. E. and Dahl, O. P. 2014. "Techno-Economic Analysis of Simplified Microcrystalline Cellulose Process," BioResources 9(3): 4741–4755. doi: 10.15376/biores.9.3.4741-4755.

Vidal, R. et al. (2007) 'Environmental assessment of biodegradable multilayer film derived from carbohydrate polymers', Journal of Polymers and the Environment, 15(3), pp. 159–168. doi: 10.1007/s10924-007-0056-5.

Wickham H. 2016. "ggplot2: Elegant Graphics for Data Analysis." New York: Springer-Verlag. ISBN 978-3-319-24277-4, https://ggplot2.tidyverse.org.

Zang, G., Shah, A. and Wan, C. 2020. "Techno-economic analysis of an integrated biorefinery strategy based on one-pot biomass fractionation and furfural production," Journal of Cleaner Production 260: 120837. doi: 10.1016/j.jclepro.2020.120837.

Chapter 3

Recycling or Landfilling? Assessing Sustainability of Nutrient Recovery

Abstract

Using a hybrid input-output approach, we systematically traced sustainability footprints of a nutrient recovery strategy from sewage sludge. The results obtained were then compared with the most common landfilling practice. Overall, accounting for infinite upstream spillover effects, using sewage sludge for organic fertiliser production generates more jobs and reduces more greenhouse gas emissions than landfilling does. By contrast, landfilling stimulates the economy more and reduces energy carrier use more. Nevertheless, the environmental benefit is certain at a high-level of confidence, as estimated by a Monte Carlo simulation. Hence, this nutrient recovery strategy can stimulate more sustainable and resilient agricultural processes.

Keywords: life cycle; wastewater; circular bioeconomy; bio-based system; supply-chain shock

This chapter is based on: Tassinari, G., Boccaletti, S., Soregaroli, C. (2023). Recycling or Landfilling? Assessing Sustainability of Nutrient Recovery; Revise and Resubmit requested by the European Review of Agricultural Economics.

3.1 Introduction

Global population growth places pressure on food supply systems, which need to cope with the higher food demand and increased consumption (Godfray et al., 2010). This situation has led to agricultural expansion and intensification (drivers of biodiversity loss), land degradation, a decline in organic matter, environmental pollution and greenhouse gas (GHG) emissions (Tamburini et al., 2020). Such developments are associated with the intensive and excessive use of agro-chemicals and fertilisers, deep tillage and luxurious irrigation (Diacono and Montemurro, 2010). The European Commission (EC) is thus committed to developing sustainable and resilient food production systems (Sperling et al., 2022) through ambitious growth action plans, such as the Green Deal (EC, 2019).

The link between a prosperous society, a competitive economy and a healthy planet places sustainable agro-food systems at the heart of the European Green Deal (EC, 2020). Agro-food systems play a crucial role in ensuring food security and achieving key targets by 2030, such as reducing GHG emissions by 55%, pesticide use by 50% and fertiliser use by 20% compared with 1990 figures (EC, 2020; Montanarella and Panagos, 2021). However, supply chain shocks and shortages, such as those arising from the COVID-19 pandemic, climate-induced extreme events[1] and the Ukraine–Russia conflict, have brought to the fore the systemic structural weaknesses in agricultural systems, thus weakening the ability to achieve these sustainability targets (Behnassi and Haiba, 2022; Sperling et al., 2022).

Disruptions in the trade of raw materials and primary products induce greater volatility in agri-food commodities and fertiliser prices, which have skyrocketed (Behnassi and Haiba, 2022). In the fertiliser market, prices already doubled between the summer of 2020 and the end of 2021 (Smith, 2022) because of (i) the rising price of natural gas (which accounts for 80% of the operating costs in nitrogen fertiliser production), (ii) market distortions because of the pandemic (e.g. China suspended fertiliser exports until the end of June 2022 to ensure domestic availability) and (iii) the increase in agricultural commodity prices, which incentivised higher fertiliser use. The war in Ukraine has exacerbated this trend, causing fertiliser prices to increase by 3% to 43% (depending on the fertiliser type) in March 2022 compared with February 2022 prices (EC, 2022) and impacting the European Union (EU)[2], in particular. Although the current increase in fertiliser prices has been mitigated to some extent by the current agronomic season[3], there are major concerns

[1] In the US, for example, Hurricane Ida created disruptions in the fertiliser industry, causing several plants to temporarily shut down, increasing barge shipping costs and exacerbating fertiliser price increases (Beghin and Nogueira, 2020).

[2] Based on Eurostat data, Russia is the main European trading partner for fertilisers (Fertilizers Europe, 2022). In 2020, the EU imported 1,120 million euros worth of fertilisers that, together with the 372 million euros of fertilisers imported from Belarus, represents 45% of the total value of fertiliser import.

[3] Immediate fertiliser needs have already been met, and current trade takes place mainly at the local

about the risk of fertiliser unavailability in the coming period. Changes in fertiliser prices are also more volatile than changes in the prices of other agricultural inputs because of demand rigidity, which has been enforced because fertilisers are essential to production and have few substitutes (Beckman and Riche, 2015).

While there are handy substitutes for some agricultural production inputs (e.g. capital for labour), chemical fertilisers tend to have few alternatives. One potential solution, along with technological developments and smart farming (Moysiadis et al., 2021), is the application of organic fertilisers, such as manure and treated sewage sludge[4] (Kumar Bhatt et al., 2019). To address the economic effects of the Ukraine crisis, for example, Italy enacted an urgent directive (Decree-Law No 21/2022) to allow for the replacement of chemical fertilisers with sludge-based organic fertilisers.

Sewage sludge is the by-product of municipal or industrial wastewater treatment plants (WWTPs). Because of population growth, urban planning and industrial development, the volume of sewage sludge produced has rapidly increased (Kumar et al., 2017). In addition, the implementation of Directive 91/271/EC (Council of the European Communities, 1991) to improve wastewater collection and treatment is also causing a significant increase in annual sewage sludge production in the EU (Kelessidis and Stasinakis, 2012). A common end use of sewage sludge is landfilling, especially for some (new) EU Member States (e.g. Malta, Croatia and Romania), where 67 to 100% of sewage sludge is landfilled (Hudcová et al., 2019; Kelessidis and Stasinakis, 2012). However, if properly treated and processed, sewage sludge can be recycled as a resource for organic fertiliser production.

Rich in nutrients and organic matter, stabilised sewage sludge can improve soil fertility, texture and chemical–physical properties. Zaman et al. (2004) and Diacono and Montemurro (2009) found that compared with treatment using chemical fertilisers alone, the long-term application of sewage sludge significantly increases the amounts of total N and soluble organic C, microbial biomass, protease, deaminase and urease activity, thus promoting nutrient mineralisation and increasing nutrient availability for future cropping seasons. Jamil et al. (2006) demonstrated a 90% increase in wheat yield—compared to unfertilised control soil—achieved by adding 40 metric tonnes of treated sewage sludge per hectare of land. Similarly, Singh and Agrawal (2010) found a 111% increase in rice yield with the application of 45 tonnes of sewage sludge per hectare. All cited studies have stressed the importance of using properly treated sludge to avoid critical levels of soil contamination (e.g. heavy metals, organic compounds and pathogens).

level, from import ports and local retailers to farms (RoboResearch, 2022).

[4]Sewage sludge is prohibited in organic farming because it is not explicitly mentioned in Annex I of EU 889/2008 (Løes and Adler, 2019), which regulates the allowed fertilisers, soil conditioners and nutrients. However, initiatives are being promoted to amend EU 889/2008 and permit the use of sewage sludge on certified organic land, as improvements in the quality of recyclable sewage sludge have been recognised (Løes et al., 2017).

Assessing the sustainability of sludge management has become an even more crucial task for scientists (Kumar et al., 2017) given the global scenario and disruptions in the natural gas and fertilizer markets. Although several studies have been conducted to assess the sustainability of sludge management strategies, there is a lack of a systematic approach (Yoshida et al., 2013). In particular, the decision on the system boundary is at the expert's discretion, leading to truncation errors that limit the robustness of the applied analysis (Crawford et al., 2018; Ward et al., 2018). In the present study, we address the challenge of systematically quantifying the economic, social, and environmental impacts of the nutrient recovery strategy from sewage sludge with a multi-regional input-output (IO) model capable of covering an infinite order of contributions from upstream production processes (Wiedmann, 2008; Lenzen, 2006).

The use of IO tables to assess systems-wide impacts is not new to the agri-food supply chains literature, and their popularity increases with the increase in the level of detail with which these databases reflect the industrial interdependencies within an economy (Hughes, 2003; Bess and Ambargis, 2011; Okuyama and Santos, 2014). Agricultural economists have used IO tables to examine macroeconomic spillover effects on output and employment from a change in economic activity, such as a change in the mix of forest plantings (Eiser and Roberts, 2002), the establishment of a large-scale ethanol plant (Thomassin and Baker, 2000), or a new rural development policy (Hyytiä, 2014; Cruz et al., 2017). More recently, Wahdat and Lusk (2022) used IO tables to assess the vulnerability of food industries to upstream industries from the perspective of intermediate inputs and labor.

Our study contributes to this strand of literature in several ways. First, we contribute to the production economics literature by studying the nature of input use of a waste nutrient recovery strategy, using an IO and structural path analysis. Second, by comparing landfilling with recycling of treated sludge for agriculture purposes, we add to the bioeconomy supply chain literature. Finally, we improve the methodological framework of multi-regional IO impact assessment by combining it with a Monte Carlo simulation to address the uncertainty of model parameters. Researchers, policymakers and practitioners can benefit from our research to improve waste management in support of a more circular and sustainable strategy, to ensure greater farm resilience to supply chain disruptions and plan for better supplier diversification within an upstream industry.

3.2 Methodological framework

In any impact assessment, the determination of an appropriate system boundary affects the reliability of the results. Identifying which activities to include in the analysis is often left to the judgment of the practitioner (Rajagopal, 2017), whose arbitrary selection of a finite boundary can lead to truncation errors (Crawford, 2018). To avoid cutting out relevant interactions with the wider economy, we applied a hybrid IO impact assessment, which can handle infinite supply chain systems (Wiedmann, 2008) by combining detailed process-based data with a multiregional input–output (MRIO) database. Several researchers have successfully applied the same techniques to estimate the carbon footprint of a selected sector (Malik et al., 2014; Lenzen, 2018; Wei et al., 2021).

3.2.1 Hybrid input–output analysis

MRIO databases are spatially explicit representations that describe the economic interdependencies among sectors and global agents (e.g. households, governments, the capital sector and stocks) within and between countries. Let T be an $N \times N$monetary MRIO transaction matrix represented by

$$T = \begin{bmatrix} T_{1,1} & T_{1,2} & \cdots & T_{1,r} \\ T_{2,1} & T_{2,2} & \cdots & T_{2,r} \\ \vdots & \vdots & \ddots & \vdots \\ T_{r,1} & T_{r,1} & \cdots & T_{r,r} \end{bmatrix} \quad (1)$$

where diagonal submatrices $T_{i,i}$ list the domestic inter-industry flows, and off-diagonal submatrices $T_{i,j}$describe the international trade transactions between regions (r). Accordingly, the $N \times M$matrix of the final demand by global agents is

$$Y = \begin{bmatrix} Y_{1,1} & Y_{1,2} & \cdots & Y_{1,r} \\ Y_{2,1} & Y_{2,2} & \cdots & Y_{2,r} \\ \vdots & \vdots & \ddots & \vdots \\ Y_{r,1} & Y_{r,1} & \cdots & Y_{r,r} \end{bmatrix} \quad (2)$$

where $Y_{i,i}$represents the final demand satisfied by domestic production, and $Y_{i,j}$directs imports to the final demand. Therefore, total economic output (x), expressed in monetary units, can be summarised as the sum of the intermediate and final demands,

$$x = Te^{T} + Ye^{Y} \quad (3)$$

where e are the row summation vectors of sizes $N \times 1\left(e^{T}\right)$ and $M \times 1$ $\left(e^{Y}\right)$. Eq. 3 can be subjected to Leontief's quantity demand-driven formalism and transformed into

$$x = \left(I - T\hat{x}^{-1}\right)^{-1} Ye^{Y} \quad (4)$$

where I is an $N \times N$identity matrix, and the hat symbol identifies the diagonalisation of the total economic output. Lenzen and Rueda-Cantuche (2012) provided a clear explanation of how to derive Eq. 4 from Eq. 3 for single-region supply and use tables. The Leontief quantity model is driven by product demand. Assuming proportional relations between inputs and outputs, there is a direct effect on the total output given a final demand variation and additional indirect effects captured by the so-called Leontief inverse matrix $(1 - T\hat{x}^{-1})^{-1}$, which propagates demand-side shocks upstream (i.e. to the suppliers of affected industries).

This global MRIO framework can be used in sustainability assessments by extending the system using external (also known as satellite data) accounts (e.g. environment, employment and energy). Let Q be the $P \times N$ satellite block including Pindicators (i.e. CO2-eq. emissions and full-time equivalents [FTEs]). $q = Q\hat{x}^{-1}$ represents the on-site impacts per unit of industrial output (i.e. CO2-eq. or FTE per monetary unit). Multiplying these intensities by Leontief's inverse matrix as

$$m = Q\hat{x}^{-1}(I - T\hat{x}^{-1})^{-1} \tag{5}$$

we obtain a matrix of multipliers m, which contains the total impacts embodied in a unit of final demand rather than per unit of industrial output, thus including indirect spillover effects. However, MRIO sectors are generally aggregated and not sufficiently detailed (Lenzen, 2000; Suh et al., 2004) for the purpose of our analysis. We augment the MRIO table with new columns and rows simulating selected industries and products (Malik et al., 2015). This hybridisation procedure provides both specificity and completeness to the impact analysis; process-based data provide a detailed representation of on-site impacts, while IO analysis captures the total (direct and indirect) impacts (Leontief and Ford, 1970), eliminating truncation errors (Crawford et al., 2018; Malik et al., 2019).

3.2.2 Production layer decomposition

After the MRIO database is integrated with a new detailed sector and the IO analysis is conducted, the estimated total multiplier m can be further decomposed via production layer decomposition. Noting that the series expansion of the Leontief matrix can be written as $L = I + A + A^2 + ... + A^n$ (Lenzen and Rueda-Cantuche, 2012), the basic IO equation becomes

$$Q^* = q\#Ly^* = q\#y^* + q\#\mathrm{A}y^* + q\#\mathrm{A}^2y^* + ... + q\#\mathrm{A}^ny^* \tag{6}$$

where denotes element-wise multiplication, and y^* denotes the non-zero element final demand vector, corresponding to the modelled monetary final demand shock. The term A^ncaptures the contributions from supply chains of nth order, and the sum of all these contributions is called the nth production layer (Lenzen, 2018). Thus, the term $q\#y^*$represents the direct impacts on production, the first-order term $q\#\mathrm{A}y^*$ refers to the impacts on direct suppliers, $q\#\mathrm{A}^2y^*$ indicates the impacts on the suppliers of direct suppliers and so on. For example, $q\#y^*$represents the on-site impacts of sludge-based organic fertiliser production, $q\#\mathrm{A}y^*$ indicates the impacts on direct suppliers, including lime producers who supply lime as input for the sludge stabilisation, and $q\#\mathrm{A}^2y^*$ denotes the impacts on the suppliers of suppliers, such as the energy industry, which provides the energy needed to produce the amount of lime purchased from organic fertiliser production in order to stabilise sludge.

3.3 Empirical application

In this study, we relied on the latest version (v.8.3) of Exiobase for the year 2020 (Stadler et al., 2018). Exiobase follows a standard supply-use structure featuring 49 regions ($r = 49$) (27 EU Member States, 17 major economies and five regions in the rest of the world), 163 industries and 200 products ($N = 49 \times (163 + 200) = 17787$), and four global agents ($M = 49 \times 4 = 196$). As a tool for impact assessments, Exiobase includes several economic, social and environmental indicators (Malik et al., 2019). For this study, we used economic stimulus (estimated as the total purchased input value), employment, GHG emissions and energy carrier use ($P = 4$). In the hybridisation process, we added a new column and row to these MRIO matrices (T, Y, and Q) to simulate the production of a specific organic fertiliser from sewage sludge.

Process-based data (Table 3.1) were collected through a case study conducted in Pavia province, Northern Italy, in 2020. Pavia is the leading Italian and European producer of rice, with about 80,000 hectares cultivated. The area has intensive farming systems and a low livestock farming density. Over the years, this has created a great demand for organic fertilisers to restore soil fertility and correct pH variations as a result of mineral fertilisation and submersion for rice cultivation. Site selection was based on purposive and convenience criteria on i) the representativeness of a complete, qualified and actual biobased ecosystem, ii) accessibility of processing plant data and iii) willingness of the companies to participate in the study. Data collection was conducted at both the sectoral and product levels. Several sources of information, including semi-structured interviews, questionnaires and legal reports, were triangulated to gather evidence.

The biorefinery collects about 165,000 metric tonnes of sewage sludge from 45 WWTPs annually, serving 226 municipalities. Each WWTP holds a tender, and the company that offers the best techno-economic conditions (lowest price) is awarded the contract for sludge collection. The large availability of biomass allows many players to enter the sector. Organic fertilisers produced from sewage sludge, on the other hand, currently have no economic value, which is hampered by consumer scepticism about the safety of their use for human health and the environment. Therefore, the company charges money only for the waste collection service (on average, 100 euros per metric tonne of sewage sludge), which is counted in this framework as final consumption (Y).

By-product withdrawal is handled by land transportation. Before being collected, sludge is analysed for its chemical profile. The biomass is treated as a resource rather than waste and is conditioned by sulphuric acid, lime and gypsum to produce 160,000 tonnes

Table 3.1: Process data used for augmentation

Item	Quantity	Unit	Value	Unit	Exiobase code
Sewage sludge	165,000	MT	100	EUR/MT	Final consumption expenditure
Sulphuric acid	130	MT	40	EUR/MT	Chemicals nec
Lime	3,000	MT	63	EUR/MT	Cement, lime and plaster
Gypsum	20,000	MT	11	EUR/MT	Stone
Sewage sludge handling	165,000	MT	13	EUR/MT	Other land transportation services
Organic fertiliser spreading	165,000	MT	6	EUR/MT	Other land transportation services
Chemical analysis of sludge			393,525	EUR	Chemicals nec
Chemical analysis of organic fertilisers			65,510	EUR	Chemicals nec
Chemical analysis of land			117,760	EUR	Chemicals nec
Photovoltaic electricity	430,000	kWh	0.09	EUR/kWh	Electricity by solar photovoltaic
Gasoline	115,000	l	0.69	EUR/l	Motor gasoline
Water	6,600	m3	0.38	EUR/m3	Steam and hot water supply services
Research investment			5,600,000	EUR	Research and development services
Fixed capital amortisation			360,000	EUR	Operating surplus: Consumption of fixed capital
Compensation of employees			2,571,530	EUR	Compensation of employees; wages, salaries and employers' social contributions
Net operating surplus			3,151,917	EUR	Operating surplus: Remaining net operating surplus

of organic fertilisers and biosolids. On average, the dry matter content is 28.5%. The chemical characteristics per dry matter include 1.6% total nitrogen, 1.5% total phosphorus, 0.4% potassium and 24% organic C content. Only after an evaluation of the chemical profile conformability are the organic fertilisers distributed free of charge to local farmers. The biorefinery serves a total of 10,000 hectares, mainly for rice or corn production. In addition to the distribution of organic fertilisers, consulting services are provided for plowing and fertilisation. Next, a third chemical control is carried out on the soil. The collected data were reported in Exiobase codes identified by correspondence tables using the Harmonized System and the Statistical Classification of Products by Activity. The entire production requires a total of 10.42 million euros per year, 84 employees (FTE) and 5.48 TJ of energy carriers. No information was available regarding GHG emissions. Therefore, the CO2 emissions reported by Murray et al. (2008) for lime stabilisation (550 kg of CO2 per dry tonne of sludge at 20% dry matter content) were used as a reference.

For comparative purposes, we used the 'Landfill of waste: Food' sector of Exiobase. Although generic and aggregated, the sector is also representative of the landfilling of sewage sludge because in Italy, food waste and sewage sludge are often stored and landfilled together. Finally, as the two waste management strategies are perfect substitutes, we considered the collection and landfill service offered at the same price (100 euros per tonne of sewage sludge) as the service supplied by the biorefinery. However, we recognise that there are several sources of uncertainty in the analysis that need to be addressed.

3.3.1 Uncertainty analysis

To address part of the uncertainty surrounding our assumptions, we determine the stochastic variation of the total impacts using a Monte Carlo simulation (Lenzen et al., 2010). Uncertainty is propagated using standard deviations σ_Q, σ_T,and σ_Y for perturbating the basic data items Q, T, and Y, with the perturbated footprints $m*$ calculated from 10,000 simulations (Lenzen et al., 2018). The dispersion (or uncertainty) of estimated footprint measures is then derived from the statistical distribution of the perturbations.

More specifically, we approximated the logarithmic absolute error of $\log x$ as

$$\sigma_{\log_{10} x} \approx \log_{10}(x + \sigma_x) - \log_{10}(x) = \log_{10}\left(\frac{x + \sigma_x}{x}\right) = \log_{10}(1 + r_x) \tag{7}$$

where r_x is the relative standard deviation (RSD) of x. The log-normality assumption ensures that the Monte Carlo perturbations do not extend towards negative values that pose problems for IO analysis. Hence, the perturbed entries of the MRIO coefficient can be computed as $Q^P = 10^{\log_{10} Q + \nu\sigma_{\log_{10} Q}}$,$T^P = 10^{\log_{10} T + \nu\sigma_{\log_{10} T}}$ and $Y^P = 10^{\log_{10} Y + \nu\sigma_{\log_{10} Y}}$, where νdenotes a vector of random numbers normally distributed $\nu \in N(0|1)$. The logarithmic perturbations can be computed as Eq. 7. The perturbated $x^P$is obtained by summing $T^P$and $Y^P$to maintain the balance in the IO table (Wei et al., 2021). In this exercise, we assumed that the satellite accounts (matrix Q) exhibit an RSD of 30%, whereas the final demand (Y) and transaction matrices (T) exhibit a relatively low RSD of 10% to avoid over-perturbations of gross output (x), which is known with a relatively high degree of confidence (Lenzen et al., 2010).

3.4 Results

3.4.1 Direct and indirect impacts

Total impacts (m) include impacts directly related to industrial production (q) and indirect externalities triggered along the upstream supply chains. Table 3.2 shows the results, comparing the sustainability of the nutrient recovery strategy with landfilling for a million-euro demand shock, which corresponds to 10,000 tonnes of processed (recycled or landfilled) sewage sludge (recalling that both waste collection services cost 100 euros per tonne of sewage sludge).

Table 3.2: Triple bottom line impact assessment per 1 million-euro demand shock

Triple Bottom Line Indicators	Nutrient Recovery (x)		Landfill (y)		(x-y)	
	q (direct impacts)	m (total impacts)	q (direct impacts)	m (total impacts)	q (direct impacts)	m (total impacts)
Economic stimulus (million euros)	0.63	1.27	0.67	1.49	0.04	0.22
Employment (FTE)	5.09	20.97	6.99	19.85	1.9	1.12
GHG (t CO2 eq.)	1100	1,296.13	3,678.12	4,066.67	2,578.12	2,770.54
Energy use (TJ)	0.33	10.64	0.24	6.72	0.09	3.92

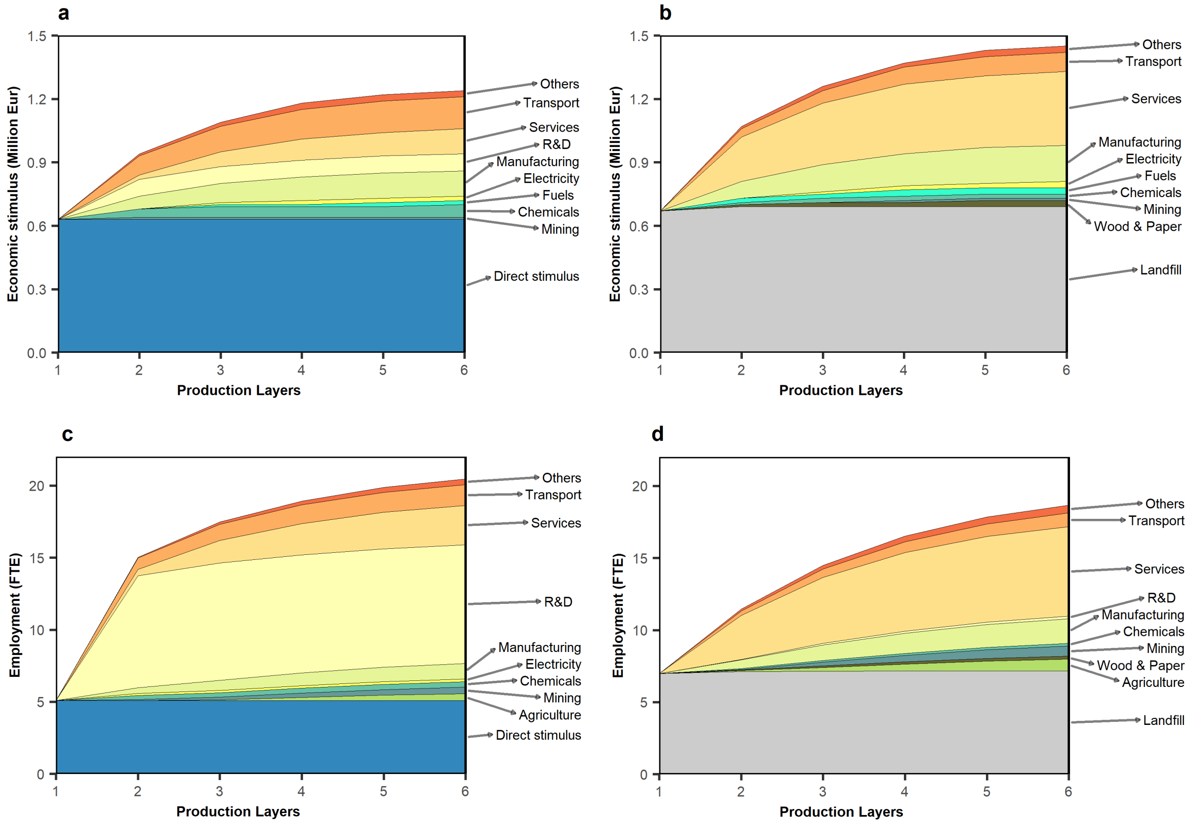
The direct impacts (q) of nutrient recovery express the intensity of the satellite accounts collected from the case study divided by the total industrial output (e.g. 84 employees per 16.5 million euros of industrial output). Direct landfill impacts, on the other hand, are those reported by the MRIO (Exiobase) database for the waste landfill sector. The multipliers (m) characterise the total impacts embodied in a unit of final demand and are derived with the IO analysis (Eq. 5). Overall, accounting for all upstream spillover effects, using sewage sludge for organic fertiliser production generates more jobs and reduces more GHG emissions than landfilling does. By contrast, landfilling stimulates the economy more and reduces energy carrier use more. Comparing q with m, we observe that energy consumption is mainly indirect (on average, 96.6% $\left[1 - \frac{1}{2}\left(\frac{0.33}{10.64} + \frac{0.24}{6.72}\right)\right]$ of the total impact), and so are the social (70.5%) and economic (52.7%) impacts. By contrast, GHG emissions are mainly direct (87.6%).

3.4.2 Production layer decomposition

Several upstream suppliers are required to provide inputs that are ultimately necessary for the waste management service that WWTPs purchase. The PLD analysis unravels the contributions to the footprints of different sectors disaggregating the total impacts (m) by upstream production layers. Each production layer signifies the sum of all contributions from supply chains of nth order (Section 2.2). We provided two sets of PLDs (Figure 1-2) for both nutrient recovery and landfilling strategies (for a demand shock of one million), illustrating the different sectors involved in the cascading effects and footprints associated with the two sewage sludge management strategies.

Social and economic footprints: Figure 3.1 illustrates the socioeconomic requirements of the two sewage sludge management alternatives. Layer 1 illustrates the on-site impacts occurring at the production plants reported in Table 3 (e.g. 0.63 million of economic stimulus and 5.09 FTEs promoted by one million services purchased for recycling 10,000 tonnes of sewage sludge). Layer 2 includes all contributions to footprints from direct suppliers (e.g. producers of lime, sulphuric acid and gypsum [Table 1] in the nutrient recovery strategy). Layer 3 includes the suppliers of suppliers (e.g. suppliers of the energy needed in the production of lime purchased by the waste management industry). After the fifth production layer, the graphs tend to converge to the total impact reported

Figure 3.1: Cumulative production layer decompositions for the socioeconomic footprints of the nutrient recovery (a, c) and landfilling (b, d) strategies.



Note: Industries are aggregated according to the Supplementary Information scheme provided in the online version of the paper

in Table 3, as the contributions from additional suppliers of suppliers become marginal. Six production layers are sufficient to account for more than 96% of the socioeconomic impacts. Excluding footprint contributions after the second production layer would cause a truncation error of 26% in economic terms and 32% in social terms.

The nutrient recovery strategy needs to source many inputs from the transportation, research and development (R&D), manufacturing and chemical sectors in order to support capital and operating costs. Further upstream, direct suppliers need inputs, with business services (including wholesale and retail trade), electricity, fuels and mining products as the major commodities. The substantial differences between the nutrient recovery and landfilling strategies mainly concern R&D and mining, which are socioeconomically relevant in the nutrient recovery strategy but not in landfilling, in which other business activities (including commission trade and renting of machinery and equipment) play the most important role.

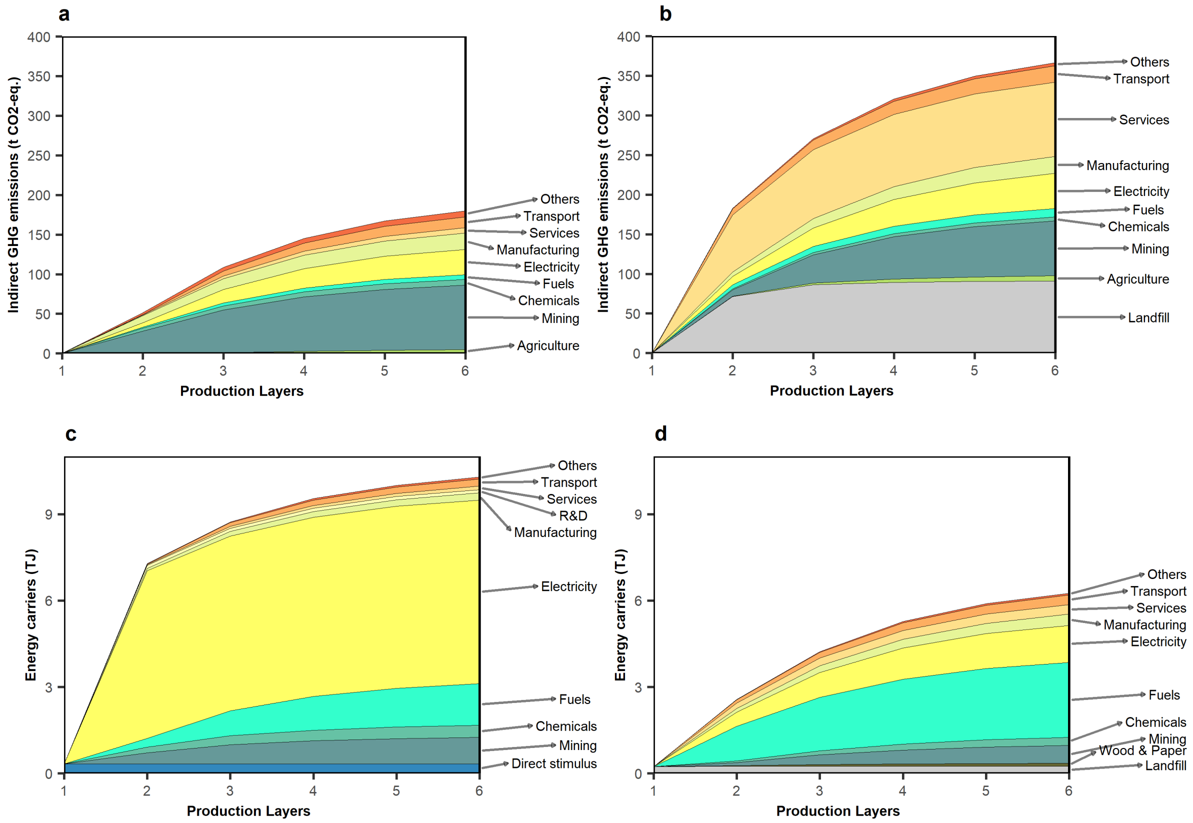
Regarding the geographical distribution of footprints, most inputs are purchased domestically, and the main trading partners are the same for both strategies. The nutrient recovery strategy accounts for 94% of the economic stimulus and 87% of jobs at the national level, +4% and +13%, respectively, compared with landfilling. Germany is the most economically involved country, given its high trade of industrial machinery, equipment and motor vehicles with Italy, followed by China and the US for chemical imports. At the social level, jobs abroad are mainly related to mining (in particular, the extraction of crude petroleum) in the Asia and Pacific area.

Environmental and energy footprints: Figure 3.2 illustrates the indirect GHG emissions and energy consumption footprints. As much as 95% of the environmental impacts are domestic in origin (Table 3). For illustration purposes, we represented only indirect emissions (Figure 2), so production layer 1 starts from the origin. For completeness, 1,100 and 3,678.12 t CO2-eq. (q in Table 3) should be recalled in Figures 1a and 2a, respectively. Including direct impacts, six production layers account for 98% of the total GHG emissions m. Excluding impacts beyond direct suppliers (i.e. production layers beyond the second), the underestimation of footprints is 10% for the nutrient recovery strategy and 4% for landfilling, which are lower than those for other footprints because GHG emissions occur primarily on-site.

The main sectors responsible for indirect GHG emissions in waste management strategies are mining, energy, transportation and other services. Mining includes the externalities associated with natural gas and crude oil extraction in Russia, Asia and the Pacific. In addition to these, landfilling generates the intermediate demand for several emission-intensive services for the disposal of other wastes (e.g. paper, wood and textiles), some of which are imported (e.g. landfilling from Portugal).

In terms of energy carriers (including electricity, heat, and solid, liquid and gaseous fuels), the nutrient recovery strategy involves greater energy demand than landfilling. During sewage sludge recycling for nutrient recovery, energy is required to treat and stabilise the biomass with chemicals (sulphuric acid and lime) and minerals (gypsum). In our case study, the biorefinery relies mainly on renewable energy (photovoltaics), which accounts for 56% of the total energy consumption. On the contrary, the total energy consumption of landfilling comes mainly (30%) from petroleum refineries. The rest come precisely from the chemical and mining industries, whose energy costs usually account for most of the gross production costs. In this case, if the impact assessment had stopped at the direct supplier level, it would have underestimated the total energy consumption by 30% for the nutrient recovery strategy and 59% for landfilling. Finally, in line with the profile of responsibility for GHG emissions, part of the total energy consumption comes from abroad (16% for nutrient recovery and 37.6% for landfilling), particularly natural gas imported from Russia and Middle Eastern countries and nuclear electricity from France.

Figure 3.2: Cumulative production layer decompositions for the environmental and energy footprints of the nutrient recovery (a, c) and landfilling (b, d) strategies.



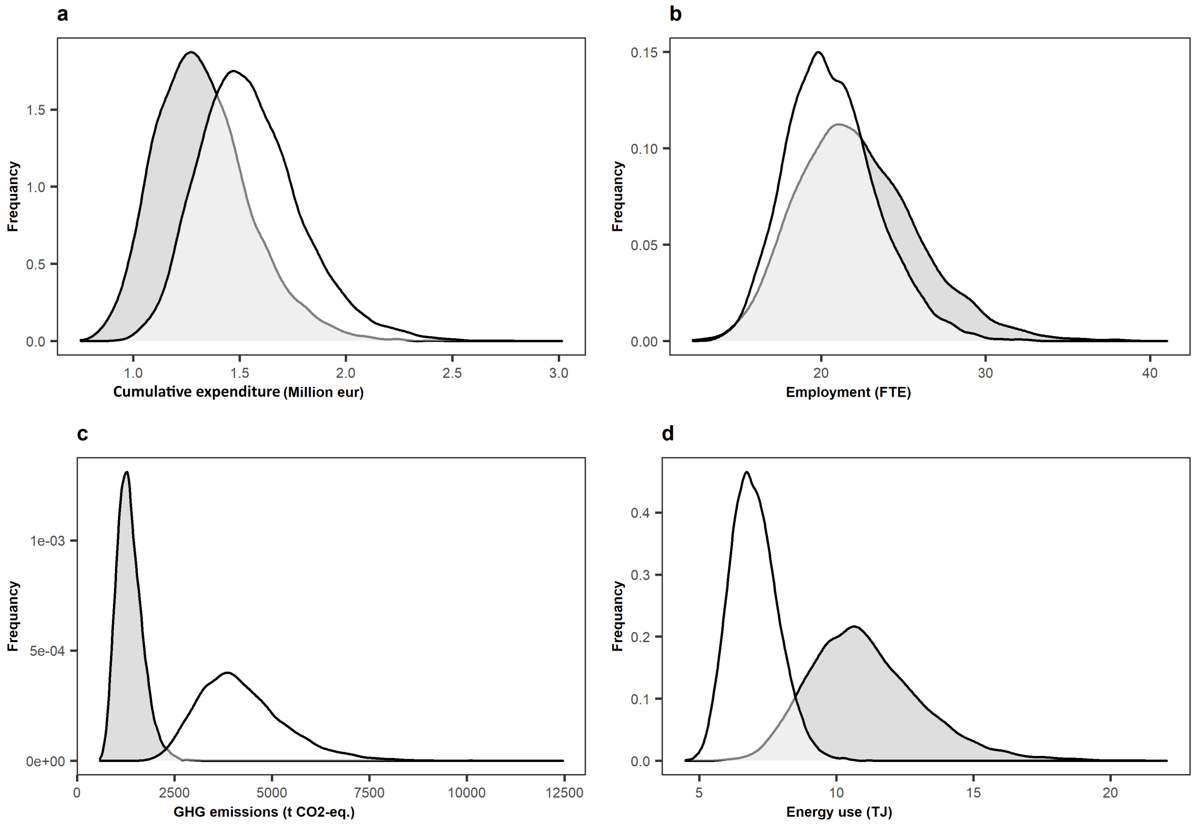
3.4.3 Uncertainty analysis

The uncertainty of total impacts was determined by uncertainties in basic data, which refer to uncertainties of the data sources and to the assumptions of the IO analysis (e.g. fixed proportions or constant prices). To estimate the errors associated with multipliers (m), we run 10,000 Monte Carlo simulations, including parametrical uncertainty of the entire MRIO database, process-based data and satellite data accounts assuming log-normally distributed errors in the basic statistical matrices Q, T, and Y.

Figure 3.3 shows the frequency distributions of the perturbated values of the results of the hybrid IO analysis using Monte Carlo simulations. We found that the multipliers of nutrient recovery are certain at the 95.5% level of confidence: (i) between 1.00 and 1.73 million euros of economic stimulus, between 16.84 and 28.92 FTEs, between 885.55 and 1952.63 t CO2-eq., and between 8.16 and 14.84 TJ.

Table 3 compares the frequency distributions (normalised by the baseline results) of the waste management strategies using the interquartile range (IQR) and coefficient of quartile variation (CQV). Except for GHG emissions, the same perturbations of the data

Figure 3.3: Frequency distributions of perturbed multipliers obtained from 10,000 Monte Carlo iterations.



Legend: grey = nutrient recovery; white = landfilling

sources generated more uncertain multipliers (m) for the nutrient recovery strategy than for landfilling. This can be observed by comparing the IQR or CQV values between waste management strategies (Table 3). These results can be explained as follows. Considering that the multiplier computation involves numerous additions of elements in T, the errors in the source data cancel out because of their stochastic nature. Emissions were less affected by this condition, being mainly on-site (Table 3). The same applies to the nutrient recovery strategy, which has a production function (Table 1) involving fewer sectors than the general landfilling sector of Exiobase.

Based on their variance and median, the distributions of an indicator may overlap with one another and contradict the baseline comparison made in Table 1. Despite the higher uncertainty, the distributions of GHG emissions overlap less than the other indicators do, and only in 0.05% of cases does the nutrient recovery strategy generate more GHG emissions than landfilling does. Similarly, the probability that this type of sewage sludge recycling consumes less energy in total than landfilling is also low (1.19%). For socioeconomic indicators, on the other hand, overlaps are likelier, with a 24.05% probability

Table 3.3: Statistics for the uncertainty of the perturbed multipliers normalised by baseline values.

Indicators	Economic stimulus		Employment		GHG emissions		Energy use	
Strategy	NR	L	NR	L	NR	L	NR	L
Q1	0.92	0.926	0.935	0.943	0.859	0.84	0.907	0.95
Median	1.028	1.025	1.044	1.029	1.008	0.998	1.019	1.032
Q3	1.149	1.136	1.168	1.127	1.19	1.192	1.155	1.124
IQR	0.229	0.21	0.234	0.185	0.331	0.352	0.247	0.173
CQV	0.111	0.102	0.111	0.089	0.162	0.173	0.12	0.084

Note: NR, Nutrient Recovery; L, Landfilling; Q1, First quartile; Q3, Third quartile; IQR, Interquartile range; CQV, Coefficient of quartile variation.

that replacing landfilling with sewage sludge recycling for nutrient recovery would possibly stimulate the economy more and with 37.62% generating fewer jobs than before.

3.5 Discussion

The literature shows that experts' discretion regarding which activities and products to include in the total impact assessment often leads to truncation errors (Crawford, 2018). Applying a hybrid IO analysis, instead, we handle the infinite upstream supply chains involved in the sustainability footprints of the nutrient recovery strategy from sewage sludge and landfilling. The results indicate that the sum of the impact contributions from supply chains of the sixth order (i.e. six production layers) accounts for more than 95% of the total footprints. In addition, studying the indirect impacts of the waste management strategies resulted crucial to assessing their sustainability.

Considering indirect impacts can overturn conclusions drawn from a superficial first glance. For example, considering only the on-site (direct) social impacts, recycling sewage sludge for agricultural purposes appears to be less labour intensive than landfilling, potentially reducing 190 jobs for one million tonne of sewage sludge recycled rather than landfilled. However, when the analysis also considers the indirect social impacts associated with the final demand for the sludge management service (i.e. the FTEs promoted upstream to meet the demand for inputs needed in order to recycle sewage sludge for agricultural purposes), the opposite is true, and for every tonne of sludge not landfilled but recycled, 112 jobs are generated, especially for the RD sector. Indeed, R&D plays a crucial role along the entire supply chain, promoting production safety and plant efficiency. Technological innovation, patents and investment influence the competitiveness of waste management strategies. In this study, RD investments also represent a cost-saving opportunity for agricultural producers, as technological progress is directed towards the provision of no-cost services related to precision agriculture (e.g. detailed information maps of soil conditions).

Overall, sewage sludge management can have significant economic and environmental impacts. In 2020, the amount of sewage sludge disposed of in landfills in Italy was more than 1.6 million tonnes (53.3% of the total amount). According to Table 2, in the extreme scenario in which the entire amount disposed of in landfills is directed to the nutrient recovery strategy (i.e. 160 million of demand shock), there is a potential reduction in total GHG emissions of 0.44 million tonnes CO2-eq. (0.277 t CO2-eq. per tonne of sewage sludge recycled instead of landfilled). However, this emission reduction comes at a cost. Recycling sewage sludge otherwise landfilled and producing organic fertiliser can reduce the (intermediate) use of goods and services by all industries in the economy (i.e. economic stimulus) by 35.2 million (22 euros per tonne of sludge). Hence, these findings suggests that the nutrient recovery as a strategy to reduce GHG emissions would cost to society (in terms of economic stimulus) 79.4 (22/0.277) euros per tonne of CO2-eq. Moreover, if all sewage sludge destined for landfills were converted to organic fertiliser, Italy would experience an increase in energy consumption of 627.2 TJ (0.392 GJ per tonne of sewage sludge). The reduction in GHG emissions takes this increase into account. However, special attention must be given to the energy resources and sources used by the country so as not to compromise the resulting environmental benefits. Nonetheless, as the uncertainty analysis shows, GHG emission reduction was the most certain (at a 99.95% confidence level) among the estimated impacts.

This study suggests how the circular management of sewage sludge for agricultural purposes can contribute significantly to the goals of the European Green Deal. In addition to reducing emissions, efficient nutrient recovery from sewage sludge has the potential to strengthen agricultural production processes, making them more efficient and resilient, with high yields and high profit margins. Therefore, policymakers need to ensure a more consistent and up-to-date regulatory framework. In particular, the European Directive 278/1986 should consider the technological and scientific developments achieved in recent years to prevent harmful effects on soils, vegetation, animals and humans when regulating the use of sewage sludge in agriculture. In addition, while various regulations consistently commit companies to numerous stringent safety standards, they are not adequately brought to the public's attention. Achieving an efficient and circular transition requires consumer awareness (Qi and Roe, 2017) of the responsibilities and protections associated with sludge management, which ensure food safety and the absence of environmental damage if properly respected. Several countries, such as the UK, the US, Australia and New Zealand, have developed specific assurance schemes for fertilisers made from sewage sludge. This scheme reduces barriers to use by formalising the products, establishing standardised production procedures, limiting contaminant content and testing protocols (Moya et al., 2019).

3.6 Conclusion

Replacing sewage sludge landfilling with other waste management solutions can result in significant impacts on the socioeconomic balance of the existing economic system, as well as on the environment. Several studies have demonstrated the agronomic benefits of applying treated sewage sludge on soil properties and crop yields, but to the best of our knowledge, none has consistently focused on the overall economic, social and environmental implications in a single framework. Accordingly, the present study was designed to consistently determine the footprints for all sustainability dimensions of agricultural sewage sludge treatment and to compare them with those of landfilling.

Using a hybrid IO analysis, we evaluated direct and indirect footprints. Overall, the results show that recycling sewage sludge for agricultural purposes promotes job creation and reduces GHG emissions more than landfilling does; the latter, by contrast, stimulates greater economic growth and lower energy consumption. Indirect impacts play a key role in these sustainability performances, which are characterised by different levels of uncertainty. We accounted for uncertainties in the results with an error propagation method based on Monte Carlo simulations. The GHG emissions per final demand of sludge treatment were the most uncertain of the estimated footprints. However, even when the uncertainty of the estimate is considered, the production of organic fertilisers from sewage sludge otherwise destined for landfills has a high probability of reducing GHG emissions in the country where production takes place. There is also likely to be a significant increase in energy consumption, which we should pay special attention to, so as not to compromise the resulting environmental benefits.

The generalisability of our results is subject to certain limitations. One source of weakness is the focus on a single case study and country. In fact, the results correspond to a specific nutrient valorisation strategy involving sewage sludge in Italy, where the biorefinery sources the primary inputs. If modelling is done in a different country, the direct impacts might be similar, but the total impacts will change according to the specific sourcing of inputs of the country's sectors. Furthermore, IO analysis requires several assumptions, such as a fixed production structure, constant returns to scale and fixed commodity prices that limit the ability to cope with modern economic systems. Therefore, the analysis mainly emphasises short-term effects. Next, although we attempted to capture stochastic variations of data and calculation procedures, our uncertainty analysis did not address systematic error sources (such as changes in import structure or choice of currency conversion factors). Finally, further research on the sustainability implications of sewage sludge management should conduct a systematic comparison with other waste management strategies (including incineration, composting, biogas production).

References

Beckman, J. and Riche, S. (2015). Changes to the natural gas, corn, and fertilizer price relationships from the biofuels era. Journal of Agricultural and Applied Economics 47(4): 494–509. doi: 10.1017/aae.2015.22

Beghin, J. and Nogueira, L. "A Perfect Storm in Fertilizer Markets." CAP Series 21-1107, Center for Agricultural Profitability, University of Nebraska-Lincoln, Nov. 17, 2021.

Behnassi, M., El Haiba, M. Implications of the Russia–Ukraine war for global food security. Nat Hum Behav 6, 754–755 (2022). https://doi.org/10.1038/s41562-022-01391-x

Bess, R., and Z.O. Ambargis. 2011. "Input-Output Models for Impact Analysis: Suggestions for Practitioners Using RIMS II Multipliers." BEA Working Papers No. 0081, Bureau of Economic Analysis. https://ideas.repec.org/p/bea/wpaper/0081.html

Council of the European Communities. (1991). Council Directive of 21 May 1991 concerning urban waste water treatment (91/271/EEC). Official Journal of the European Communities L: 135/40-52.

Crawford, R. H. et al. (2018) 'Hybrid life cycle inventory methods – A review', Journal of Cleaner Production, 172, pp. 1273–1288. doi: 10.1016/j.jclepro.2017.10.176.ù

Cruz, L. et al. (2017) 'Assessing an agri-food development strategy: A bi-regional input-output model with resource-constrained sectors', European Review of Agricultural Economics, 44(5), pp. 860–882. doi: 10.1093/erae/jbw028.

Diacono, M. and Montemurro, F. (2010). Long-term effects of organic amendments on soil fertility. A review. Agronomy for Sustainable Development 30(2): 401–422. doi: 10.1051/agro/2009040.

Eiser, D. and Roberts, D. (2002) 'The employment and output effects of changing patterns of afforestation in Scotland', Journal of Agricultural Economics, 53(1), pp. 65–81. doi: 10.1111/j.1477-9552.2002.tb00006.x.

European Commission. (2019). The European Green Deal. European Commission 53(9): 24. doi: 10.1017/CBO9781107415324.004

European Commission. (2020). Sustainable Europe Investment Plan. (accessed July 2022)

European Commission. (2022). Price dashboard. (accessed July 2022)

Fertilizers Europe. (2022). EU key partners in fertilizer trade (2020).(accessed July 2022)

Godfray, H. Charles J., John R. Beddington, Ian R. Crute, Lawrence Haddad, David Lawrence, James F. Muir, Jules Pretty, Sherman Robinson, Sandy M. Thomas, and Camilla Toulmin. 2010. "Food Security: The Challenge of Feeding 9 Billion People." Science 327(5967):812–18.

Godfray, H. C. J., Beddington, J. R., Crute, I. R., Haddad, L., Lawrence, D., Muir,

J. F., Pretty, J., Robinson, S., Thomas, S. M., Toulmin, C. (2010). Food Security: The Challenge of Feeding 9 Billion People. Science, 327(5967), 812–818. https://doi.org/10.1016/j.geoforum.2018.02.030

Hudcová, H., Vymazal, J. and Rozkošný, M. (2019). Present restrictions of sewage sludge application in agriculture within the European Union. Soil and Water Research 14(2): 104–120. doi: 10.17221/36/2018-SWR

Hughes, D. W. (2003). Policy uses of economic multiplier and impact analysis. Choices, 18(2), 25-29.

Hyytiä, N. (2014) 'Farm diversification and regional investments: Efficient instruments for the CAP rural development targets in rural regions of Finland?', European Review of Agricultural Economics, 41(2), pp. 255–277. doi: 10.1093/erae/jbt022.

Jamil, M., Qasim, M. and Umar, M. (2006). Utilization of sewage sludge as organic fertiliser in sustainable agriculture. Journal of Applied Sciences 6(3): 531–535. doi: 10.3923/jas.2006.531.535

Kelessidis, A. and Stasinakis, A. S. (2012). Comparative study of the methods used for treatment and final disposal of sewage sludge in European countries. Waste Management 32(6): 1186–1195. doi: 10.1016/j.wasman.2012.01.012

Kumar, V., Chopra, A. K. and Kumar, A. (2017). A review on sewage sludge (biosolids) a resource for sustainable agriculture. Archives of Agriculture and Environmental Science 2(4): 340–347. doi: 10.26832/24566632.2017.020417

Kumar Bhatt, M., Labanya, R. and Joshi, H. C. (2019). Influence of long-term chemical fertilizers and organic manures on soil fertility – A review. Universal Journal of Agricultural Research 7(5): 177–188. doi: 10.13189/ujar.2019.070502

Lenzen, M., Sun, Y. Y., Faturay, F., Ting, Y. P., Geschke, A., Malik, A. (2018). The carbon footprint of global tourism - Supplementary Information. Nature Climate Change, 8(6), 522–528. https://doi.org/10.1038/s41558-018-0141-x

Lenzen, M. and Rueda-Cantuche, J. M. (2012). A note on the use of supply-use tables in impact analyses. Sort 36(2): 139–152.

Lenzen, M., Wood, R. and Wiedmann, T. (2010). Uncertainty analysis for multi-region input-output models – A case study of the UK'S carbon footprint. Economic Systems Research 22(1): 43–63. doi: 10.1080/09535311003661226

Leontief, W. (1986). Input-Output Economics. New York, Oxford University Press.

Liu, B., Wei, Q., Zhang, B., Bi, J. (2013). Life cycle GHG emissions of sewage sludge treatment and disposal options in Tai Lake Watershed, China. Science of the Total Environment 447: 361–369. doi: 10.1016/j.scitotenv.2013.01.019

Løes, A. K. and Adler, S. (2019). Increased utilisation of renewable resources: Dilemmas for organic agriculture. Organic Agriculture 9(4): 459–469. doi: 10.1007/s13165-018-00242-2

Løes, A. K., Bünemann, E. K., Cooper, J., Hörtenhuber, S., Magid, J., Oberson, A.,

Möller, K. (2017). Nutrient supply to organic agriculture as governed by EU regulations and standards in six European countries. Organic Agriculture 7(4): 395–418. doi: 10.1007/s13165-016-0165-3

Lombardi, L., Nocita, C., Bettazzi, E., Fibbi, D., Carnevale, E. (2017). Environmental comparison of alternative treatments for sewage sludge: An Italian case study. Waste Management 69: 365–376. doi: 10.1016/j.wasman.2017.08.040

Malik, A., Lenzen, M., Ely, Rô. N., Dietzenbacher, E. (2014). Simulating the impact of new industries on the economy: The case of biorefining in Australia. Ecological Economics 107: 84–93. doi: 10.1016/j.ecolecon.2014.07.022

Malik, A., Lenzen, M., Ralph, P. J., Tamburic, B. (2015). Hybrid life-cycle assessment of algal biofuel production. Bioresource Technology 184: 436–443. doi: 10.1016/j.biortech.2014.10.132

Malik, A., McBain, D., Wiedmann, T. O., Lenzen, M., Murray, J. (2019). Advancements in input-output models and indicators for consumption-based accounting. Journal of Industrial Ecology 23(2): 300–312. doi: 10.1111/jiec.12771

Montanarella, L. and Panagos, P. (2021). The relevance of sustainable soil management within the European Green Deal. Land Use Policy 100(June 2020): 104950. doi: 10.1016/j.landusepol.2020.104950

Moya, B., Parker, A. and Sakrabani, R. (2019). Challenges to the use of fertilisers derived from human excreta: The case of vegetable exports from Kenya to Europe and influence of certification systems. Food Policy 85(February 2018): 72–78. doi: 10.1016/j.foodpol.2019.05.001

Moysiadis, V., Sarigiannidis, P., Vitsas, V., Khelifi, A. (2021). Smart farming in Europe. Computer Science Review 39: 100345. doi: 10.1016/j.cosrev.2020.100345

Murray, A., Horvath, A. and Nelson, K. L. (2008). Hybrid life-cycle environmental and cost inventory of sewage sludge treatment and end-use scenarios: A case study from China. Environmental Science and Technology 42(9): 3163–3169. doi: 10.1021/es702256w

Murray, A., Nelson, K. L. and Horvath, A. (2008). Supporting Information Hybrid Life-Cycle Environmental and Cost Assessment of Sewage Sludge Treatment and End-Use Scenarios: A Case Study from China Department of Civil and Environmental Engineering. Berkeley, CA: University of California.

Okuyama, Y., Santos, J. R. (2014). Disaster impact and input–output analysis. Economic Systems Research, 26(1), 1-12.

Qi, D. and Roe, B. E. (2017). Foodservice composting crowds out consumer food waste reduction behavior in a dining experiment. American Journal of Agricultural Economics 99(5): 1159–1171. doi: 10.1093/ajae/aax050

RoboResearch (2022). The Russia–Ukraine War's Impact on Global Fertilizer Markets. Available at: https://research.rabobank.com/far/en/sectors/farm-inputs/the-russia-ukraine-war-impact-on-global-fertilizer-markets.html: :text=Higher fertilizer prices and

Singh, R. P. and Agrawal, M. (2010). Variations in heavy metal accumulation, growth and yield of rice plants grown at different sewage sludge amendment rates. Ecotoxicology and Environmental Safety 73(4): 632–641. doi: 10.1016/j.ecoenv.2010.01.020

Smith, A. (2022). The Story of Rising Fertilizer Prices, ARE Update. Available at: https://coststudies.ucdavis.edu/en/current (accessed July 2022)

Soriano-Disla, J. M., Navarro-Pedreño, J. and Gómez, I. (2010). Contribution of a sewage sludge application to the short-term carbon sequestration across a wide range of agricultural soils. Environmental Earth Sciences 61(8): 1613–1619. doi: 10.1007/s12665-010-0474-x

Sperling, F., Havlik, P., Denis, M., Valin, H., Palazzo, A., Gaupp, F., Visconti, P. (2022). Toward resilient food systems after COVID-19. Current Research in Environmental Sustainability 4(May 2021): 100110. doi: 10.1016/j.crsust.2021.100110

Stadler, K., Wood, R., Bulavskaya, T., Södersten, C. J., Simas, M., Schmidt, S., Usubiaga, A., Acosta-Fernández, J., Kuenen, J., Bruckner, M., Giljum, S., Lutter, S., Merciai, S., Schmidt, J. H., Theurl, M. C., Plutzar, C., Kastner, T., Eisenmenger, N., Erb, K. H., ... Tukker, A. (2018). EXIOBASE 3: Developing a time series of detailed environmentally extended multi-regional input-output tables. Journal of Industrial Ecology 22(3): 502–515. doi: 10.1111/jiec.12715

Tamburini, G., Bommarco, R., Wanger, T. C., Kremen, C., van der Heijden, M. G. A., Liebman, M., Hallin, S. (2020). Agricultural diversification promotes multiple ecosystem services without compromising yield. Science Advances 6(45): eaba1715. doi: 10.1126/SCI-ADV.ABA1715

Tarne, P., Lehmann, A. and Finkbeiner, M. (2018). A comparison of Multi-Regional Input-Output databases regarding transaction structure and supply chain analysis. Journal of Cleaner Production 196: 1486–1500. doi: 10.1016/j.jclepro.2018.06.082

Thomassin, P. J. and Baker, L. (2000) 'Macroeconomic impact of establishing a large-scale fuel ethanol plant on the Canadian economy', Canadian Journal of Agricultural Economics, 48(1), pp. 67–85. doi: 10.1111/j.1744-7976.2000.tb00266.x.

Wahdat, Ahmad Zia, and Jayson L. Lusk. 2022."The Achilles Heel ofthe U.S. Food Industries: Exposure to Labor and Upstream Industries in the Supply Chain." American Journal of Agricultural Economics 1–20. https://doi.org/10.1111/ajae.1233120U.S. FOOD INDUSTRIES'EXPOSURE TO INPUTS

Ward, H. et al. (2018) 'Truncation Error Estimates in Process Life Cycle Assessment Using Input-Output Analysis', Journal of Industrial Ecology, 22(5), pp. 1080–1091. doi: 10.1111/jiec.12655.

Wei, W., Li, J., Chen, B., Wang, M., Zhang, P., Guan, D., Meng, J., Qian, H., Cheng, Y., Kang, C., Feng, K., Yang, Q., Zhang, N., Liang, X., Xue, J. (2021). Embodied greenhouse gas emissions from building China's large-scale power transmission infrastructure. Nature Sustainability 4(8): 739–747. doi: 10.1038/s41893-021-00704-8

Yoshida, H., Christensen, T. H. and Scheutz, C. (2013). Life cycle assessment of sewage sludge management: A review. Waste Management and Research 31(11): 1083–1101. doi: 10.1177/0734242X13504446

Zaman, M., Matsushima, M., Chang, S. X., Inubushi, K., Nguyen, L., Goto, S., Kaneko, F., Yoneyama, T. (2004). Nitrogen mineralization, N2O production and soil microbiological properties as affected by long-term applications of sewage sludge composts. Biology and Fertility of Soils 40(2): 101–109. doi: 10.1007/s00374-004-0746-2

Chapter 4

Emerging Bio-based Alternatives: From Innovation Niches to Dominating Products?

Abstract

In the absence of historical data for econometric analyses of innovative, emerging bio-based products, our study provides qualitative and quantitative insights that can support policymakers on their journey through the complexities of the transformation toward a low-carbon bioeconomy. This study aims to understand which of the considered policies—investment cost subsidies and operating cost subsidies—is more effective in stimulating the bio-based market share and increasing the total market welfare. Following our objective, we build a stylized partial equilibrium model to analyze and understand the inner workings of mutually linked markets for fossil-based and innovative bio-based products. The effect of various potential policies on the market share in a competitive equilibrium is of particular interest as this indicator is closely related to the bioeconomy transition and the EU greenhouse gas emission reduction target. We calibrate the theoretical model using data for representative innovative biotechnology to produce 1,4-Butanediol to provide numerical estimates of the modeled policy effects. Finally, we perform a Monte Carlo simulation to relax the model assumptions and capture the uncertainty over the estimated efficiency of policy instruments.

Keywords: partial equilibrium model; subsidy; innovation, green deal; bioeconomy

4.1 Introduction

With the adoption of the Green Deal, the European Commission (EC) has confirmed its commitment to addressing the climate and environmental challenges facing our society (European Commission, 2019). The Deal, portrayed as "Europe's man on the moon moment" (Von der Leyen, 2019), is an ambitious and costly step to make Europe a climate-neutral continent by 2050, ensuring at the same time a fair and prosperous society through a competitive, resource-efficient economy (European Commission, 2019).

The Sustainable Europe Investment Plan (investment pillar of the European Green Deal) is expected to mobilize at least a quarter of the European Union's (EU) long-term budget for climate-related purposes (European Commission, 2020). At least one trillion euros of private and public investment is to be invested to achieve the key 2030 targets that include reducing greenhouse gas emissions by at least 55 percent compared to the 1990 levels, doubling the 2019 share (19.7 percent) of renewable energy sources, and reducing the energy consumption via improvements in energy efficiency by at least 32.5 percent. The new EU taxonomy regulation is argued to ensure the efficient movement of capital toward truly green investments (Taxonomy Regulation: Regulation (EU) 2020/852).

The bio-based circular economy, defined here as an economy based on renewable resources and designed to fulfill maximum efficiency while respecting the waste hierarchy (Vanhamaki et al., 2019), can play a crucial role in achieving these goals (European Commission, 2018). Creating greener value chains and more cost-effective industrial processes is believed to modernize and strengthen the EU industrial base. The production of innovative sustainable bio-based products (such as bio-solvents, bio-polymers, bio-surfactants, bio-lubricants, or bio-fuels) could reduce dependency on fossil fuels in critical European industries such as construction, packaging, textiles, chemicals, cosmetics, pharmaceutical ingredients, or consumer goods (European Commission, 2018). If a competitive market could be created, allocative and production efficiency would be achieved. Therefore, it is in the interest of the EU policymakers to support the maximum possible industry participation in a sustainable bio-based market, ensuring that the natural resources are efficiently allocated to provide the maximum satisfaction achievable by society.

Public actions in support of bio-based productions are mostly biased toward the bio-energy sector (Kircher, 2015). The most common way to support bioenergy and other renewable solutions is capital investment subsidies (IRENA, 2022). Along with the novelty of renewable technologies, bio-based production requires an intensive initial capital investment due to the low (energy) density and heterogeneity of biomass, which requires expensive pre-treatment and processing facilities (Jouvet et al., 2012). Fixed investment subsidies are, therefore, implemented for faster industry growth (Newes et al., 2011). In addition to fixed investment subsidies, operating cost subsidies (e.g., tax credits) are common worldwide and were widely used for first-generation biofuels (de

Gorter & Just, 2009). The stated policy objectives aim to reduce fossil-based products, environmental pollution, and CO2 emissions, as well as promote the adoption or diffusion of existing bio-based technology.

Together with the interest of institutions, concerns about the environment also have an increasingly visible influence on consumption, which shifts steadily toward purchasing green products (Morone et al., 2021). In 2008, Eurobarometer European Commission (2008) reported that 75 percent of Europeans were ready to buy green products even if they were more expensive than conventional ones. Several studies have recently shown the existence of a "green premium" linked to an increasing consumers willingness to pay for bio-based products. Reinders, Onwezen and Meeusen (2017) showed how global brands, such as Coca-Cola, could benefit from introducing bio-based product attributes to differentiate or reposition its product in the market. In addition, Morone et al. (2021) suggested that even a moderate increase in the price of fossil-based products can significantly impact consumer preference for more sustainable products. Labeling certification plays a key role in providing consumers with complete information on the true sustainability of these products (Morone et al., 2021). The "green" label, however, requires onerous certification and bio-based producers face higher costs (Dell'Erba, 2021).

Bio-based products may be identical to their fossil-based alternatives (so-called drop-in solutions; for example, ethylene, propylene, and 1,4-butanediol) or entirely new (so-called dedicated; for example, polylactic acid and polyhydroxyalkanoates). Drop-in solutions, which are homogeneous products for fossil-based equivalents, can be processed in available infrastructure. Thus, they are more likely to expand faster than dedicated bio-based products that need to develop new markets (De Jong et al., 2012). How quickly the shift from fossil to biomass sources will occur depends on the cost competitiveness of these inputs (Asada and Stern, 2018). Once commercialized, a product needs to capture market share for the firm to expand its production.

The effectiveness of policy instruments has been widely studied for biofuels (Clancy & Moschini, 2018), but neglected for alternative drop-in intermediate products. This article aims to understand how well policies—investment cost subsidies and operating cost subsidies—perform as incentives in stimulating higher market share for renewable inputs. In the absence of historical data for econometric analyses of innovative, emerging products, our study provides qualitative and quantitative insights that can support policymakers on their journey through the complexities of the transformation towards a low-carbon bioeconomy.

Following our objective, we build a stylized partial equilibrium model to analyze and understand the inner workings of mutually linked markets for 'dirty' (i.e., fossil-based) and 'clean' (i.e., emerging renewable bio-based) products. The effect of policy tools on the competitive market share in equilibrium is of particular interest as this indicator is closely related to the bioeconomy transition and the EU greenhouse gas (GHG) emission reduction target. We illustrate and calibrate the model using the empirical case of 1,4-Butanediol (1,4-BDO) to provide a numerical analysis of the policy tool impacts. Finally, we perform a Monte Carlo simulation to relax assumptions on parameter calibration and capture the effects of uncertainty on welfare changes.

4.2 A model

Consider two market segments that use an intermediate input for further production. The first segment is represented by firms that demand only clean intermediate input having a bio-based carbon content of 100% by mass, and the second segment is indifferent between bio-based and fossil-based intermediate inputs as both are considered equal. This means that for the second market segment, the two types of intermediate inputs are perfect substitutes, implying that the firms buying the product are willing to pay the lower price of both products. Denoting the price of 'clean' renewable (subscript c) intermediate input as p_c and the price of 'dirty' fossil-based (subscript d) chemical input as p_d, we assume that $p_d < p_c$, which is consistent with the historical pricing pattern (Spekreijse et al., 2019). We denote the demand for clean input as $x_c\left(p_c\right)$ and for fossil-based input as $x_d\left(p_d\right)$.

On the supply side, there are M dirty firms producing the fossil-based input and N clean firms producing the bio-based alternative. All firms are price takers in the input and output markets and are identical within their type. The clean firms would preferably sell all their production to the clean demand segment; however, because it is possible that their target demand will be smaller than the cumulative supply, they will sell the remaining quantity of clean production to the other (indifferent) demand segment at the lower price. Each clean firm can, however, choose optimally the share (f) of its production (q_c) that it will offer in the clean market segment. Thus, in the short-run, the profit (π_c) of each clean producer is

$$\pi_c = f q_c p_c + (1-f)\, q_c p_d - C\left(q_c\right) - F_c \tag{1}$$

where $C\left(q_c\right)$ is the operating cost based on output quantityq_c.[1] F_c is the capital amortization cost for a facility producing the bio-based product.

The first-order condition for equation (1) with respect to q_c determines the optimal level of production for each clean firm

$$q_c : f p_c + (1-f)\, p_d = MC\left(q_c\right) \tag{2}$$

Equation (2) is a reformulation of the usual condition for profit maximization of a competitive firm, except now the marginal revenue equals the weighted average of

[1]The cost function is determined by the underlying production technology.

the prices the clean firm receives. The optimal solutions to (2) can be written as $q_c(p_c, p_d, v_c, w_c)$, where v and w denote the input prices. How sensitive this producer is to a change in the monetary benefits due to the price difference between fossil-based and bio-based alternatives is captured by a function represented by equation (3)

$$f = \xi(p_c, p_d) \tag{3}$$

where $\xi(\cdot)$ is increasing in its argument (i.e., relative price), meaning that a greater gap between bio-based and fossil-based intermediate input prices in favor of the former leads to a greater shift-out of the supply for the clean market segment.

On the fossil-based side, the production technology for each firm determines the profit $\pi_d(p_d, q_d)$

$$\pi_d = p_d q_d - C(q_d) - F_d \tag{4}$$

where $C(q_d)$ is the operating cost for the dirty production depending on output quantity and input prices; F_d capital amortization cost. The first-order condition corresponding to equation (4), determines the supply of fossil-based input

$$p_d = MC(q_d) \tag{5}$$

Finally, equations (6) and (7) are market-clearing conditions that require that total demand equals total supply for each market segment. The demanders are divided into two groups. The first is a group of clean firms that are determined to purchase the bio-based intermediate input; hence their demand depends only on p_c. The second group consists of firms indifferent to the origin of the product. Their choice falls on a mix of input levels such that they minimize production costs, including emission costs. Because the clean segment of the demand for the intermediate input will only buy the product from bio-based producers, market equilibrium requires that

$$x_c(p_c) + x_d(p_c, p_d) = N f q_c \tag{6}$$

The demanders who are indifferent between clean and dirty inputs will buy all supply of fossil-based providers and the rest from bio-based producers

$$x_d(p_c, p_d) = N(1 - f)q_c + M q_d \tag{7}$$

The market equilibrium is determined by solving the system of equations (1) – (7) forp_c, p_d, q_c, q_d, f, M, and$N$.

Expected welfare is the sum of intermediate consumer and producer surplus, damages from environmental externalities, and government cost of policies, which can be simulated as reductions in $F_c$and $C(q_c)$ for investment cost subsidies and operating cost subsidies, respectively.

4.2.1 Functional forms

The model requires the specification of functional forms for production functions, profitability functions, cost functions, demand functions, and the allocation of clean production. For the set of production types i(c and d), we assumed firms produce quantity q_i according to a Leontief technology that exhibits decreasing returns to scale

$$q_i = \min \{\alpha_i z_i, \beta_i k_i\}^{1/t_i} \tag{8}$$

where z_i and k_i are inputs factors, $\alpha_i, \beta_i > 0$ technological coefficients, and $t_i > 1$ ($1/t_i$ determines returns to scale). The corresponding cost function is

$$C(q_i, v_i, w_i) = \left(\frac{v_i}{\alpha_i} + \frac{w_i}{\beta_i}\right) q_i^{t_i} \tag{9}$$

with v_i, w_i unit input prices for raw materials and other costs, respectively. The assumption of decreasing returns to scale allows us to determine the number of dirty (M) and clean (N) firms in the equilibrium.

Next, we used the logistic function (10) to model the propensity of clean producers to allocate their bio-based production between clean and indifferent consumers

$$f(p_c, p_d) = \frac{A}{1 + Ge^{-cr}} + D \tag{10}$$

where r is the relative price; parameters A and D relate to the asymptotes of the logistic function asymptotes; and parameters G and c to its shape.

The profitability (ω_i)of each firm is assumed to take the constant elasticity form

$$\omega_i = E_i {p_i}^{\sigma_i^p} {v_i}^{\sigma_i^v} {k_i}^{\sigma_i^k} {F_i}^{\sigma_i^K} \tag{11}$$

where E_i is a scaling factor andσ_is are the profitability elasticities with respect to input (v_i, k_i, K_i) and output (p_i) prices.

Regarding the demand side, the model assumes two groups of firms willing to purchase the intermediate input. For the demand of clean firms that are determined to purchase only bio-based inputs, we assume a constant elasticity form

$$X_c = B_c {p_c}^{\gamma_c} \tag{12}$$

with γ_c the own-price elasticity of demand X_c and B_c the scale parameter. For the second group, we assume indifferent firms minimize the total cost

$$\min C = p_d x_d + p_c x_c + \frac{1}{2}\theta(\mu_d x_d + \mu_c x_c)^2 s.t. q_p = a x_d + b x_c \tag{13}$$

with μ_i denoting emission intensities, θ the unit cost associated to emissions, q_p quantity of final product, and a and b are technological coefficients. From equation (13), we derive the conditional factor demand functions

$$x_d = \frac{b\,(ap_c - bp_d)}{\theta(a\mu_c - b\mu_d)^2} + \frac{\mu_c}{(a\mu_c - b\mu_d)} q_p \tag{14}$$

$$x_c = \frac{a\,(bp_d - ap_c)}{\theta(a\mu_c - b\mu_d)^2} - \frac{\mu_d}{(a\mu_c - b\mu_d)} q_p \tag{15}$$

Based on equations (14) and (15), own-prices and cross-prices demand elasticities can be computed as follows (will be used in the calibration stage)

$$\eta_d = -\frac{b^2 p_d}{b\,(ap_c - bp_d) + \theta\,(a\mu_c - b\mu_d)\,\mu_c q_p} \tag{16}$$

$$\eta_{dc} = \frac{abp_c}{b\,(ap_c - bp_d) + \theta\,(a\mu_c - b\mu_d)\,\mu_c q_p} \tag{17}$$

$$\eta_c = -\frac{a^2 p_c}{a\,(bp_d - ap_c) - \theta\,(a\mu_c - b\mu_d)\,\mu_d q_p} \tag{18}$$

$$\eta_{cd} = \frac{abp_d}{a\,(bp_d - ap_c) - \theta\,(a\mu_c - b\mu_d)\,\mu_d q_p} \tag{19}$$

Turning to welfare, producers surplus is defined as the total profit that producers earn by producing and selling the (clean or dirty) product. A change in producer surplus is measured the difference in profits after and before a policy has been imposed.

Because the demand function of the clean consumer is assumed to be a hyperbola, we set the choke price to 10000 USD/kg and keep it at that level in all simulations. Finally, the cost of externalities is given by the difference in CO2 emissions multiplied by the marginal external cost of emissions (normalized to 1 USD/ton of CO2 eq.), while the cost of the policy is estimated as the difference in costs ($F_i or w_i$) times the number of clean firms (N).

4.3 Parameterization and empirical illustration

Model parameters are calibrated such as to replicate the 1,4-butanediol (BDO) market condition in 2021. 1,4-BDO is a key chemical building block used in the production of plastics, elastic fibers, polyesters, polyurethanes, and pharmaceuticals (Taylor et al., 2015). With an average price of \$3/kg (Business analytiq, 2022), the global 1,4-BDO market value was \$6.88 billion in 2021 and is expected to grow at a compound annual rate of 8.1 percent by 2030 (Grand View Research, 2022). The production is mainly concentrated in the Asia-Pacific region (60.7 percent), North America (20.1 percent), and Europe (15 percent) (Grand View Research, 2022). BDO is currently manufactured industrially from petrochemical feedstocks, mainly via Reppe chemistry using acetylene and formaldehyde as raw materials or via Mistubishi and Toyo Soda techniques employing butadiene (Taylor et al., 2015). Growing concerns about environmental impacts have encouraged the production of 1,4-BDO through renewable, low-cost raw materials (Silva, Ferreira and Borges, 2020). In particular, Genomatica commercialized a technology to produce 1,4-BDO by direct bioconversion from plant sugars[2] (Satam, Daub, and Realff, 2019). The bio-based production has been shown to be interchangeable with conventional BDO with no change in product performance, manufacturing procedures, or equipment (DSM Engineering Plastics, 2013). The current global bio-BDO market size was evaluated at $552.9 million (Research and Markets, 2022) but the production is mostly consumed internally by manufacturers for further processing.

We begin the parameterization (see Table 1 for a complete summary) by normalizing the input and output prices of dirty production to 1. This is a convenient parameterization, allowing direct comparison of parameters between productions. Since bio-BDO is not freely available in the market, we must assume p_c , which we conservatively set to be twice the value of p_d ($r = 2$). The same is assumed about the fixed cost ($F_c = 2$) due to the maturity of fossil-based production compared to bio-based one. Next, f can be calibrated from equation (10), which requires parameters A, D, and c.

When r approaches infinity in the logistic function, f approaches $A + D = 1$, from which we can calibrate A. When r approaches zero, the limit of $\frac{A}{1+G} + D$ approaches zero, and D can be calibrated assuming $G (G = 2)$. $c$is then calibrated as

$$c = -\frac{\ln\left[\frac{1}{G}\left(\frac{A}{f_0 - D} - 1\right)\right]}{r} \tag{20}$$

assuming the producer intends to allocate f_0 (almost the entire production, that is, 99 percent) to the bio-based market segment. By normalizing prices, input quantities can be allocated according to the relative share of cost items. Knowing that

$$q_i = t_i \frac{(v_i z_i + k_i w_i)}{\bar{p}_i} \tag{21}$$

with $\bar{p}_i$ equal to the average selling price (recalling that clean producer can sell bio-BDO for both prices), we set $z_c = 1.5$ and $k_c = 1.5$ such that for $t \to 1$, $q = 3$ (unit price of conventional BDO). Given F_i, the clean production cost consists of raw materials (30 percent), other operating costs (30 percent), and fixed costs (40 percent), according to Satam and Realff (2020). In contrast, dirty production depends less on fixed costs (25

[2]Genomatica's GENO BDO® technology has so far been licensed to BASF (world's leading producer of BDO), Novamont, and Cargill in a joint venture with HELM, called Qore.

Table 4.1: Baseline values and calibration

Description	Parameter	Unit	d	C	Source
Supply elasticities	ϵ	-	1.7	1.2	Assumed
Return to scale	1/t	-	0.63	0.55	Calibrated
Selling price	p	\$/kg	1	2	(Business analytiq, 2022)
Ratio between prices	r	-		2	Assumed
Initial allocation propensity	ϕ_0	-		0.99	Assumed
Parameter of the logistic function	G	-		2	Assumed
Parameter of the logistic function	A	-		1.5	Calibrated
Parameter of the logistic function	e	-		2.72	Calibrated
Parameter of the logistic function	c	-		2.85	Calibrated
Parameter of the logistic function	D	-		-0.5	Calibrated
Allocation propensity	ϕ	-		0.99	Calibrated
Feedstock	z	kg	1.5	1.5	(Chauvel and Lefebvre, 1989; Satam, Daub and Realff, 2019; (Pääkkönen, Tolvanen and Kokko, 2019; Satam and Realff, 2020)
Feedstock unit price	v	\$/kg	1	1	Normalized
Others	k	kg	1.5	1.5	Calibrated
Others unit price	w	\$/kg	1	1	Normalized
Feedstock Leontief coef.		-	7.96	4.3	Calibrated
Others Leontief coef.	β	-	7.96	4.3	Calibrated
Production capacity	q	kg	4.76	2.76	Calibrated
Fixed cost	F	\$	1	2	Assumed
Profitability	ω	-	0.19	0.1	Calibrated
Profitability scalar parameter	E	-	0.19	0.1	Calibrated
Elasticity of profitability with respect to p	σ_p	-	1	1	Assumed
Elasticity of profitability with respect to v	σ_v	-	-1	-1	Assumed
Elasticity of profitability with respect to w	σ_w	-	-1	-1	Assumed
Elasticity of profitability with respect to F	σ_F	-	-1	-1	Assumed
Input coef.	a	kg/kg	1	1	(Chauvel and Lefebvre, 1989)
Input coef.	b	kg/kg	1	1	(Chauvel and Lefebvre, 1989)
THF production	q_p	kg	1	1	Normalized
Indifferent demand	x	kg	0.95	0.05	Assumed; calibrated
Emissions per BDO	μ	CO2eq	2.5	1	(Forte et al., 2016)
Emissions unit costs		$\$/(CO2eq.)^2$	0.27	0.27	Calibrated
Number of firms	M;N	-	0.2	0.03	Calibrated; assumed
Bio-Demand elasticity	γ	-		-0.5	Assumed
Scalar	B	kg/\$		0.05	Calibrated
Clean demand	X	kg		0.03	Calibrated
Own-price demand elasticity	η	-	-1.7	-64.67	Computed
Cross-price elasticity	$\eta_c d$	-	3.4	32.33	Computed
Producer surplus		\$	0.15	0.02	Computed
Total cost of the indifferent consumer; Clean Consumer surplus		\$	1.86	8.95	Computed
Cost of Externalities		\$	2.373	0.07	Computed

percent) and more on operating costs (raw material and other operating cost each at 38 percent). Regarding the profitability, we calibrate it as

$$\omega_c = \frac{q_c \bar{p}_c}{z_c v_c + k_c w_c + \delta_c F_c} - 1 \tag{22}$$

Based on equation (11), we calibrate E_i assuming σ_i's are unitary elastic. The profitability depends on the decreasing return to scale parameter, which is calibrated by computing the own-price elasticity of supply (ε_i) from the first-order conditions ($t_i = 1/\varepsilon_i + 1$). Hence, profitability depends on supply elasticities. We set $\varepsilon_c = 1.2$ so that $\omega_c = 0.1$ and $\varepsilon_d = 1.7$ so that the profitability (in relative terms) of fossil-based production is twice as high.

Turning now to the demand-side, calibration requires values for emissions μ_i and parameters (a, b) of final production q_p. Without loss of generality, we set $\mu_d = 2.5$ and $\mu_c = 1$, which is in line with the academic literature that has shown a 60 percent reduction in GHG emission from bio-BDO compared to conventional BDO (Forte et al., 2016; De Bari et al., 2020). Regarding final applications and q_p, the major consumption of BDO relates to tetrahydrofuran (THF) production in the polymers and plastic industry (Cukalovic and Stevens, 2008). Thus, we considered THF producers as intermediate consumers. 1 kg of THF is obtained by dehydration of 1 kg of BDO (Li and Chen, 2019), thus parameters a, b are set equal to one for both the clean and dirty product. By normalizing $q_p = 1$, we can directly interpret x_c and x_d as shares of fossil- and bio-based consumption from the indifferent consumer. In the baseline, we assume $x_d = 0.95$. Next, the cost associated with emissions must satisfy

$$\theta = \frac{b\left(a p_c - b p_d\right)}{\left(a\mu_c - b\mu_d\right)^2 \left[x_d - \frac{\mu_c}{(a\mu_c - b\mu_d)} q_p\right]} \tag{23}$$

and equation (15) can now be used to define the x_c. Finally, we set $N = 0.033$ such that the share of bio-based consumption in total consumption, that is, $(X_c + x_c)/(X_c + x_c + x_d)$, is equal to 0.08 in baseline (Research and Markets, 2022), knowing that

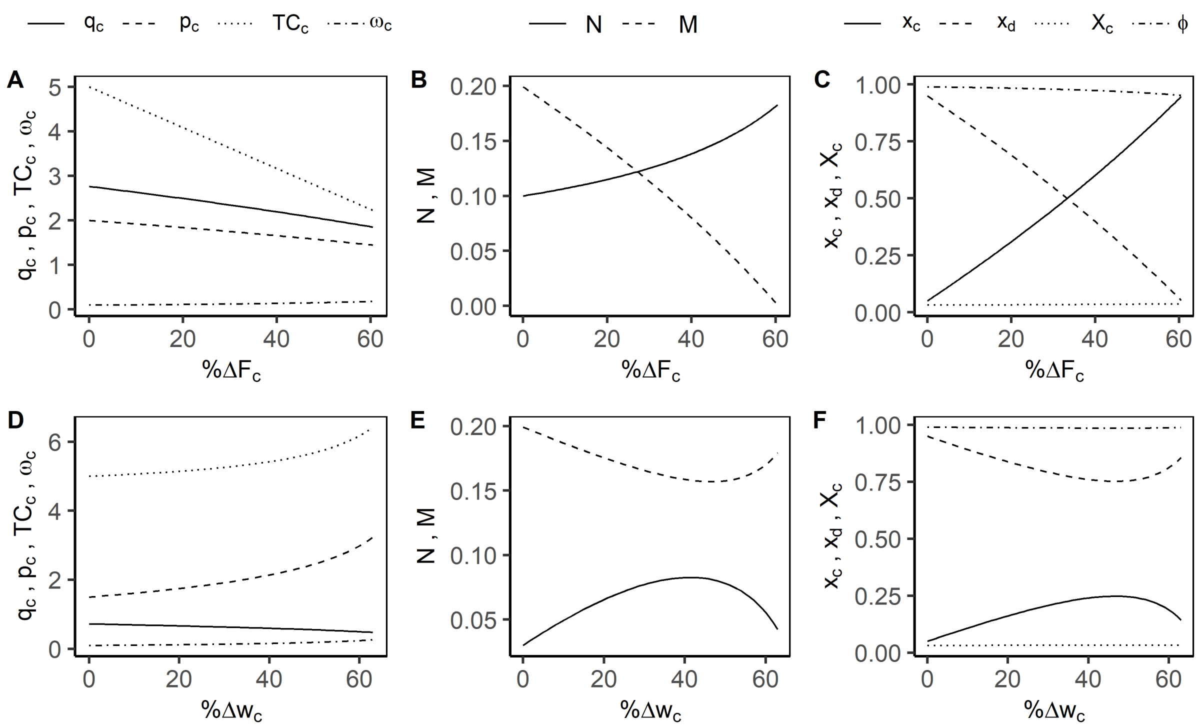
$$X_c = N f q_c - x_c \tag{24}$$

and

$$B_R = \frac{N f q_R - x_R}{{p_R}^{\gamma_R}} \tag{25}$$

with an inelastic demand by clean consumers (-0.5). Note that we allow N to be a positive real number and not an integer since we are more interested in relative changes than in absolute (unobservable) values. Table 1 further summarizes demand elasticities and welfare components, computed ex-post.

Figure 4.1: Effects in endogenous variables of a reduction.



4.4 Results and discussion

The model we developed is used to analyze what-if scenarios in which policies exist in support of innovative bio-based (clean) productions. We report here the results of two scenarios for a relative reduction in F_c or w_c to simulate investment cost subsidies and operating cost subsidies, respectively. The model provides insights into several indicators, including relative changes in total welfare, the share of bio-based consumption, the number of firms, and profitability, allowing for a qualitative and quantitative comparison of the two what-if scenarios. We begin by illustrating how policy scenarios affect the endogenous model variables (Figure 4.1). The scenarios are modeled with an upper limit of a 60-percent reduction in parameters to avoid unfeasible solutions, such as negative consumption quantities (due to equations (14) and (15)).

As designed, both policy instruments affect the production function of the clean producers in the short run. When investment cost (capital) subsidies are provided, fixed costs fall, and more firms produce (i.e., enter the market, Figure 1.B) due to assumed perfect competition with no barriers to entry or exit. Thus, N grows exponentially for linear growth in subsidies that reduce F_c. In the aggregate market, the supply increases and the price falls, narrowing the gap with the selling price of the fossil-based substitute, from 2 to 1.45 (at the 60-percent upper limit of the capital subsidies scenario). Given the new price and the higher number of firms, clean producers produce (q_c) and earn (π_c) less. This, however, does not imply that the profitability ω_c of the bio-based product needs to follow the same path. In fact, the more capital subsidies affect F_c, the more the relative profitability of bio-based production (Figure 1.A) increases, up to 0.18 at the upper limit of the scenario[3]. On the demand side (Figure 1.C), the lower price increases the consumption of bio-based product, especially by the indifferent consumer, since the clean consumer demand is assumed to be inelastic. The bio-based share rises, therefore, from 8 to about 95[4] percent in the extreme scenario, overtaking the fossil-based consumption after a 32.5 percent reduction in F_c. On the fossil-based market segment, demand and supply plummet, affecting mainly the number of firms M with negligible changes in firm's profit, quantity q_d and price p_d.

Compared with capital subsidies, operating cost subsidies affect operating costs. Here, diminishing returns to scale directly affect policy efficiency. With operating cost subsidies, the producer's marginal costs decrease, and consequently, plants will produce more than before. As marginal costs decrease, the price will also decrease, leading to an increase in demand for the bio-based product. This is true for any simulated policy level relative to the baseline condition. However, as the operating cost subsidy increases, marginal productivity decreases. Hence, assuming homogeneous plants, to satisfy the same market quantity, there may be N' firms producing q_c'or fewer plants N'' producing q_c'' with $q''_c > q'_c$. For this phenomenon, the number of firms and the quantity demanded is concave as a function of the operating cost subsidy level, while the selling price is convex. Thus, there is a policy level (different from the upper limit of the simulation) where bio-based share of demand is maximized (i.e., 27 percent of the total production) and bio-based price reaches its minimum (1.87), that is, at 47 percent reduction in w_c. Unlike the capital subsidies, there is no level of the policy instrument (in our model) such that fossil-based demand is overtaken by the bio-based production. However, contrary to the capital subsidies scenario, the profit of bio-based firms increases along with the profitability of the bio-based product.

4.4.1 Welfare effects

In our framework, total welfare is the sum of the (dirty and clean) producer surplus and the clean consumer's surplus, minus the total cost of the indifferent consumer, the cost of externalities, and the costs of the policy scenario.'

Figure 4.2 outlines the various components of welfare gains and losses in the bio-based and fossil-based markets under the simulated policy scenarios. Fossil-based market loses a producer surplus of more than 99 percent in the scenario in which F_c is reduced by

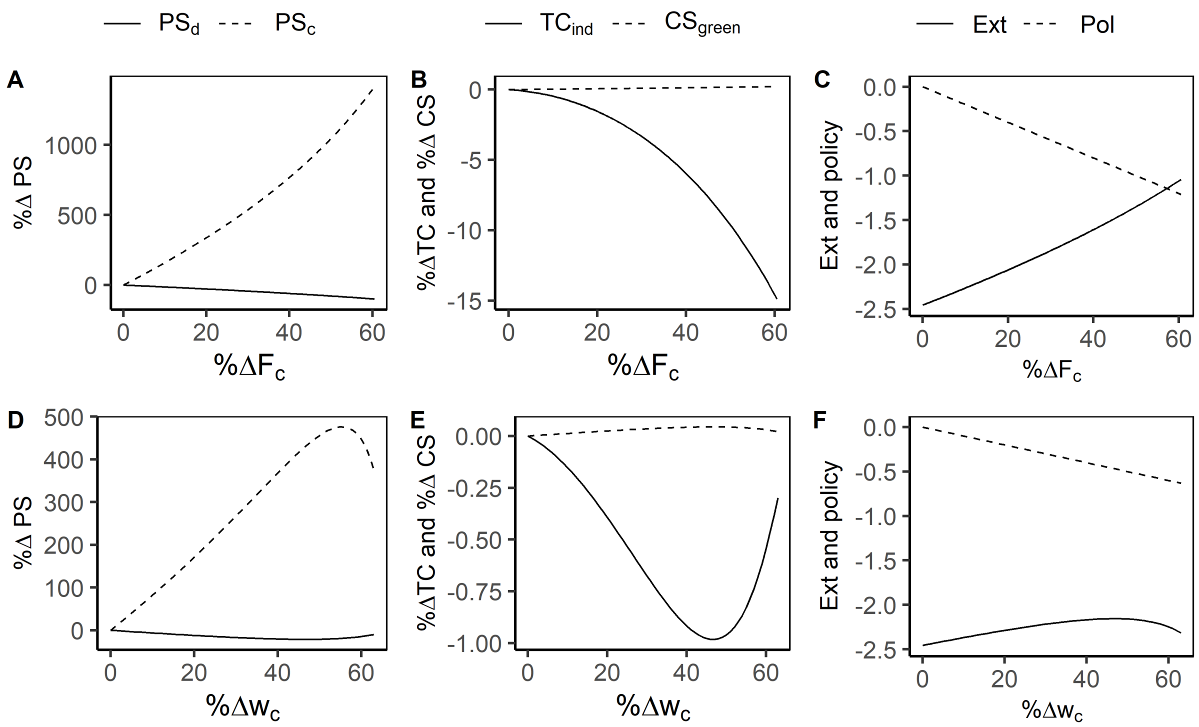
[3]Note that this is conditional to the unit elasticity given by equation (11). With elasticity $\sigma_F > -0.4$ profitability ω_d decreases as fixed costs decrease.

[4]Note that the y-axis in Figure 1C and D can be interpreted directly as a percentage due to the parameterization of $q_d = 1$.

60 percent, while the clean producer's surplus rises to more than 1400 percent. Note that the high percent values for clean production are due to the low baseline values and the high demand elasticities. In the operating cost subsidy scenario (Figure 2D), according to the explanation of the effects in endogenous variables of a reduction in w_c, there is a policy level different from the upper limit of the simulation where welfare components are maximized. The surplus of dirty producers loses at most 21.19 percent from the base level (when the policy scenario reduces by 47 percent the unit price of the other operating costs w_c), while the clean producer surplus rises at most to 476 percent (when the policy scenario reduces by 55 percent w_c). As for (intermediate) consumers, in the capital subsidy scenario (Figure 1B), both types of consumers benefit from falling prices of biobased products. The surplus of green consumers increases by 0.21 percent compared to the baseline scenario, while the total cost of the indifferent consumer (recalling that we did not derive the consumer surplus for the indifferent market segment but used eq.13 to calculate it) decreases by 14.8 percent. The high relative changes in the clean producer surplus compared to the small relative gain in the consumers surplus are again driven by differences in demand elasticities. As for the operating cost subsidy scenario (Figure 2E), compared with the baseline scenario, the green consumer's surplus increases by a maximum of 0.04 percent, while the total cost of the indifferent consumer decreases by a maximum of 0.98 percent (that is, when the policy level reduces the input price of other operating costs by 47 percent). In terms of the cost of the externalities, capital subsidies (Figure 2C) reduce their total cost by 57.5 percent at the upper limit of the simulation, which is when 86 percent of the baseline fossil-based consumption is replaced by bio-based production. Operating cost subsidies (Figure 2F), instead, reduce by 12.2 percent at the 47 percent reduction level, which is when 19.2 percent of fossil-based consumption is replaced. In terms of government losses, as simulated, policies have a negative linear cost, with subsidies costing more than operating cost subsidies for the same percentage reduction.

Combining all welfare components, Figure 4.3 shows the relative changes in total welfare for the simulated policy instruments. In the baseline, we obtained a total welfare value of 4.8. Expected welfare increases in both simulations with a decreasing marginal effect. The capital subsidy improves total welfare to 5.9 (+22.9 percent) at the upper limit of the scenario (60 percent reduction in F_c). Compared with capital subsidy, the operating cost subsidy raises the total baseline welfare at most to 5.1 (+6 percent) when reducing of 48 percent the w_c. In the baseline, subsidizing the fixed cost amortization of a biobased production will have better results in terms of improving total welfare than subsidizing its operating costs. Uncertainty about the welfare effects of the best performing policy levels is studied through Monte Carlo simulations.

Figure 4.2: Changes in welfare components after a reduction policy instruments simulation.



Legend: PS: Producer Surplus; CS: Consumer Surplus; TC: Total Cost of the indifferent consumer; Ext: Cost of externalities; Pol: Cost of the policy

4.4.2 Monte Carlo simulation

To gain further insights into the total welfare improvements of the simulated policy scenarios, we performed a Monte Carlo simulation by changing the baseline values of assumed parameters. Table 2 summarizes the minimum and maximum values of the parameters for which a PERT distribution was assumed in the uncertainty analysis. Supply elasticities were adjusted so that the range of values goes from inelastic values (min = 0.5) to a maximum value that guarantees non-negative profitability (the constant elastic functional form (11) does not allow negative values, that is, $E_i > 0$). The demand elasticity of clean consumers has also been extended to simulate an elastic demand. A variation of ±50 percent is assumed for the remaining parameters, including the difference between initial selling prices, input quantities, fixed costs, and the number of producers. Unitary elasticities of profitability σ_i have not been changed (a limitation of the model made explicit in the conclusion section).

Based on the defined PERT distributions, we ran 10,000 interactions and recalibrated the model, obtaining 4,349 potential baselines (unfeasible iterations reporting negative quantities or a negative number of clean producers were excluded). For each of these feasible 4,349 iterations, we shocked each F_c with a reduction of 60 percent and,

Figure 4.3: Relative changes in total welfare.

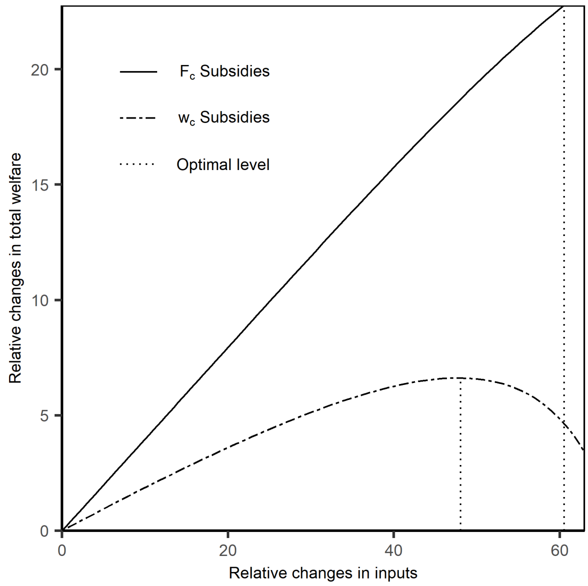


Table 4.2: Range of baseline values for Monte Carlo simulations

Description	Parameter	d min	d max	C min	C man
Supply elasticities	ϵ	0.5	3	0.5	1.5
Selling price	p	-	-	1	3
Feedstock	z	0.75	2.25	0.75	2.25
Others	k	0.75	2.25	0.75	2.25
Fixed cost	F	-	-	1	3
Emissions per BDO	μ	1.25	3.75	-	-
Number of firms	N	-	-	0.015	0.045
Bio-Demand elasticity	γ	-	-	-1.5	-0.5

independently, each w_c with a reduction of 47 percent (simulating the shocks of the best-performing policy level in terms of total welfare observed earlier in Figure 3). Solving the model after shocks, we got 1,885 feasible iterations (out of 4,349) in the case of fixed cost subsidies and 1,848 iterations in the case of operating cost subsidies. Figure 4.4 and the descriptive statistics in Table 3 illustrate the density distributions of the resulting final model solutions (with Figure 4B focusing on the peaks of the distributions).

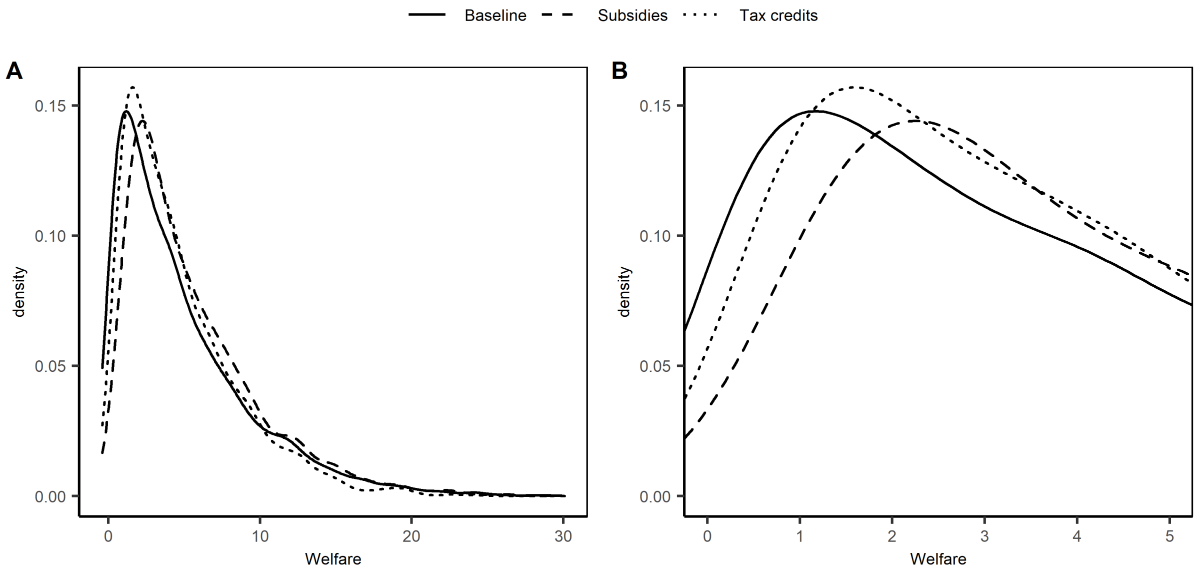
By relaxing some of the model assumptions, the medians of total welfare show lower values than the baseline estimates (5.8 and 5.1 for the case of fixed cost subsidy and operating cost subsidy, respectively). The simulation of capital subsidies continues to perform better (in terms of total welfare improvement) than operating cost subsidies. The median of the former, in fact, shows a growth in total welfare of 20 percent over the median of

Table 4.3: Statistics for the uncertainty of the perturbed baseline values

Indicator	Baselines	Subsidies	Tax credits
Q1	1.49	2.29	1.74
Median	3.57	4.29	3.59
Q3	6.84	7.53	6.43
IQR	5.36	5.25	4.68
CQV	0.64	0.53	0.57
N	4349	1885	1848

Note: Q1, First quartile; Q3, Third quartile; IQR, Interquartile Range; CQV, Coefficient of Quartile Variation.

Figure 4.4: Density distributions.



the 4,349 baselines, while the latter of 0.56 percent. Regarding the distribution of total welfare, the capital subsidies show a larger absolute variance (given by the interquartile range) compared to the operating cost subsidy scenario. However, looking at the relative variance (given by the coefficient of quartile variation), the results of the simulation with the optimal subsidy denote less uncertainty than the operating cost subsidies simulation.

4.5 Conclusions

Global challenges require onerous efforts. The European Union, with the Green Deal, as well as other countries and other development strategies, have bet on bio-based productions to link a prosperous and competitive economy with a healthy planet. But how to invest in clean production so that it can become dominant instead of a niche? In the absence of historical data for econometric analysis, our study provides qualitative and quantitative insights that can support policymakers in their journey through the complexities of transformation to a low-carbon bioeconomy. Notably, we simulated two different scenarios with either investment cost subsidies or operating cost subsidies for innovative bio-based production. Both simulations generate higher total welfare than the baseline scenario, with fixed-cost subsidies performing significantly better. Moreover, in this scenario alone, the clean production market share exceeded that of the dirty (fossil) production, thus making the bio-based product predominant.

The study has several limitations. The first limitation concerns the profitability function (equation 11). The constant elasticity functional form prevents us from modeling negative values, thus generating infeasible solutions in the calibration and in Monte Carlo simulations. Ae solution is to use a linear functional form. Other limitations concern the indifferent consumer demands we derived (equations 14 and 15). Because they are linear, they can turn to negative values generating infeasible solutions (negative demand quantities or negative numbers of firms). Furthermore, the own-price demand elasticity derived for the clean market segment makes the model highly sensitive, preventing the simulation of all possible scenarios (such as a 100 percent reduction in fixed costs). Finally, other policy instruments should be studied for a more in-depth comparison of what-ifs scenarios, such as feedstock costs subsidies (i.e., shocking v_c).

References

Asada, R. and Stern, T. (2018) ‘Competitive Bioeconomy? Comparing Bio-based and Non-bio-based Primary Sectors of the World’, Ecological Economics. Elsevier, 149(March), pp. 120–128. doi: 10.1016/j.ecolecon.2018.03.014. Business analytiq (2022): Available at: https://businessanalytiq.com/procurementanalytics/index/14-butanediol-14-bdo-prices/

Chauvel, A. and Lefebvre, G. (1989) ‘Monomers for polyester synthesis’, in Petrochemical Processes. Major Oxygenated, Chlorinated and Nitrated Derivatives. Paris: Publications Ifp, p. 328.

Clancy, M. S., Moschini, G. (2018). Mandates and the incentive for environmental innovation. American journal of agricultural economics, 100(1), 198-219.

Cukalovic, A. and Stevens, C. V. (2008) ‘Feasibility of production methods for succinic acid derivatives: a marriage of renewable resources and chemical technology’, Biofuels, Bioproducts and Biorefining, 2.6, pp. 505–529.

De Bari, I. et al. (2020) ‘From cardoon lignocellulosic biomass to bio-1,4 butanediol: An integrated biorefinery model’, Processes, 8(12), pp. 1–18. doi: 10.3390/pr8121585.

De Jong, E. et al. (2012) ‘Product developments in the bio-based chemicals arena’, Biofuels, Bioproducts and Biorefining, 6(6), pp. 606–624. doi: 10.1002/bbb.1360.

De Gorter, H., Just, D. R. (2009). The economics of a blend mandate for biofuels. American Journal of Agricultural Economics, 91(3), 738-750.

Dell'Erba, M. (2021). Sustainable Digital Finance and the Pursuit of Environmental Sustainability. In Sustainable Finance in Europe (pp. 61-81). Palgrave Macmillan, Cham.

DSM Engineering Plastics (2013) 'Achieving higher bio-based content in DSM Arnitel® copolyesters: Results from testing 1,4-butanediol (BDO) made with Genomatica's process technology.'

Eurobarometer & European Commission (2008) 'Attitudes of European citizens towards the environment (Special Eurobarometer 295)', (March).

European Commission (2018) A sustainable Bioeconomy for Europe: strengthening the connection between economy, society and the environment. doi: 10.2777/478385.

European Commission (2019) 'The European Green Deal', European Commission, 53(9), p. 24. doi: 10.1017/CBO9781107415324.004.

European Commission (2020) Sustainable Europe Investment Plan.

Eurostat (2022) Total production by PRODCOM list. Available at: https://ec.europa.eu/eurostat/web/prodcom/data/database.

Forte, A. et al. (2016) 'LCA of 1,4-butanediol produced via direct fermentation of sugars from wheat straw feedstock within a territorial biorefinery', Materials, 9(7), pp. 1–22. doi: 10.3390/MA9070563.

Genomatica (2016) 'Novamont opens world ' s first commercial plant for bio- production of a major intermediate chemical', Globe News Wire, pp. 1–3. Grand View Research (2022) Available at: https://www.grandviewresearch.com/industry-analysis/1-4-butanediol-market

IRENA (2022), Bioenergy for the energy transition: Ensuring sustainability and overcoming barriers, International Renewable Energy Agency, Abu Dhabi.

Jin, Y. and Drabik, D. (2022) 'CRISPR Rice vs conventional rice dilemma of a Chinese farmer*', Australian Journal of Agricultural and Resource Economics, 66(2), pp. 424–446. doi: 10.1111/1467-8489.12465.

Jopp, K. (2022) BDO: The Pacemaker Stumbles. Available at: https://www.chemanager-online.com/en/topics/chemicals-distribution/bdo-pacemaker-stumbles.

Jouvet, P. A., Le Cadre, E., Orset, C. (2012). Irreversible investment, uncertainty, and ambiguity: The case of bioenergy sector. Energy Economics, 34(1), 45-53.

Kircher, M. (2015). Sustainability of biofuels and renewable chemicals production from biomass. Current opinion in chemical biology, 29, 26-31.

Li, K. and Chen, K. (2019) 'Cyclodehydration of 1,4-butanediol over Zr-Al Catalysts: Effect of Reaction Medium', Materials.

Morone, P. et al. (2021) 'Consumer willingness to pay for bio-based products: Do certifications matter?', International Journal of Production Economics. Elsevier B.V., 240(July), p. 108248. doi: 10.1016/j.ijpe.2021.108248.

Newes, E., Inman, D., Bush, B. (2011). Understanding the developing cellulosic biofuels industry through dynamic modeling. Economic Effects of Biofuel Production, 373-404.

Pääkkönen, A., Tolvanen, H. and Kokko, L. (2019) 'The economics of renewable CaC2 and C2H2 production from biomass and CaO', Biomass and Bioenergy. Elsevier Ltd, 120(August 2018), pp. 40–48. doi: 10.1016/j.biombioe.2018.10.020.

Reinders, M. J., Onwezen, M. C. and Meeusen, M. J. G. (2017) 'Can bio-based attributes upgrade a brand? How partial and full use of bio-based materials affects the purchase intention of brands', Journal of Cleaner Production. Elsevier Ltd, 162, pp. 1169–1179. doi: 10.1016/j.jclepro.2017.06.126.

Satam, C. C., Daub, M. and Realff, M. J. (2019) 'Techno-economic analysis of 1,4-butanediol production by a single-step bioconversion process', Biofuels, Bioproducts and Biorefining, 13(5), pp. 1261–1273. doi: 10.1002/bbb.2016.

Satam, C. C. and Realff, M. J. (2020) ' Comparison of two routes for the bio-based production of economically important C 4 streams ', Journal of Advanced Manufacturing and Processing, 2(3), pp. 1–12. doi: 10.1002/amp2.10054.

Silva, R. G. C., Ferreira, T. F. and Borges, É. R. (2020) 'Identification of potential technologies for 1,4-Butanediol production using prospecting methodology', Journal of Chemical Technology and Biotechnology, 95(12), pp. 3057–3070. doi: 10.1002/jctb.6518.

Spekreijse, J. et al. (2019) Insights into the European market for bio-based chemicals, European Commission. doi: 10.2760/673071.

Taxonomy Regulation: Regulation (EU) 2020/852 of the European Parliament and of the Council of 18 June 2020 on the establishment of a framework to facilitate sustainable investment and amending Regulation (EU) 2019/2088.

Taylor, R. et al. (2015) From the Sugar Platform to biofuels and biochemicals: Final report for the European Commission Directorate-General Energy. Available at: https://ec.europa.eu/energy/sites/ener/files/documents/EC Sugar Platform final report.pdf.

Vanhamaki, S. et al. (2019) 'Bio-based circular economy in European national and regional strategies', International Journal of Sustainable Development and Planning, 14(1), pp. 31–43. doi: 10.2495/SDP-V14-N1-31-43.

Von der Leyen, U. (2019) Press remarks by President von der Leyen on the occasion of the adoption of the European Green Deal Communication.

Webster, I. (2022) Value of $1 from 2019 to 1989. Available at: https://www.officialdata.org/2018-dollars-in-1989?amount=1: :text=%241 in 2018 is equivalent,price increase of -50.64%25.

Conclusion

Synthesis

In this thesis, I have evaluated alternative pathways for sustainable innovation in the field of the bioeconomy. Empirical studies have been performed using the case study research method, which is *"the conventional way of doing process or implementation evaluations"* (Yin, 2014; p.222), especially for complex contemporary context-dependent phenomena. The bioeconomy is a complex set of bio-based systems that rely on new knowledge in the biological field to produce new products (Zilberman, 2019). It aims to foster productive harmony between humans and nature, i.e. meet the social, economic, and environmental needs of present and future generations (Fitzgerald, 2017). Despite the clear promise of the bioeconomy, different sustainability models have been proposed to achieve it (Vivien et al., 2019). This is reflected by the different narratives and case study approaches in the field of bioeconomy, as demonstrated in Chapter 1 (Tassinari et al., 2021). Below, I briefly summarize the responses to the four research questions (RQs) of my thesis.

RQ.1: How do different visions of the bioeconomy influence case study approaches?

Chapter 1 distinguishes between the three main interpretations of the bioeconomy. The ecological-economic interpretation argues for a sustainability model dictated by strict ecological constraints. In this area, empirical observation through case studies is conducted primarily to quantitatively explore the environmental effects (e.g., land use and land cover changes) associated with a bio-based system, supporting a global and interdisciplinary perspective and providing policymakers with reference standards for ecological compensation. The interpretation of the bioeconomy as an industrial revolution limits the analysis to the exploration of biotechnologies that are applied in specific contexts to gather evidence, mainly qualitative, of their adoption in support of wider diffusion. The third and most common interpretation of the bioeconomy, a renewable carbon economy based on biomass transformation, gathers empirical, mainly qualitative, evidence to explore the potential properties, opportunities, and obstacles associated with biomass sources (rather than a biotechnology application), emphasizing knowledge creation, entrepreneurial experimentation, and market formation. Common to all bioeconomy interpretations is the central role played by both context and agri-food systems. Based on the experience and knowledge discussed in Chapter 1, Chapter 2, 3, and 4 report three case studies to explore key aspects of relevant contemporary sectors in the bioeconomy.

RQ.2: Do biodegradable food packaging films from agro-food waste pay off?

Given the urgent need to redesign plastics to achieve more sustainable production and consumption processes, Chapter 2 presented the ex ante economic evaluation of a new biorefinery concept related to food packaging. Specifically, we studied, at different plant scales, the minimum selling price of a polylactide (PLA)/polyhydroxybutyrate (PHB) biofilm produced from potato peels. Considering the available quantity of this agri-food by-product in Europe, the potential total production capacity of the biofilm could increase European bioplastic production by almost 25%. The conditions for concrete market development, however, remain complicated. The baseline scenario showed that the threshold price of biofilm for a break-even point of four years ranges from 9.7 to 37.2 euros per kg depending on the biorefinery plant size. Large or small plants perform economically better than medium-sized plants given their lower resource competition. Considering the current price level for fossil-based plastics of about 1 to 1.5 euros per kg, bio-based production is unlikely to be economically competitive and attractive to industries. Therefore, further development is needed, mostly to increase conversion efficiency at low fixed costs. Finally, engineered biotechnology should benefit from extension to other feedstocks while considering biomass availability the most influential factor for profitability, as demonstrated with the Monte Carlo uncertainty analysis.

RQ.3: Recycling or landfilling? What are the differences in their sustainability footprint performances?

In contrast with the previous chapter, Chapter 3 analyzes the overall sustainability performance of a mature bio-based production. Given the rigidity of demand and the volatility of chemical fertilizer prices, which are also subject to contemporary geopolitical tensions (such as the Russia–Ukraine conflict), we were interested in studying the potential bio-based alternatives in this field. Several studies have illustrated the agronomic benefits of recycling sewage sludge, which is usually used as landfill. We demonstrated the differences in sustainability footprints when sewage sludge is recycled and when it is used for landfill. The results compare the total economic, social, and environmental impacts of the final demand for service. The study demonstrated a trade-off between the different sustainability dimensions, with the recycling strategy performing better socially (+5.6% job creation than landfilling) and environmentally (-68.2% GHG emissions).

On the other hand, landfilling performs better economically (+17.3% economic stimulus than recycling) and in terms of energy (-36.8% energy consumption). Considering the uncertainty of estimates, we measured the probability that these statements are contradictory. Processing the same amount of sewage sludge, there is a probability of 0.05% that the recycling strategy generates more greenhouse gas emissions than landfilling, of 1.19%

that it has lower energy consumption, of 24.05% that it stimulates the economy more, and of 37.62% that it generates fewer jobs. Moreover, using a multi-regional input–output analysis, we demonstrated (i) the importance of indirect impacts in sustainability performance assessments (notably, in energy consumption), (ii) the context dependency of the production process, and (iii) systematicity as a key element for easier and more efficient comparisons between sustainability dimensions.

RQ.4: How well do policies—investment cost subsidies and operating cost subsidies—perform as incentives for stimulating a higher market share for renewable bio-based input alternatives?

While the second and third chapters dealt with the sustainability performance of a product or process, Chapter 4 evaluated an ex ante what-if policy scenario supporting a transition to a low-carbon bioeconomy. In the absence of historical data, we developed a partial equilibrium model. The policy scenarios were simulated using the 1,4-BDO market as an instrumental case study, for which a drop-in bio-based alternative exists. Both policy simulations generate higher total welfare than the baseline scenario (without policy), with investment cost subsidies performing better than operating cost (excluding feedstock costs) subsidies. In the case of the 1,4-BDO market, investment cost subsidies generate a maximum total welfare that is higher than that achieved by the operating cost subsidies. Moreover, only in the investment-cost-subsidies scenario, there are policy levels (with a fixed costs reduction of 32.5% or more) in which the fossil-based market segment is overtaken by the bio-based one. Finally, the Monte Carlo simulation shows less uncertain results for the investment cost subsidies than the second policy. Overall, despite the lack of historical data, the study provides important qualitative and quantitative insights that could be useful to policymakers for a transition to a low-carbon bioeconomy.

Comparison

Empirical evidence from case studies shows that the bioeconomy responds pragmatically to contemporary global environmental, social, and economic challenges. Regardless of whether they are innovative or mature, bio-based systems need better assessments to facilitate their adoption or diffusion, thus making the bioeconomy a fertile topic for research and development. All decision-making associated with a transition to a low-carbon bioeconomy should be based on the best scientific evidence and knowledge. A better systematization of case studies in the bioeconomy domain would facilitate such scientific progress by supporting the transparency, replicability, and comparability of empirical evidence collection.

The combination of case studies in this thesis demonstrates the complexity and heterogeneity of bio-based systems. Bio-based productions can be characterized by different degrees of technological maturity based on the different types of feedstocks, and each has

Table 1: Attributes of the case studies in this thesis

Attribute	Chapter 2	Chapter 3	Chapter 4
Bioeconomy vision	Biomass-based economy	Biomass-based economy	Biotechnology
Production	Food-packaging polymers	(Bio-based) Fertilizers	(Drop-in bio-based) Chemicals
TRL	6–7	9	9
Feedstock	By-product (potato peels)	Organic waste (sludge)	Primary (sugar crops)
Case	Profitability of a new food-packaging biofilm	Sustainability performances of a mature recycling strategy	Efficiency of policy instruments for bio-based industries
Unit of analysis	Product-based	Process-based	Policy-based
Context	International perspective	International perspective	International perspective
Data sources	Lab-based	Company-based	Literature-based
Data analysis	Ex ante cost–benefit analysis (and Monte Carlo simulations)	Ex post multi-regional input–output analysis (and Monte Carlo simulation)	Ex ante policy assessment with a partial equilibrium (and Monte Carlo simulation)
Indicators as results	Minimum product selling price	Economic stimulus, employment, GHG emissions, energy carriers' consumption	Bio-based market share, total welfare

its own opportunities, limitations, and requirements. By definition, all of them suffer from pressing competition with fossil-based industries. The latter can benefit from and capitalize on decades of research and development that maximizes their production efficiency. In contrast, the innovative side of the bioeconomy suffers from a lack of (i) a comprehensive official statistical database, (ii) a transparent methodology for data collection, and (iii) integrated data and indicators in the value chain (Kardung and Wesseler, 2019). This thesis presents three methodological ways to address more systematic and holistic approaches to analyzing sustainability performance. Table 1 summarizes the main features of these empirical studies.

The development of a bioeconomy that is as sophisticated and efficient as the fossil fuel-based one requires targeted initiatives (as shown in Chapter 2). Due to the low TRL, researchers might not have information about the exact industrial configuration of bio-based processes. Moreover, plant cost data are sensitive and rarely shared by companies. Reviewing the literature, we found that economic feasibility studies for innovative bio-based projects are based on pilot-scale data that are later extended to the most cost-effective commercial level (e.g., a large production scale with a long payback period). However, sustainability could be achieved sooner if techno-economic studies, in collaboration with industries, explored multiple nonlinear developments of the possible investment scenarios. Chapter 2 is based on this conceptual framework and evaluates a new biorefinery concept according to feedstock availability for different industrial scales and logistical settings.

When technological maturity is achieved (i.e., high TRL), other issues may come to force in sustainability assessments. Notably, the decision regarding the boundary of the system to be examined is left to the expert's discretion, leading to truncation errors that limit the robustness of the analysis (Crawford et al., 2018; Ward et al., 2018). Chapter 3 addressed this challenge by systematically quantifying the economic, social, and environmental impacts of bio-based production with a multiregional input–output model that can capture an infinite order of contributions from upstream production processes (Lenzen, 2000).

Due to the lack of data, the Monte Carlo method has proven to be a valuable tool in model uncertainty assessments for the bioeconomy. Monte Carlo simulations play a prominent role in stochastic simulations by generating a random sample of the model utilized. This technique makes it possible to obtain an estimate of the entire probability distribution of the indicators and significantly reduces the time and resources that would have been required for data collection (which is sometimes impossible). Moreover, by combining uncertainty analysis with regression analysis, estimates of the effects of various inputs can be obtained, i.e. which factors contribute the most to the uncertainty of the indicator can be demonstrated on the basis of n combinations of inputs and the corresponding outputs.

One common aspect that emerged from all case studies concerns the ambiguity of the European policy framework. We have seen how policy ambiguity hurts investment in bio-based production. This is the case with the European Directive on single-use plastics (Chapter 2), which came into force on July 3, 2021 and banned the use of synthetic polymers (including biopolymers such as PLA and PHB) for single-use food packaging. The definition of "biopolymers" and "single use," however, had varying interpretations—to the extent that individual Member States adopted divergent rules. For instance, only 11 of the 27 EU countries have enforced expanded polystyrene bans so far, and Italy has exempted biodegradable single-use plastics, sparking a legal dispute with the EC (Gore-Langton, 2022). A similar ambiguity can also be seen in the Taxonomy Regulation, which regulates common sustainability criteria to define sustainable investments but leaves undefined the common methodologies that can be used for their measurement. A third example is presented in Chapter 3; we have seen that the European Directive 278/1986 governing the use of sewage sludge in agriculture does not fully match the contemporary needs and expectations, as stated by the EC itself (EC, 2022). The EC is thus considering reviewing the Sewage Sludge Directive as part of the development of an Integrated Nutrient Management Plan in the New Circular Economy Action Plan (EC, 2020). As a result, given the uncertainty of the future regulatory framework, the ambiguity of the institutional policy framework disincentivizes investment in innovative bio-based production, which is already suffering from higher investment costs compared to conventional production.

Limitations and future research

The generalizability of these results is subject to certain limitations. In addition to the limitations mentioned in each study, I want to point out a limiting aspect common to all chapters that, if addressed, could lead to important research developments. All the studies considered in this thesis are static. The analysis in Chapter 1 is limited to a sample of case studies selected over a single year. This choice was necessary to limit the number of cases but without excluding variety, which is the research focus of the chapter. The study provides an initial set of recommendations for case study research in bioeconomics. However, it cannot identify trends and patterns in use that would serve *"as an instrument to reflect, as a research community, on our progress"* (Dubé and Paré, 2003, p.599).

A cost–benefit analysis is the process used in Chapter 2 to measure the benefits of an investment in a bio-based product. Our approach ignores the strategic value of delaying immediate action in decision-making. In a dynamic framework with future learning, what matters is the uncertainties, the extent to which the benefits and costs are irreversible, and the existence of opportunities to postpone investment changes (Dixit and Pindyck, 1994). The real-options theory is the methodological approach employed to evaluate such decision-making processes under uncertainty, learning, irreversibility, and the ability to delay actions (Wesseler and Zhao, 2019). Investors might be deterred from investing in a bio-based product not only because of companies failing to achieve satisfactory net present value but also because of ambiguity in policies, as mentioned earlier. The real-option theory can offer important insights for investments in bioeconomies by assessing investment risks created by policy ambiguity.

Chapter 3 and 4 also use static models. Chapter 3 studies sustainability footprints based on a Leontief input–output model. Input-output (IO) analysis has become a popular tool for performing various types of consumption-based impact assessments (Södersten and Lenzen, 2020). The model we have presented is based on a one-year input–output computation, which means that it cannot be used as a forecasting tool because the technological coefficients would remain unchanged over the forecast period. However, dynamic input–output models can help relax the strong assumption of fixed production technology and can be used to explore different policy scenarios over time, estimate future capacity requirements, and estimate projections of structural and technological changes (Södersten and Lenzen, 2020).

Finally, in Chapter 4, we developed a static partial equilibrium model. This means that it does not predict the speed of the adjustment of prices and quantities after a policy change (Hallren and Riker, 2017). Good reasons to introduce dynamics into this framework would be modelling technological progress or exogenous changes in demand.

Contributions

Global challenges such as climate change, food security, or energy security urgently require solutions (Lund Declaration, 2015), which will, in turn, require onerous investments (EC, 2019). Due to the intention of supporting sustainable development, environmental, social, and governance criteria have become increasingly important for companies, investors, and financial markets. On April 29 2022, the EU published the initial requirements for standardized reporting in the draft European sustainability reporting standards (ESRS) with the Corporate Sustainability Reporting Directive (CSRD). The CSRD is a key element of the EU sustainable finance package. The EC anticipates that this proposal can transform the company reporting ecosystem to improve the quality and consistency of sustainability information. With this decision, the CSRD will gradually extend to more than 50,000 companies (Holder, 2022), each of which will represent a source of empirical evidence as well as a case study. However, CSRD has been criticized by financial markets, consumer associations, and consultancy agencies for being too superficial and inconsistent with current international standards such as the International Financial Reporting Standards (IFRS) and Sustainability Disclosure Standards (SDS). Although progress is being made in defining the common criteria by which to determine whether an economic activity qualifies as sustainable (Taxonomy Regulation), methods for measuring these sustainability criteria remain to be established.

The methodologies adopted in this thesis provide holistic and systematic approaches for assessing sustainability performance. Gathering empirical evidence through these tools can have an impact in the following spheres: (i) environmental, with more companies becoming capable of comparing eco-performance and identifying inefficiencies; (ii) economic, with more comprehensive information for investments in sustainable activities; and (iii) political, to assess ex ante the policy scenarios for achieving strategic sustainable goals. In addition, industries can benefit from relevant sustainability indicators for communicating with consumers, citizens, and policymakers about the associated activities and possible legislative ambiguities. Close collaboration between universities, industries, and policymakers (thus increasing the intensity of interdisciplinary research) is crucial for standardization in either the reporting or the collection and analysis of primary data. Primary data will generate valid and comparable empirical evidence and the secondary data required for the bioeconomy innovation and development process.

The findings from this thesis represent several contributions to the current academic literature. The current body of research on the case study method suggests that systematization is a key factor in overcoming the concerns related to lack of trust in the credibility, validity, and generalizability of the procedures employed in a case study research (Dubé and Paré, 2003; Barratt, Choi and Li, 2011; Yin, 2014)

This thesis contributes to case study systematization in the field of bioeconomy by describing the heterogeneity and complexity of case studies in the field and providing important food for thought with regard to the development of standardized protocols (Chapter 1). In the field of cost–benefit analysis, to date, there has been little agreement regarding which industrial scale can be utilized to study new technologies (Cristóbal et al., 2018). The choice is limited to the most cost-effective level, compromising the robustness and comparability of the information provided. In this thesis, we have demonstrated the advantages of modeling multiple nonlinear scenarios in combination with a Monte Carlo analysis (Chapter 2). There is also growing literature that recognizes the problem of truncation errors in most common life cycle assessment approaches (Lenzen, 2000; Ward et al., 2018). To this end, this thesis aims to contribute to the growing research area of multiregional input–output analysis for sustainability assessments (Lenzen et al., 2018; Zhang et al., 2019; Wei et al., 2021) by exploring the efficiency of the method coupled with a Monte Carlo analysis for comparing the uncertainty of performance estimates (Chapter 3). Finally, a substantial number of studies have been published on ex ante policy evaluations for bio-based production (de Gorter and Just, 2009b, 2009a; Clancy and Moschini, 2018), and most of them focus on biofuels or GMOs. What remains unclear is the impact of policies on drop-in biochemicals while assuming a decreasing return to scale, as we assessed previously (Chapter 4).

References

Barratt, M., Choi, T. Y. and Li, M. (2011) 'Qualitative case studies in operations management: Trends, research outcomes, and future research implications', Journal of Operations Management. Elsevier B.V., 29(4), pp. 329–342. doi: 10.1016/j.jom.2010.06.002.

Clancy, M. S. and Moschini, G. (2018) 'Mandates and the incentive for environmental innovation', American Journal of Agricultural Economics, 100(1), pp. 198–219. doi: 10.1093/ajae/aax051.

Crawford, R. H. et al. (2018) 'Hybrid life cycle inventory methods – A review', Journal of Cleaner Production, 172, pp. 1273–1288. doi: 10.1016/j.jclepro.2017.10.176.

Cristóbal, J. et al. (2018) 'Techno-economic and profitability analysis of food waste biorefineries at European level', Bioresource Technology, 259(January), pp. 244–252. doi: 10.1016/j.biortech.2018.03.016.

Dixit, A. and Pindyck, R. (1994) Investment Under Uncertainty. Princeton, NJ: Princeton Univ. Press.

Dubé, L. and Paré, G. (2003) 'Rigor in Information Systems Positivist Case Research: Current Practices, Trends, and Recommendations', MIS Quarterly, 27(4), pp. 597–636.

EC (2019) 'The European Green Deal', 53(9), p. 24. doi: 10.1017/CBO9781107415324.004.

EC (2020) A new Circular Economy Action Plan For a cleaner and more competitive Europe. Available at: https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:52020DC0098&from=ES (Accessed: 20 December 2022).

Fitzgerald, N. D. (2017) 'Chemistry challenges to enable a sustainable bioeconomy', Nature Reviews Chemistry, 1(10). doi: 10.1038/s41570-017-0080.

Gore-Langton, L. (2022) Single market turbulence: European Plastics Converters opposes Single Use Plastics Directive ambiguity. Available at: https://www.packaginginsights.com/news/single-market-turbulence-european-plastics-converters-opposes-single-use-plastics-directive-ambiguity.html (Accessed: 20 December 2022).

de Gorter, H. and Just, D. R. (2009a) 'The economics of a blend mandate for biofuels', American Journal of Agricultural Economics, 91(3), pp. 738–750. doi: 10.1111/j.1467-8276.2009.01275.x.

de Gorter, H. and Just, D. R. (2009b) 'The welfare economics of a biofuel tax credit and the interaction effects with price contingent farm subsidies', American Journal of Agricultural Economics, 91(2), pp. 477–488. doi: 10.1111/j.1467-8276.2008.01190.x.

Hallren, R. and Riker, D. (2017) 'An introduction to partial equilibrium modeling of trade policy.', Economics Working Paper Series, pp. 1–25.

Holder, E. (2022) Flash Report — Corporate Sustainability Reporting Directive (CSRD) Approved by European Council; More Than 50,000 Organizations Affected. Available at: https://blog.protiviti.com/2022/12/01/flash-report-corporate-sustainability-reporting-directive-csrd-approved-by-european-council-more-than-50000-organizations-affected/ (Accessed: 20 December 2022).

Kardung, M. and Wesseler, J. (2019) 'EU Bio-Based Economy Strategy', in EU Bioeconomy Economics and Policies: Volume II, pp. 277–292.

Lenzen, M. (2000) 'Errors in conventional and input-output-based life-cycle inventories', Journal of Industrial Ecology, 4(4), pp. 127–148. doi: 10.1162/10881980052541981.

Lenzen, M. et al. (2018) 'The carbon footprint of global tourism - Supplementary Information', Nature Climate Change, 8(6), pp. 522–528. Available at: https://doi.org/10.1038/s41558-018-0141-x.

Lund Declaration (2015) 'Europe must speed up solutions to tackel grand challenges through alignment, research, global cooperation and achieving impact'.

Södersten, C. J. H. and Lenzen, M. (2020) 'A supply-use approach to capital endogenization in input–output analysis', Economic Systems Research, pp. 451–475. doi: 10.1080/09535314.2020.1784852.

Tassinari, G. et al. (2021) 'Case studies research in the bioeconomy: A systematic literature review', Agricultural Economics (Czech Republic), 67(7), pp. 286–303. doi: 10.17221/21/2021-AGRICECON.

Tassinari, G. et al. (2023) 'Do biodegradable food packaging films from agro-food waste pay off? A cost-benefit analysis in the context of Europe', Science of the Total Environment, 856(September 2022). doi: 10.1016/j.scitotenv.2022.159101.

Vivien, F. D. et al. (2019) 'The Hijacking of the Bioeconomy', Ecological Economics. Elsevier, 159(January), pp. 189–197. doi: 10.1016/j.ecolecon.2019.01.027.

Ward, H. et al. (2018) 'Truncation Error Estimates in Process Life Cycle Assessment Using Input-Output Analysis', Journal of Industrial Ecology, 22(5), pp. 1080–1091. doi: 10.1111/jiec.12655.

Wei, W. et al. (2021) 'Embodied greenhouse gas emissions from building China's large-scale power transmission infrastructure', Nature Sustainability. Springer US, 4(8), pp. 739–747. doi: 10.1038/s41893-021-00704-8.

Wesseler, J. and Zhao, J. (2019) 'Real Options and Environmental Policies: The Good, the Bad, and the Ugly', Annual Review of Resource Economics, 11, pp. 43–58. doi: 10.1146/annurev-resource-100518-094140.

Yin, R. K. (2014) Case study research: Design and methods. fifth edit. SAGE publications. doi: 10.1016/0149-7189(86)90052-2.

Zhang, H. et al. (2019) 'Tracing the Uncertain Chinese Mercury Footprint within the Global Supply Chain Using a Stochastic, Nested Input-Output Model', Environmental Science and Technology, 53(12), pp. 6814–6823. doi: 10.1021/acs.est.8b06373.

Zilberman, D. (2019) 'Agricultural economics as a poster child of applied economics: Big data big issues', American Journal of Agricultural Economics, 101(2), pp. 353–364. doi: 10.1093/ajae/aay101.

About the author

Gianmaria Tassinari is a dedicated researcher in the field of circular bioeconomy, hailing from Montebelluna, Treviso (Italy). He completed his bachelor's degree in animal production and his master's degree in agricultural technology and economics at the University of Bologna, where he gained a comprehensive understanding of macro and microeconomic aspects related to agriculture and environmental sciences.

While pursuing his academic studies, Gianmaria worked as a cattle dairy farmer, which provided him with practical experience and knowledge of the agricultural sector. He also completed an internship at Teagasc, Galway, Ireland.

Subsequently, Gianmaria pursued his PhD at the Agricultural Economics and Rural Policy Group of Wageningen University in the Netherlands, where he conducted innovative research on the evaluation of sustainability aspects of circular bioeconomy value chains for the H2020 BioMonitor project. His research involved employing various methodologies such as multi-regional input-output models, partial equilibrium models, mathematical programming, empirical analyses, and case studies.

Currently, Gianmaria holds a postdoctoral position at the Università Cattolica del Sacro Cuore, Cremona (Italy), where he continues to contribute to cutting-edge research in his field of expertise.

WASS Completed Training and Supervision Plan

Gianmaria Tassinari
Wageningen School of Social Sciences (WASS)
Completed Training and Supervision Plan

Wageningen School of Social Sciences

Name of the learning activity	Department/Institute	Year	ECTS*
A) Project related competences			
A1 Managing a research project			
WASS Introduction Course	WASS	2019	1
Writing the research proposal	WUR	2019	6
Efficient Writing Strategies	Wageningen in'to Languages	2019	0.8
Scientific Writing	Wageningen in'to Languages	2019	1.2
'Case Studies Dealing With The Bioeconomy in Social Sciences: State Of The Art And Challenges'	23rd ICABR Conference, Ravello	2019	0.5
'The Socio-economic Impacts of Wastewater Sludge Valorisation: The Case of Biofertilizer in Italy'	16th EAAE Congress, online	2020	0.5
'Economic and Social Performance of New Bio-based Industries: The Case of 1,4-butanediol Biorefining in Italy'	25th ICABR Conference, Ravello	2021	1
'Emerging Bio-based Alternative: From Innovation Niches to Dominating Products?'	26th ICABR Conference, Bologna	2022	1
'Assessing Impacts of Bio-Based Projects Using Hybrid Input-Output Approach in Conjunction with Monte Carlo Simulation: The Case of Nutrient Recovery from Sewage Sludge'	NC1034 Research Conference: Nutritional security and resilience through ag innovation, California	2022	1
A2 Integrating research in the corresponding discipline			
Advanced Microeconomics, UEC-51806	WUR	2018	6
Advanced Econometrics, YSS-34306	WUR	2018	6
Assessing Economics and Policies Using the Real Options Methodology	WASS	2019	3
B) General research related competences			
B1 Placing research in a broader scientific context			
Advanced Course on Economic Regulation	WASS	2019	2
Economic modelling of the bioeconomy	WASS	2019	3
Introduction to programming in R for social sciences	WASS	2019	5
B2 Placing research in a societal context			
Tools, methodologies and lessons learned from European projects for the promotion of the bioeconomy	Costa Rica (Online Webinar)	2022	1
Insights from the Bioeconomy Case Studies	Final BioMonitor Meeting with Stakeholders, Brussels	2022	1

C) Career related competences/personal development			
C1 Employing transferable skills in different domains/careers			
Presenting with Impact	WGS	2019	1
Total			**42**

*One credit according to ECTS is on average equivalent to 28 hours of study load

Financial support from Wageningen University, the G. Schieter Foundation and Biomoney for printing this thesis is gratefully acknowledged.

Cover design by Gianmaria Tassinari
Printed by Proefschriftmaken on FSC-certified paper