A COMPARATIVE LIFE CYCLE ASSESSMENT (LCA) BASED BIODIVERSITY IMPACT ASSESSMENT OF HIGH-DENSITY POLYETHYLENE

Comparison of end-of-life treatment options: incineration and landfill, mechanical recycling, and chemical recycling

Lappeenranta–Lahti University of Technology LUT

Master's thesis in Sustainability Science and Solutions

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Examiners: Professor, D.Sc. (Tech.) Ville Uusitalo

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ABSTRACT

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A comparative life cycle assessment (LCA) based biodiversity impact assessment of high-density polyethylene - Comparison of end-of-life treatment options: incineration and landfill, mechanical recycling, and chemical recycling

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Keywords: Biodiversity, LCA, plastic recycling, HDPE, LC-Impact, ReCiPe 2016, IMPACT World+

This master's thesis aimed to investigate the potential biodiversity impacts of virgin HDPE granulate production and end-of-life (EoL) treatment options on species composition, specially focusing on potentially disappeared fraction of species. The assessment compared three different EoL treatment options: 1. incineration and landfill disposal, 2. mechanical recycling through extrusion, and 3. chemical recycling through pyrolysis.

As there is no globally accepted LCIA method for biodiversity impacts, three methods were selected for impact evaluation and comparison: ReCiPe 2016, LC-Impact, and IMPACT World+. The results obtained indicated that all methods placed the scenarios in the same order, with Scenario 2 showing the lowest impact and Scenario 1 having the highest impact. In all methods, impacts of climate change had the largest contribution to the overall result, while notable contributions from acidification and land use were also noted.

Differences among the methods were noted in the shares of processes and impact categories from the overall results. In particular, landfilling from the processes and eutrophication and ecotoxicity from the impact categories showed largest variation between the methods. Therefore, the selection of LCIA method can influence the conclusions regarding which impact categories and processes should be prioritized for actions. Recycling methods were found to be better options from the perspective of biodiversity impacts, and increasing and optimizing their contributions could reduce current biodiversity impacts of plastics.

TIIVISTELMÄ

Lappeenrannan–Lahden teknillinen yliopisto LUT

LUTin energiajärjestelmien tiedekunta

Ympäristötekniikka

Veera Leinonen

Vertaileva elinkaariarviointiin perustuva korkeatiheyksisen polyeteenin biodiversiteettivaikutusten arviointi - Elinkaaren loppuvaiheen käsittelyvaihtoehtojen vertailu: poltto ja kaatopaikka, mekaaninen kierrätys, ja kemiallinen kierrätys

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81 sivua, 31 kuvaa ja 17 taulukkoa

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Avainsanat: Biodiversiteetti, LCA, muovin kierrätys, HDPE, LC-Impact, ReCiPe 2016, IMPACT World+

Tämän diplomityön tarkoituksena oli selvittää neitseellisten HDPE granulaattien valmistuksen ja loppukäsittelyvaihtoehtojen mahdolliset biodiversiteettivaikutukset lajikoostumukseen eli potentiaalinen lajikato. Tutkimuksessa vertailtiin kolmea eri loppukäsittelyvaihtoehtoa: 1. poltto ja kaatopaikkasijoitus, 2. mekaaninen kierrätys ekstruusion kautta ja 3. kemiallinen kierrätys pyrolyysin kautta.

Koska kansainvälisesti hyväksyttyä LCIA menetelmää ei ole saavutettu, biodiversiteettivaikutusten arviointiin valittiin kolme menetelmää ja suoritettiin näiden vertailu. Menetelmiksi valittiin: ReCiPe 2016, LC-Impact ja IMPACT World+. Saadut tulokset osoittivat, että kaikki kolme menetelmää asettivat skenaariot samaan järjestykseen, jossa Skenaario 2 aiheutti pienimmän vaikutuksen ja Skenaario 1 suurimman. Kaikissa menetelmissä ilmastonmuutoksen aiheuttamilla vaikutuksilla oli suurin osuus kokonaisvaikutukseen, lisäksi happamoitumisen ja maankäytön sekä maankäytön muutoksen aiheuttamat vaikutukset korostuivat.

Menetelmien välillä havaittiin eroja prosessien ja vaikutusluokkien osuuksissa kokonaisvaikutukseen. Erityisesti prosesseista kaatopaikkasijoitus ja vaikutusluokista happamoitumisen sekä ekotoksisuus osoittivat suurinta vaihtelua menetelmien välillä. Tähän pohjautuen havaittiin, että LCIA-menetelmän valinnalla voi olla vaikutusta johtopäätöksiin, kuten siihen mihin vaikutusluokkiin ja prosesseihin vähennystoimien tulisi kohdistua. Kierrätysmenetelmien havaittiin olevan biodiversiteettivaikutusten kannalta parempia vaihtoehtoja, ja niiden osuuden lisääminen ja optimointi voisi vähentää nykyisiä muovien aiheuttamia biodiversiteettivaikutuksia.

SYMBOLS AND ABBREVIATIONS

Roman characters

V	volume	m^3

Compounds

CO	Carbon monoxide
CO_2	Carbon dioxide
HCl	Hydrochloric acid
NO_x	Nitrogen oxides
SO_x	Sulphur oxides

Units

°C	Celsius
cm^3	cubic centimetre
d	day
g	gram
h	hour
J	Joule
l	liter
m	meter
m^2	square meter
m^3	cubic meter
min	minute
Pa	pascal

t	ton, tonne
W	watt

Abbreviations

%	percentage
ABS	Acrylonitrile Butadine Styrene
BIA	Biodiversity Impact Assessment
CF	Characterization factor
e	equivalent
EF	Effect Factor
E/MSY	extinction per million species per year
EoL	End-of-Life
EU	European Union
FAE	Fly Ash Extraction
FF	Fate Factor
GHG	Greenhouse Gas
HANPP	Human appropriation of the biosphere’s net primary production
HDPE	High-density polyethylene
HVC	Higher value chemicals
ISO	International Organization for Standardization
IW	IMPACT World+
LC	LC-Impact
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory

LCIA	Life Cycle Impact Assessment
LDPE	Low-density polyethylene
LHV	Lower Heating Value
MSW	Municipal Solid Waste
NPP	Net Primary Production
PM	Particulate Matter
PP	Polypropylene
PS	Polystyrene
PDF	Potentially Disappeared Fraction of species
PW	Plastic Waste
RE	ReCiPe 2016
VOC	Volatile Organic Compounds
VS	Vulnerability Score
WEEE	Waste Electrical and Electronic Equipment
XF	Exposure Factor

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1 Introduction

In everyday life, people benefit from a variety of different products including plastics. Plastics can be found in almost every product area, and they are important material for the development of societies. Plastics, for example, extend the lifespan of food products, work as a material for medical applications and protect us in traffic or sports. (Tenhunen and Pöhler, 2020.) When compared to steel, wood, or cement, plastics have advantages like light weight, durability, and energy efficiency (Zou et al., 2023). The characteristic properties of plastic make them highly versatile and irreplaceable material for various applications (Hammer et al., 2012). Population increase, and the need for higher living standards have radically increased the production of polymers during the past decades (Achilias et al., 2007). In 2019, the global production of polymer resin and fibers was 459.75 Mt. If plastic production increase continues on the current path, it is estimated that in 2060 the number will be 1.23 Gt. (Geyer et al., 2017; OECD, 2022a.) Globally, the largest quantity of plastics is produced for the packaging industry. In 2019, the quantity was 142.60 Mt, accounting for around 31% of total polymer resin and fiber production (OECD, 2022d).

The downside of plastic production is the several negative impacts to human health, animals, and the environment (Tenhunen and Pöhler, 2020). In 2019, 353 Mt of plastic waste was generated globally, of which 9% was recycled, 19% incinerated, 50% landfilled, and 22% ended up mismanaged. Mismanagement refers to uncontrolled dumpsites, burning in open pits, or ending up in terrestrial or aquatic environments. (OECD, 2022b.) The biological and chemical composition of conventional plastic products makes the degradation period in the natural environment long, causing plastic waste to accumulate in the natural environment or landfills. This can cause severe environmental pollution like air, groundwater, and marine pollution. (Zou et al., 2023.) Plastic production contributes to climate change, which poses multiple threats to nature and humanity. Worldwide, 98% of plastics are fossil-based, and only 2% comes from renewable raw materials. Greenhouse gas (GHG) emissions are released in several life cycle stages, such as manufacturing and incineration. In 2019, plastic production and incineration accounted for more than 850 Mt of GHG emissions. (Tenhunen and Pöhler, 2020.) Global GHG emissions were 49.8 Gt in 2019, meaning that plastic production and incineration accounted for around 1.7% (World Resource Institute, 2019).

The environmental impacts of plastic production include land use changes, pollution, poor water quality, chemical and waste contamination, and contributions to climate change. All these factors contribute to ecosystem degradation leading to loss of biodiversity. (WHO, 2015.) Biodiversity is the variety of life on Earth where a variety of species and organisms work together in ecosystems to maintain balance and support life. Healthy ecosystems provide us with multiple essentials resources, making life on Earth possible. Nature is also essential for our society's ability to cope with global changes, health threats, and disasters. Currently, scientists are estimating that one million species out of an estimated eight million are threatened with extinction, many even within decades. Some consider that we are currently witnessing the sixth mass extinction, and this one is highly due to human actions. (European Commission, 2020.)

Negative environmental impacts from plastic production presents multiple challenges to achieve environmental goals set both at the global and national level. The Paris Agreement on Climate change is an international and legally binding agreement on climate change targeting to limit the rise in global average temperature to below 2 °C (Ympäristöministeriö, n.d.). Current estimates project that cumulative emissions from the entire life cycle of plastics will exceed 56 Gt CO_2e by 2050, accounting to 10-13% of the total global carbon budget based on 1.5 °C target (Tenhunen and Pöhler, 2020). The European Union (EU) has set Plastics Strategy under Circular Economy Action Plan, aiming to reduce marine litter, GHG emissions, and dependence on fossil fuels. (European Commission, 2018.) The EU Packaging and packaging waste Directive sets targets for recycling of plastic packaging: 50% by 2025 and 55% by 2030 (European Commission, 2024a). The EU's Biodiversity Strategy for 2030 is targeting to protect nature and reverse the degradation of ecosystems, setting Europe's biodiversity on a path to recovery by 2030. The strategy includes multiple specific commitments and actions to restore Europe's biodiversity. (European Commission, 2020.) Negative environmental impacts throughout the life cycle of plastic must decline to support these targets.

To identify and tackle negative biodiversity impacts resulting from plastic production, more information about the comprehensive impacts and "hot spots" is needed. Life cycle assessment (LCA) is a comprehensive method to examine the environmental impacts of products or services throughout their life cycle, e.g., from natural resource acquisition, via production and usage to end-of-life management. The International Organization for

Standardization (ISO) provides standards and reports that define the accepted framework and terminology of LCA. (Curran, 2015.) During the past two decades, multiple improvements have been made to LCA methodology, one of these being the inclusion of biodiversity. Even though biodiversity assessment is still challenging, it is possible to perform biodiversity impact assessments (BIA) based on LCA. In the life cycle impact assessment (LCIA) phase of LCA, the biodiversity impact can be quantified. Several impact assessment methods that include biodiversity exist, but all methods have some gaps or weaknesses. (Manders, 2022; Damiani et al., 2023.)

BIA of plastic production is still a new subject and only a couple of publications exist. Most comprehensive being publication by Nessi et al. (2020) which is European Commissions publication and showcases 10 LCA case studies using Product Environmental Footprint (PEF) method, and in addition, quantifies other relevant environmental information: GHG emissions from consequence of indirect land use (iLUC), biodiversity impacts, and macro- and micro-plastic generations. The study focuses on different plastic products and compares production scenarios. A study by Leppäkoski et al. (2022) studied climate and biodiversity impacts of low-density polyethylene (LDPE) production through power-to-X process focusing on land use change. In addition, several LCA studies on the subject can be found but they do not quantify biodiversity impacts. More information on the biodiversity impacts of plastic production is still needed to support the sustainable development of plastic production.

This thesis assesses the biodiversity impacts of HDPE granulate production and compares end-of-life (EoL) treatment options for separately collected post-consumer plastic waste. HDPE is a type of polyolefin, which are one of the most used forms of plastic. In 2019, the share of HDPE in global primary plastic production by polymer was 12.1%, equating to 55.54 Mt (OECD, 2022c). Several everyday products like plastic bags, toys, and detergent bottles and other packaging materials are made from HDPE (EuRIC, 2020). Virgin HDPE granulates are produced from fossil-based raw materials. Other production methods include recycling processes utilizing plastic waste as raw material or the utilization of biobased raw materials. This thesis evaluates recycling methods from the perspective of EoL treatment options. The goal of the study is to quantify the biodiversity impacts and more specifically potentially disappeared fraction of species associated with the production of virgin HDPE granulates and its EoL treatment, as well as to compare different EoL treatment options.

Three different EoL scenarios are formed and assessed: combination of incineration with energy recovery and landfill, mechanical recycling, and chemical recycling.

The conducted BIA follows the ISO 14040:2006 framework for LCA. The BIA is performed using cradle-to-gate with EoL, that includes all life cycle stages from raw material acquisition to the phase where HDPE granulates leave the production factory and its EoL treatment, but limiting out the further use of granulates to produce plastic products and their consumer use. The BIA is carried out using SimaPro software where three suitable impact assessment methods for quantifying biodiversity impacts are embedded: ReCiPe2016, IMPACT World+ and LC-IMPACT. Results are calculated using all three of the impact assessment methods and the methods are compared. The data collection is limited to EU area and is based on secondary data from scientific papers and life cycle inventory (LCI) databases.

Research questions of the study:

1. What are the biodiversity impacts of HDPE granulate production and EoL treatment options?
2. Which product life cycle stage has the most significant impact on biodiversity loss?
3. How do assessed LCIA methods differ in their evaluation of biodiversity impacts?

The thesis begins with a theoretical framework that is divided into chapters 2 and 3. Chapter 2 introduces what biodiversity is, its current state and how biodiversity can be assessed. Chapter 3 presents the current state of plastic production in the EU, covering environmental problems, regulations, and targets. This chapter also delves into more detail on HDPE, its production, and the assessed EoL treatment options. Following the theoretical chapters, Chapter 4 describes the study methodology, including the requirements of comparative LCA, LCA of recycling processes, assessment of biodiversity in LCA, and the selected LCIA methods. Chapter 5 presents the LCA-based BIA, including all mandatory parts specified in ISO 14040: goal and scope definition, inventory analysis, impact assessment and result interpretation. At the end of the thesis, the discussion examines the implications of the conducted study for other areas of research and identifies potential improvements for further development. The thesis is finalized by presenting conclusions that summarize the main results and findings. This thesis is carried out in collaboration with Sweco Finland Oy.

2 Biodiversity

In this chapter, biodiversity is explained in more detail in three sub-chapters. The first part explains what biodiversity is and its significance. The second part reviews the current state of biodiversity on a global scale and the third part addresses reasons for biodiversity loss. The chapter also examines the EU's targets related to biodiversity.

2.1 Definition and significance

Biological diversity or biodiversity refers to the variety of life on Earth. Multiple different definitions of biodiversity can be found, but the definition from Convention on Biological Diversity is quite comprehensive and largely accepted: "Biological diversity means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complex of which they are part; this includes diversity within species, between species and of ecosystems." (Gaston and Spicer, 2004.) As also mentioned in the definition, biodiversity can be divided into three distinct levels: species diversity, genetic diversity, and ecosystem diversity.

The species diversity refers to the number of distinct species on Earth from single-celled bacteria to multicellular kingdoms like animals. A general way to depict the level of species diversity is through species richness, which can refer to either the number of species or a list of species inhabiting a given area. Species diversity contributes to ecosystem health, and each species has an important role in the functioning of an ecosystem. Genetic diversity refers to the biological variation within species, both between different populations and between individuals in one population. It is important for species to maintain reproductivity, adapt to changes, and resist diseases. Ecosystem diversity refers to the variation in ecosystems within geographical location, including habitats, communities, and ecological processes. Ecosystem diversity is essential for sustaining ecosystem functions and facilitates vital interactions that support the survival of many plant and animal species. (Sher, 2022.)

Biodiversity can be considered as an indicator of the well-being of life on Earth. A diverse environment is more sustainable and resilient, ensuring prosperous life for all life forms.

Biodiversity enables functioning ecosystems where each life form has a unique role in ensuring the stability and resilience of ecosystems. (European Commission, 2020.)

Biodiversity provides a variety of fundamental materials and services to people, some of which are irreplaceable. The value provided by biodiversity and ecosystem services can be divided into direct-use values, indirect-use values, and non-use values. Direct-use values refer to benefits that are derived directly and are consumed or utilized by humans. These include goods such as food, timber, medicine, and recreation. (Gaston and Spicer, 2004.) For example, biodiversity works as the foundation for all our food-related industries and services, more than 2 billion people are dependent on wood as their primary energy source, and natural products support human health by providing medicine (Gaston and Spicer, 2004; IPBES, 2019). Indirect-use values are the non-consumptive benefits that indirectly support human well-being and ecosystem functioning without harvesting or destroying resources. These include services such as carbon sequestration, water purification, pollination, and climate regulation. For example, the natural environment maintains suitable living conditions on Earth by cycling elements such as nitrogen, carbon, hydrogen, and oxygen. These cycles modify physical and chemical conditions, creating an environment that makes life possible. (Gaston and Spicer, 2004.) Also, 75% of global food crop types are dependent on animal pollination (IPBES, 2019). Non-use values refer to biological resources that are not directly or indirectly exploited. These include values such as maintaining biodiversity for future direct or indirect use, preserving biodiversity and ecosystem resources for future generations, and cultural and societal importance. (Gaston and Spicer, 2004.)

2.2 Current state

Concern about biodiversity loss caused by human action has been recognised as a major environmental problem facing the Earth. Conservation biologists have recognised this threat since decades but now it has gained wide concern. (Damiani et al., 2023.) Our society has put too much pressure on the world's biodiversity, and measures to preserve and restore it are needed. Due to human action, almost 75% of Earth’s surface has been occupied, leaving nature and wildlife with only a small fraction of what they had. It is estimated that during past four decades global wildlife population has decreased approximately by 60% due to human activities, and this trend is persisting. (European Commission, 2020.) Significant

harm is caused due to deforestation, as forests are one of the most species-rich habitat types. Land transformation from forests to agricultural use is currently responsible for 85% of the 28 000 species that are close to extinction. (Hald-Mortensen, 2023.)

Assessing the current state of biodiversity, especially on a large scale, is not straightforward due to its complex and multidimensional nature. Measuring and monitoring is challenging, and multiple methods to measure and report biodiversity have been developed. (Watermeyer et al., 2020.) Planetary boundaries define a safe operating space for humanity in terms of respecting Earth's systems, allowing humanity to develop and thrive also in the future. Safe operating space has been defined for nine planetary systems: climate change, novel entities, stratospheric ozone depletion, atmospheric aerosol loading, ocean acidification, biochemical flows, freshwater change, land-system change, and biosphere integrity. In six of the planetary systems, the safe operating space has been already exceeded, biosphere integrity being one of them. (Rockström et al., 2009.) Figure 1 presents the planetary boundary framework.

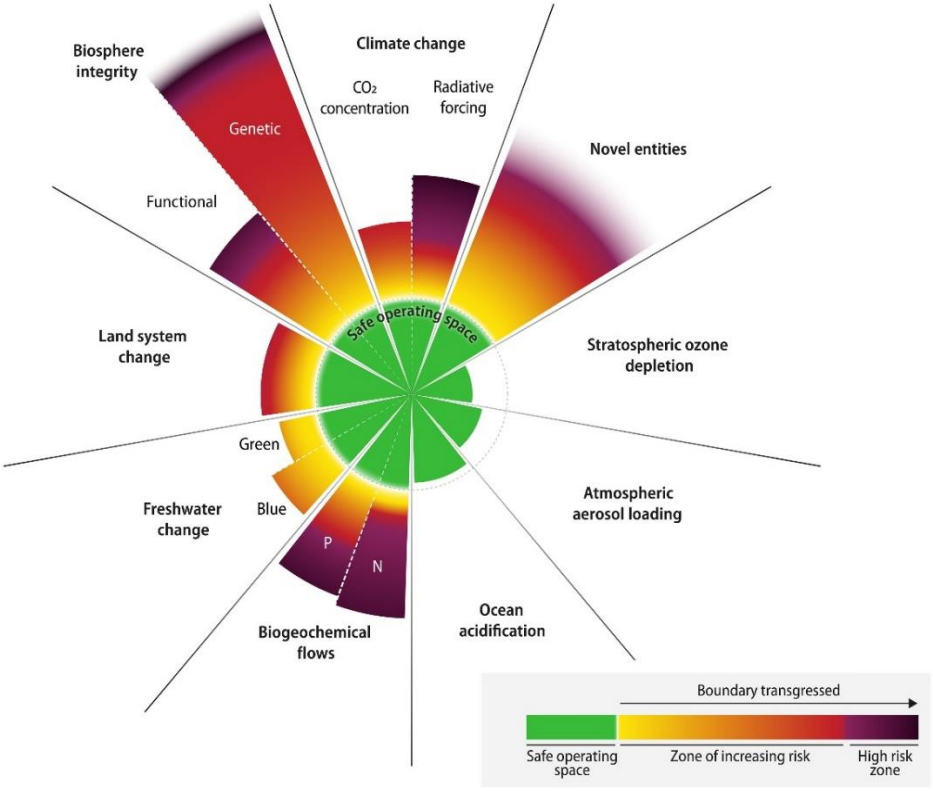

Figure 1. Planetary Boundary framework (Richardson et al., 2023.)

The planetary boundary for biosphere integrity is based on genetic diversity and planetary function dimensions on a global scale. Change in genetic diversity is assessed through

species extinction, and the safe operation space is set based on the maximum extinction rate that is compatible with preserving the genetic basis of the biosphere's ecological complexity. The safe operating space boundary is set to < 10 E/MSY (extinctions per million species per year), the high risk boundary to 100 E/MSY, and currently it is assessed to be > 100 E/MSY. Species extinctions occur also naturally, but the current extinction rates show rapid acceleration compared to natural extinction rates. It is estimated that current extinction rates are tens to hundreds of times higher than average extinction rates. (Richardson et al., 2023.)

The functional dimension is assessed using photosynthetic energy and material flow into the biosphere, i.e., net primary production (NPP). NPP is essential for ecosystems and human societies to maintain reproduction, differentiation, networking, and growth. Biomes utilize energy flow associated with NPP to maintain the ecological functions of the planet. The boundary for the safe operating space is a limit set for Human Appropriation of the biosphere's NPP (HANPP), with the reference value being the preindustrial Holocene average NPP. The unit of NPP is mass of carbon per year. The safe operating space boundary is set to <10%, the limit for high risk to 20%, with the current state being 30%. (Richardson et al., 2023.)

2.3 Drivers of biodiversity loss

To understand the negative impacts on biodiversity, it is important to identify drivers behind biodiversity loss. Direct drivers behind current biodiversity loss are land- and sea-use and change of use, direct exploitation, climate change, pollution, and invasive species (IPBES, 2019). Figure 2 presents the share of direct drivers on biodiversity loss in terrestrial, freshwater, and marine environments on a global scale.

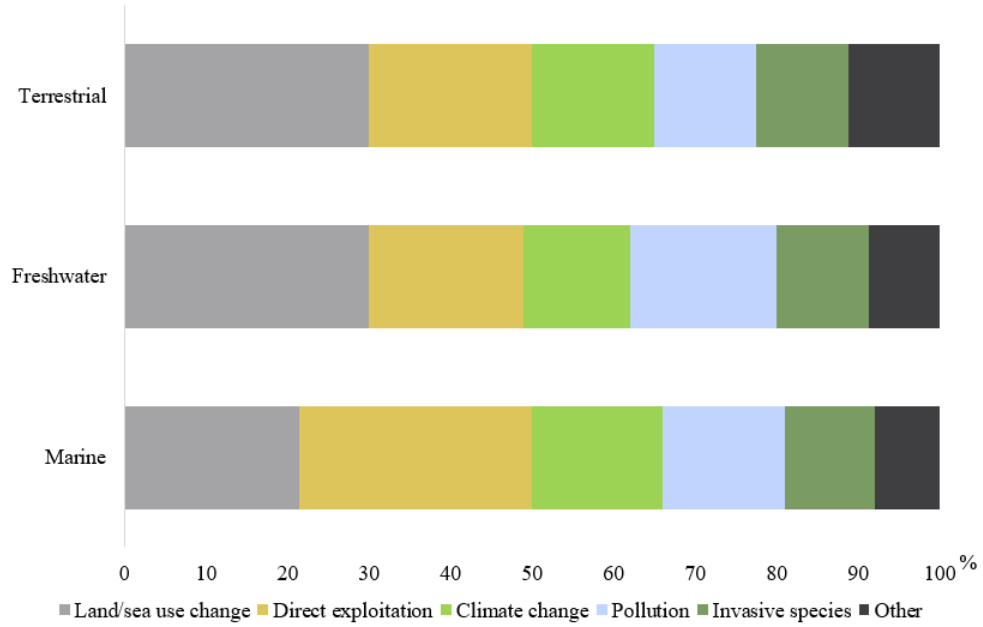

Figure 2. Relative global impact of direct drivers estimated based on global systematic review of studies published since 2005 (IPBES, 2019).

Land use change is the main direct driver in terrestrial and freshwater environments, causing, for example, habitat loss and degradation leading to harmful effects on species and populations. Land-use processes are reducing habitat availability and increasing isolation leading to patchy environments. (Fraile, 2014.) The consequences include reduced species richness and population genetic diversity, and modified community structures (Fraile, 2014).

Climate change exacerbates and speeds up the ongoing loss of biodiversity. In ocean ecosystems, climate change elevates ocean temperature and acidity and causes alternations in ocean currents. Warming oceans no longer provide habitats for certain species and pushes ecosystems and species towards the poles. On the other hand, habitats in pole areas are altering due melting of ice. The rising concentration of carbon dioxide in the atmosphere is increasing the absorption of carbon into the world's oceans, leading to acidification. Disruptions to ocean currents can have profound consequences for heat distribution, maintaining regional climates, and supporting marine food webs. In terrestrial areas, climate change is causing desertification and water shortage, leading to the loss of plant species and effects on animals dependent on these plants. Rivers, lakes, and other wetlands may dry up, leaving multiple species that rely on freshwater ecosystems without habitats. (Hald-Mortensen, 2023.)

Direct exploitation is causing declines in population sizes and alternations to ecosystems, creating a risk of species extinctions. It can also lead to a relocation of species or cause permanent damage to ecosystems, as has been witnessed with trawling in seafloor habitats. Changes in certain species can have far-reaching effects on biodiversity, due to the highly complex and interconnected nature of ecosystems. (Hald-Mortensen, 2023.)

Pollution has a high impact on biodiversity loss and ecosystem alterations, particularly due to chemical contaminants and waste. The use of toxic substances can decline both plant and insect populations, causing disturbance to ecosystems in terrestrial and water environments. (Hald-Mortensen, 2023.) Untreated urban and rural waste, oil spills, and toxic dumping have negative impacts on soil, freshwater, marine water quality, and the quality of the global atmosphere. Plastic pollution in the marine environment is causing fatal problems, affecting at least 267 distinct species like turtles, sea birds, and marine mammals. (IPBES, 2019). Ammonia from agriculture and nitrogen oxides from combustion are causing excessive nitrogen deposition, leading to loss a of biodiversity through eutrophication (EEA, 2021). In soil, an excessive amount of nitrogen can cause acidification which affects plant growth, food chains, and the quality of agricultural products. (Hald-Mortensen, 2023.)

Invasive species are non-native organisms bypassing geographical barriers mainly due to human actions. Invasive species disrupt ecosystems and cause environmental, economic, or health problems in the new area. Species are often relocated due to deliberate introduction, international trade, tourism, or carried in the ballast water used by container ships. The number of invasive species has increased since 1980, and nearly 20% of Earth's surface faces risk of plant and animal invasions (IPBES, 2019). Invasive species pose several threats, including competition with native species for resources such as food, light, and mates; alteration of habitat structure; increased predation and parasitism; changes in ecosystem processes; alterations in abiotic conditions, such as soil moisture and acidity; and loss of biodiversity. These factors can lead to the decline or extinction of local species. (Hald-Mortensen, 2023.) Figure 3 summarizes direct drivers of biodiversity loss and presents examples of actions/issues behind the direct drivers. The examples of invasive species presented are from Europe.

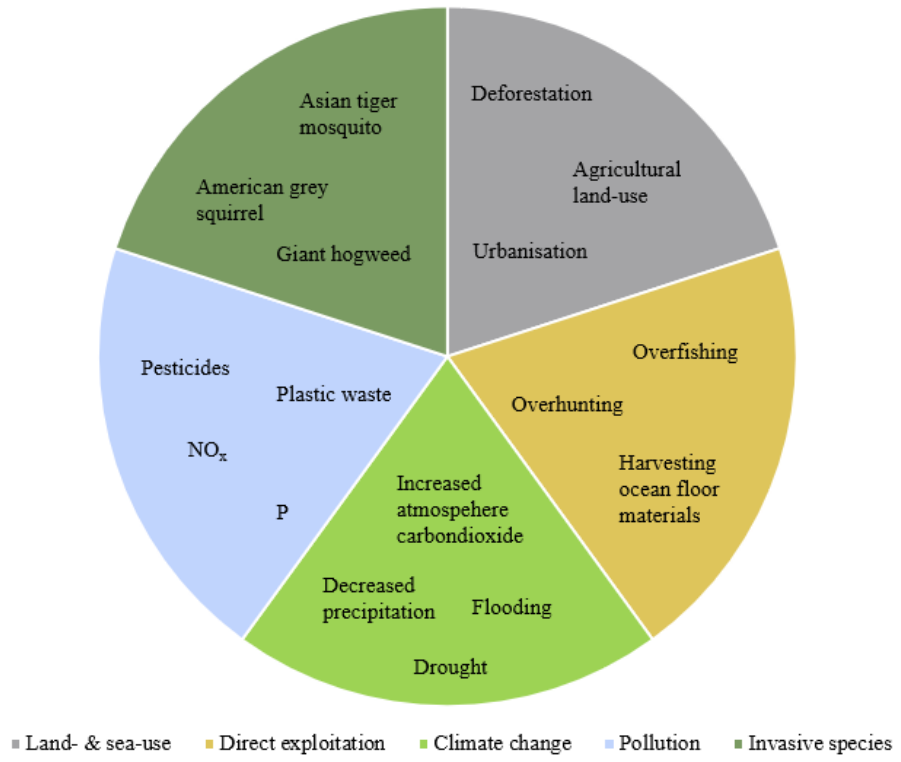

Figure 3.Examples of actions and issues behind direct drivers of biodiversity loss (Hald-Mortensen, 2023; IPBES, 2019; EFSA, 2014)

2.4 Biodiversity targets set by the EU

Current legal frameworks, strategies, and action plans to protect nature and restore habitats have not reached sufficient results. Therefore, the EU has set an ambitious Biodiversity strategy for 2030 to protect nature and reverse ecosystem degradation, guiding Europe on a path to recover biodiversity. The strategy includes several actions to establish, for example, better nature protection, restoration of degraded ecosystems, and better integration of biodiversity into decision-making, including the development of methods, criteria, and standards to measure environmental footprints of products and organisations throughout their life cycle. By 2050, the goal is that all the world's ecosystems are restored, resilient, and adequately protected. (European Commission, 2020.)

One of the recent developments in protecting and restoring the EU's biodiversity is the Nature Restoration Law that was adopted by the EU council on June 17, 2024. It is a critical part of the EU's biodiversity strategy. It is targeting to restore ecosystems, habitats, and species across the EU's land and sea areas. It aims to restore 20% of the EU's degraded land

and sea areas by 2030 and all areas in need of restoration by 2050, with a specific focus on ecosystems that have an ability to capture and store carbon and to prevent and reduce the impact of natural disasters. The Nature Restoration law includes seven main targets: those based on existing legislation, as well as targets for pollinating insects, forest ecosystems, urban ecosystems, agricultural ecosystems, marine ecosystems, and river connectivity. As part of implementation, EU countries are required to submit their National Restoration Plans by mid-2026, where they present their plans for achieving the targets. (European Commission, 2024b.)

3 Plastic production

This chapter presents the characteristics of plastic production in Europe and its associated targets. It includes a technological review of HDPE, detailing its granulate production and EoL treatment methods, as well as a discussion of the environmental and biodiversity impacts of plastic production.

3.1 Production in Europe

The plastic industry in Europe produced 58.7 Mt of plastics in 2022. The industry provides approximately 1.5 million jobs and had a turnover of 405 billion euros in 2021. Europe is one of the key plastic producers, covering around 15% of global plastic production. The industry is important for Europe's competitiveness and for meeting plastic demand across all sectors. Most of the produced plastic is used as a packaging material, which accounted for 39% of Europe's plastic production in 2022, with building and construction being the second largest with a share of 23%. However, the industry is in the middle of a change. Regulations and targets require changes to current production methods, as 80.3% of produced plastic was still produced from fossil-based raw materials in 2022. The share of mechanical recycling was 18.5%, bio-based 1.0%, and chemical recycling 0.2%. (PlasticsEurope, 2022; PlasticsEurope, 2023.) In 2020, out of 29.5 Mt of collected post-consumer waste 42% were sent for energy recovery, 34.5% were recycled, and 23.5% were landfilled (PlasticsEurope, 2022).

The EU is working to tackle plastic pollution and marine litter and to achieve a circular plastic economy. Rules and guidelines have been set for certain plastic categories like single-use plastics, plastic packaging, and microplastics. (EEA, 2024) The EU's Plastic Strategy under Circular Economy Action Plan is targeting to change plastic designing, production, use, and recycling in the EU towards greater sustainability. Packaging and packaging waste Directive aims to reduce plastic packaging waste and increase its recycling and other forms of recovery. (European Commission, 2024c) The plastic industry is also highly committed to meeting these targets and has set a roadmap for plastics in Europe to be circular and

achieve net-zero emissions by 2050. Investments are being made in new production methods and to enhance current technologies to achieve these goals. (PlasticsEurope, 2022.)

3.2 High-density polyethylene

High-density polyethylene is a type of polyolefin produced all over Europe. Polyethylene plastics are used in large quantities, and the production is fairly cheap. Produced HDPE is used, for example, in the production of bottle caps, buckets, food storage containers, and piping for water and sewer. Among polyolefins, HDPE is the most rigid and hard material due to the number of side branches. It is produced from ethylene monomers which are mainly derived from crude oil-based fuels like naphtha and consist of several thousand reacted monomer units that form a molecular chain. Its chemical formula is $(C_2H_4)_n$, and in addition to these monomer units, 3 to 5 methyl side groups are typically attached per 1000 carbon atoms along the linear backbone. The side branches are usually short. HDPE has a density higher than 0.940 g/cm^3, and the typical density is between 0.941 g/cm^3 and 0.960 g/cm^3. (PlasticsEurope, 2014; Nanda and Berruti, 2021.) HDPE exhibits high tensile and compressive strength, as well as good resistance abrasion. It has one of the highest impact strengths compared to other thermoplastics. Additionally, it is easy to process, and demonstrates resistance to corrosion, moisture, and water. However, there are some disadvantages, including sensitivity to stress cracking, poor UV resistance, the potential release of hazardous fumes when heated excessively, and the production of dense smoke when burned. (Vasile and Pascu, 2005.)

3.2.1 Virgin HDPE granulate production

Chain polymerization is the primary method used to produce polyethylene. The double bond between carbon atoms is forced to open, creating a radical that subsequently attaches to another monomer molecule. The polymer type and its properties are determined by the selection of reactor type, catalyst, initiator, and possible co-monomers. (PlasticsEurope, 2014.) HDPE is produced via a low-pressure process, where the pressure is maintained under 2 MPa (Zhong et al., 2017). The main production methods include slurry suspension, gas phase, slurry suspension/gas phase, and solution processes. The reactor types used are stirred

tanks, loop reactors, fluidized beds, and combination of loop and fluidized beds in series. The primary catalyst types are Ziegler-Natta, Phillips, and metallocene. In HDPE production chain growth is accomplished by insertion. In this method monomers are inserted into the polymer at the centre of the catalyst, allowing very defined structure. Wide range of production methods, and specified properties exist for HDPE. Figure 4 presents HDPE production process through suspension with stirred tank reactors. (PlasticsEurope, 2014.)

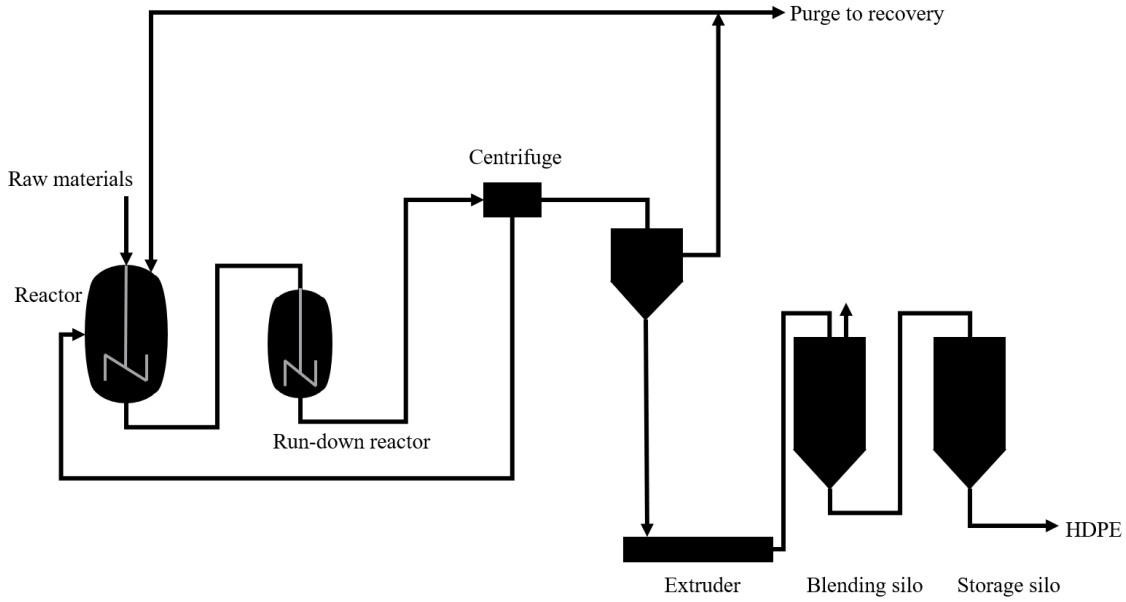

Figure 4. Representation of the flow diagram of HDPE production. Example of the suspension process with stirred tank reactors (PlasticsEurope, 2014)

3.2.2 Landfilling

Landfilling is commonly used method for managing plastic waste at its EoL in Europe. In 2020, 38% of plastic collected via mixed waste and 8% of separately collected waste plastic were placed in landfills (PlasticEurope, 2022). A Landfill functions as an ecological system, where solid waste and water serves as inputs, resulting in leachate and gas as outputs. (Wojnowska-Baryla et al., 2022)

A typical municipal solid waste (MSW) landfill consists of five waste treatment phases: initial adjustment, where aerobic biodegradation occurs; the transition from aerobic to anaerobic conditions; acid formation known as hydrolysis; methane fermentation known as methanogenesis; and the maturation and stabilisation of the solid waste. Each phase

contributes to plastic degradation. However, most polymers and plastics remain largely unchanged or only partial degradation to fragments occur. These fragments may persist as they were formed or biodegrade into gaseous products and water. (Wojnowska-Baryla et al., 2022)

The gaseous products from anaerobic conditions are carbon dioxide, while from anaerobic conditions, the gases consist of a mixture of carbon dioxide, methane, and volatile organic compounds. Landfill gases can be extracted and utilized in combustion processes to produce heat or electric energy. The landfilling of plastic waste is currently a wide concern, as it is often uncertain whether the plastic will degrade, biodegrade, or remain unchanged. (Wojnowska-Baryla et al., 2022)

3.2.3 Incineration with energy recovery

A significant share of generated plastic waste is incinerated in Europe. In 2019, 57% of plastic waste collected from mixed waste streams and 27% of separately collected plastic waste were incinerated (PlasticsEurope, 2022). During incineration, plastic waste is combusted to break down both organic and inorganic molecules, significantly reducing material volume. Additionally, energy recovery allows the energy content of plastics to be utilized in energy production. (National Research Council (US), 2000.)

The general incineration process includes the following operations: waste storage, feed preparation, combustion, ash handling, heat recovery, and air pollution control. After transportation to the incineration facility, waste is first stored in pits or piles before the preparation process begins. The waste is then treated to remove non-combustible materials to ensure proper combustion performance. In refuse-derived fuel (RDF) plants, the waste is also shredded, air-separated, and metals are recovered. This process improves the quality of the input material for combustion, increasing heating value and reducing bottom ash formation. (National Research Council (US), 2000.)

After preparation, the waste material is fed into furnace for combustion. The combustion process is a rapid exothermic reaction between the fuel and oxygen, with plastic waste serving as the fuel in this case. This process produces combustion air and gases. Due to incomplete combustion in certain parts of the furnace, non-combustible material remains

either as particulate matter known as fly ash in the flue gas or as bottom ash that collects at the bottom of the furnace. (National Research Council (US), 2000.)

Bottom ash is often placed in landfill, but it can be also used, for example, as an ingredient for ceramics (Vlasopoulos et al., 2023). Fly ash can be captured from the flue gas using extraction methods such as electrostatic precipitator or filter collectors. The produced gas can be used for electricity and/or heat production. (National Research Council (US), 2000.)

3.2.4 Mechanical recycling

Mechanical recycling is currently the main technology used to recycle post-consumer plastic waste. It is based on processing plastic waste by physical means back into secondary raw materials and products. It is most suitable for recycling well-separated and clean plastic waste. This minimizes losses and material degradation. The process can be carried out only on single-polymer waste streams. (Tenhunen and Pöhler, 2020.) Before the recycling process, the plastic waste needs to be collected and pre-treated. First, the plastics are sorted based on color and polymer type using automated and/or manual processes. After the first sorting process, plastics are shredded into flakes and possibly sorted once more, using, for example, sink/float methods. The flakes are then cleaned from dirt, contaminants, and other debris. The cleaned flakes are dried before melting and extrusion into pellets. (Yin et al., 2015.) The average efficiency of mechanical recycling in Europe is around 60%. Lost material is often used in energy production or by other industries. In some cases, it can also end up in landfills. (PlasticsEurope, 2022.)

After pre-treatment, plastic waste is melted and moulded, the most common technique used in mechanical recycling industry is extrusion. It is cheap, large-scale, solvent-free, and applicable to multiple polymers. The process combines heat and rotating screws to soften the material and after that the material is fed through a temperature-controlled barrel to form granulates. The melting temperature for HDPE rages between 200-300 °C. (Yin et al., 2015.) Figure 5 presents the reproduction process of HDPE from pre-treated plastic waste through extrusion.

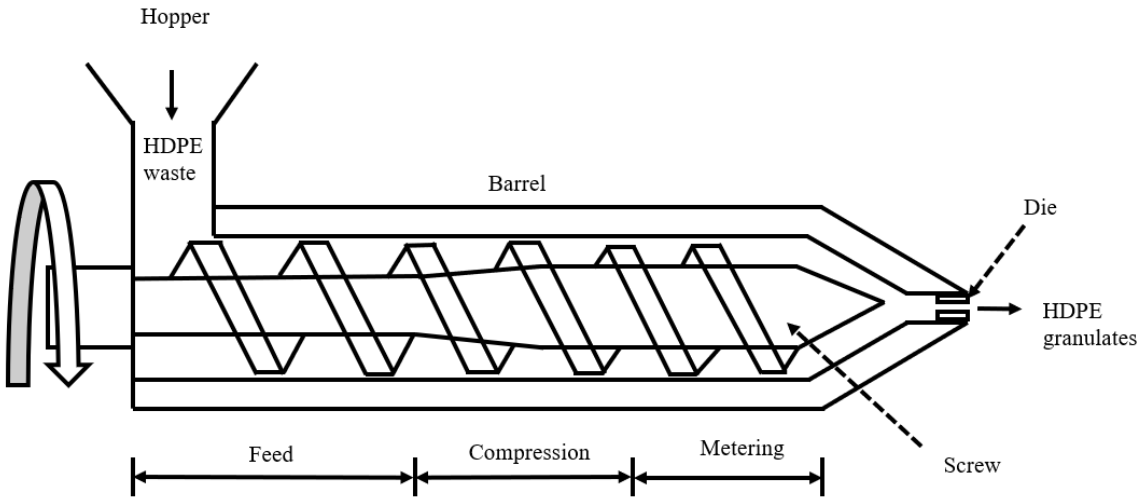

Figure 5. Schematic representation of single screw extruder (Schyns and Shaver, 2021)

The material often encounters degradation during mechanical recycling. Based on study by Oblak et al. (2015), HDPE can be mechanically recycled up to 10 times before significant changes occur in its properties. It is also stated that even after 100 recycling cycles, the material can retain 80% of its initial mechanical properties. Zhang et al. (2024) also studied effects of mechanical recycling on HDPE. In this study, results show that the material maintains its rheological properties for the first four recycling cycles. A common method to mitigate material degradation during mechanical recycling is melt blending. This approach involves blending virgin plastic with waste material or mixing different types of plastic waste, which helps reduce costs and improves material quality. (Schyns and Shaver, 2020; Yin et al., 2015.)

3.2.5 Chemical recycling

Chemical recycling makes it possible to increase recycling efficiency and keep the material longer in the cycle. It is a process that is energetically and chemically more demanding but can also utilize multi-material or contaminated products that cannot be recycled through mechanical recycling. (Tenhunen and Pöhler, 2020.) Chemical recycling is not applied yet on an industrial scale, but there are either pilot projects or small industrial and commercial scale plants operating in Europe. The biggest barrier for starting chemical recycling on an industrial scale is the lack of constant, affordable feedstock and unclear legislation. Despite these challenges, chemical recycling is considered a method of the future, also from a

sustainable development perspective. The methods are constantly evolving, and significant progress is expected in the near future. (PlasticEurope, 2022.)

In chemical recycling, polymers are broken back into substances that can be further used to manufacture plastics or other valuable products. This makes it possible to produce recycled plastic with the same properties as virgin plastics. Chemical recycling methods can be divided into processes involving mild temperatures and solvents, where products are mainly oligomers or monomers, and to high temperature processes, where the products are a mixture of char, oils, and gases. The first category includes depolymerization methods like glycolysis, hydrolysis, and aminolysis. The second category includes processes like pyrolysis, gasification, and hydrogenation. (Thiyagarajan et al., 2022.)

Pre-treatment is a crucial part of chemical recycling. It increases the cost, but a more stabilised unit operation and increased product quality are gained. The primary pre-treatment includes removal of extraneous materials and sorting of plastic types. Removal of extraneous materials consists of screening, sorting, washing, and flotation. For example, metals can be removed using techniques like magnetic or electrostatic separation. Washing consists of two separate phases, coarse and fine washing. Approximately 2-3 m^3 of water is needed to clean 1 t of polyolefin waste, which increases water use and the amount of released wastewater. To separate different plastic fractions, for example, sensor-based sorting can be utilized. It is an automated sorting method based on differences in spectral properties like transparency, colour, and clarity. (Zou et al., 2023.)

In pyrolysis, plastic waste is thermally decomposed in an oxygen-free environment. Pyrolysis processes can be divided into thermal pyrolysis, catalytic pyrolysis, and co-pyrolysis depending on whether a catalyst is used in the process. (Jiang et al., 2022.) Monomers for HDPE production can be produced through pyrolysis using two ways, either thermal cracking into monomers (one-step conversion) or production of feedstock to produce raw materials for monomer production (two-step conversion) (Pandey et al., 2020). Using a catalyst in pyrolysis is seen as a promising option to convert plastic waste into valuables, as it can decrease energy requirement and capital costs, and increase productivity and energy efficiency (Ukarde and Pawar, 2021). A suitable catalyst also lowers the melting temperature and influences the share of products (Achilias et al., 2007).

The one-step conversion process of plastic waste through pyrolysis involves cracking the polymers in a high temperature to an olefin (monomers of PP, LDPE, and HDPE) mixture and aromatic compounds like benzene, toluene, and xylene. Olefins are then separated as monomers from by-products using a fractional distillation process. Polyolefins as raw material produce a range of olefins and paraffins (gases). The highest olefins yield is obtained in a temperature of 700-750 °C. The high temperature is needed because liquid and waxes that are formed in lower temperatures crack to olefins and char in a higher temperature. When using a catalyst, needed temperature is approximately 650 °C. The commonly used pyrolysis reactor is fluidized bed, and nitrogen is used as a heat carrier and fluidized agent. In the two-step conversion process, the temperature is approximately 400-500 °C and three different pyrolysis products are received: pyrolytic liquid of condensable hydrocarbons (naphtha), high calorific value gaseous fraction and waxes consisting of paraffins, and char. Pyrolytic liquid is the desired product for HDPE production. Pyrolytic liquid is cracked utilizing conventional cracking or steam cracking to produce olefins. The gas fraction and char can be utilized in energy production. (Pandey et al., 2020; Achilias et al., 2014.)

Depolymerization, often referred to as chemolysis or solvolysis, uses combinations of chemistry, solvents, and heat to break down polymers into monomers. Depolymerization processes are operated in much lower temperatures than pyrolysis, typically under 200 °C. The advantage compared to pyrolysis is that it produces a defined product that is suitable for reprocessing polymers. Possible solvents are water, methanol, and glycols. The process is determined based on used solvent. For example, when water is used as a solvent, the process is called hydrolysis. Solvents are often used in combination with a catalyst. Possible contaminants are isolated from the process product, and monomers are used as a feedstock for conventional plastic production processes. (Thiyagarajan et al., 2022.)

3.3 Environmental and biodiversity impacts

Plastic manufacturing has a negative environmental impact at each phase of its life cycle. Conventional plastics are produced from combination of polymers and chemical additives to achieve the desired properties. Environmental impacts arise from various stages,

including, raw material acquisition, chemical manufacturing, polymerization, and EoL treatment. (EEA, 2021).

Raw material extraction

Plastic manufacturing requires substantial resource consumption, primarily derived from fossil fuels, as approximately 99% of plastics are sourced from fossil fuels, predominantly from oil and gas. In 2020 plastic industry used 7% of global oil consumption, with around 50% used as feedstock for plastic production and the other half consumed as fuel in the production process. (EEA, 2021).

The extraction of oil and gas contributes significantly to greenhouse gas (GHG) emissions and other environmental pollutants. GHG emissions primarily result from the industry's dependence on fossil fuels. The energy intensive processes involved in extracting raw materials including operation of extraction equipment and transportation generate CO_2 emissions due to combustion. Additionally, methane emissions are released into the atmosphere through both intentional venting and unintentional leaks from pipelines and gas engines during the extraction and utilization of natural gas (EEA, 2021).

Other environmental pollutants associated with fossil fuel extraction include nitrogen oxides (NO_x), sulfur oxides (SO_x), particulate matter (PM), volatile organic compounds (VOCs), heavy metals, and a wide range of chlorinated and other toxic organic chemicals. These emissions primarily arise from combustion in gas turbines and diesel engines, which generate energy for drilling operations, treatment of extracted oil and gas, and transportation activities. (EEA, 2021.)

A significant volume of wastewater, referred to as produced water, is generated during extraction. Produced water can contain harmful substances such as dispersed oil, heavy metals, and aromatic hydrocarbons. It is noted that produced water is one of the major contributors to oil pollution in the North Sea. (EEA, 2021.)

The direct effects on biodiversity can result from released wastewater containing oil and toxic chemicals, as well as from the installation and removal of platforms and drilling operations. (EEA, 2021.)

Production

Also, the production phase generates substantial GHG emissions. These emissions are released during the energy intensive processes involved in chemical production for plastic manufacturing and the conversion of polymers into plastic products. (EEA, 2021.)

In addition to GHG emissions, the production processes contribute to other forms of environmental pollution. For example, NO_x and SO_x are emitted during polymer manufacturing, which contributes to acidification. Toxic metals such as cadmium and mercury are also released into the air and water during production, these substances can accumulate in animals and plants. This accumulation can result in higher concentrations of these metals further up in the food chain, posing health risks to both wildlife and humans. Furthermore, the combinations of VOCs and NO_x emissions contributes to the formation of toxic tropospheric ozone. (EEA, 2021.) This ozone can be inhaled by humans or absorbed by plants, causing damage to human health and ecosystems (Verones et al. 2020).

Plastic production can also increase nutrient levels in water systems, leading to eutrophication, which causes decreases in oxygen levels that can results in a loss of biodiversity. (EEA, 2021.)

End-of-life

Plastic pollution poses significant threats to the environment and wildlife. Animals can suffer severe consequences, including entanglement, injury, and ingestion of plastic materials. Furthermore, additives commonly found in plastics can exacerbate their environmental impact. These additives are often not chemically bonded to the plastic matrix, which increases their potential for migration and leaching into the surrounding environment. In natural settings, factors such as temperature fluctuations and the presence of microorganisms can further enhance the likelihood of leaching, leading to greater ecological harm. (EEA, 2021.)

Even when plastics are treated at the EoL, environmental pollution still occure at various stages. The most common EoL treatment options for plastic waste are incineration and landfilling. During incineration, emissions are released at every phase of the process, including transportation, waste incineration, due generation of wastewater and ash, and during residual disposal in landfills. Diesel deriver transportation vehicles releases air pollutants like NO_x, PM, carbon monoxide (CO), and VOCs. Burning of waste in

incinerators releases emissions like dioxins and furans, PM, CO, NO_x, SO_x, HCl, and several metals. Air pollution control techniques lowers the quantity of emissions to air, but these emissions still end up in residue or wastewater. (Gaia, 2019.)

The environmental impact of landfilling is influenced by the type of landfill employed. When plastics degrade in landfills, they produce gaseous byproducts, including CO_2, methane, and VOCs. If these landfill gases are not captured and utilized, they are released into the atmosphere, contributing to greenhouse gas emissions. Additionally, the degradation process generates water, resulting in landfill leachate, which can contaminate nearby water bodies or infiltrate groundwater. (Redko et al., 2024.)

The degradation of plastics also leads to the formation of microplastics, which can enter the environment through leachate. These microplastic particles pose a significant threat to ecosystems and can have adverse effects on wildlife and human health. (Redko et al., 2024.)

In terms of soil impact, the degradation of plastic waste can release harmful substances such as preservatives, plasticizers (like dioctyl terephthalates), and coagulants into the soil. These chemicals can disrupt environmental homeostasis, alter microbial behavior, and contribute to soil ecotoxicity. The release of these chemical substances can affect soil properties, including pH levels, moisture turnover, water circulation, and redox potential, potentially leading to soil degradation. (Redko et al., 2024.)

The accumulation of plastic waste in landfills can result in the presence of toxic chemicals, microplastic particles, heavy metals, and various chemical substances in the environment. The environmental risks associated with this plastic waste deposit include altered chemical composition and parameters of the soil, disrupted diversity of the soil microbiome, and the presence of microplastic particles in soil, air, and water. These factors collectively contribute to significant ecological and health challenges that need to be addressed. (Redko et al., 2024.)

Knowledge about environmental impacts of recycling processes for plastic waste remains quite limited. However, LCA studies has shown that both mechanical recycling and chemical recycling seems better options than conventional EoL treatment options, particularly in terms of climate change (Jeswani et al, 2021; Hou et al., 2018). It is important to note that these processes do generate emissions through various activities, such as energy use, washing, catalyst utilization, and the treatment of material residues.

4 Methodology

In this chapter, the methodology of LCA-based BIA is presented. The chapter provides an overview of LCA, including the requirements for comparative LCA, the selection of approaches for assessing recycling processes, the possibility to assess biodiversity impacts in LCA, and the selected impact assessment methods used to quantify biodiversity impacts.

4.1 Life Cycle Assessment

LCA is a methodology to assess and quantify potential environmental impacts of a product or service throughout the life cycle. LCA provides tool to identify possibilities to improve the environmental performance of an assessed product or service in different phases of its life cycle. It supports in informing decision-makers, the selection of indicators of environmental performance, and marketing. According to ISO 14040 standard, LCA consists of four phases: goal and scope definition, life cycle inventory analysis (LCI), life cycle impact assessment (LCIA), and result interpretation. A complete LCA is a cradle-to-grave study considering all life cycle stages of a product or service under assessment. In this study, a cradle-to-gate with EoL study is conducted, meaning that the study includes raw material extraction and processing and EoL treatment phases. LCA is standardized by ISO and divided into two general standards, ISO 14040, covering principles and framework of LCA, and ISO 14044, covering the requirements and guidelines to perform LCA. (SFS-EN ISO 14040 2006.)

In the first phase, goal and scope definition, the goal and scope of the study are set. Goal definition includes the intended application, a reason to carry out the study, intended audience, and intended use of the study. Scope definition includes the system under study, function of the product system(s), functional unit, system boundary, allocation procedures, selection of impact categories, data requirements, assumptions, and limitations. The function of the product system(s) defines the performance characteristics of the system under assessment, and functional unit quantifies that performance and works as reference unit. All the input and output flows are determined to meet the selected functional unit(s). System boundary determines the unit processes included in the LCA and is consistent with the goal

of the study. The system boundary is often presented through flow diagram and includes all life cycle stages, inputs, outputs, and processes of the system under study. (SFS-EN ISO 14044 2006.)

In the second phase, LCI data are collected and processed based on the goal and scope definition. Data are collected for each unit process in the system boundary. Inventory analysis includes all material and energy flows and releases to air, water, and ground. The data are then utilized to quantify the inputs and outputs of unit processes in reference to the functional unit. Data collection methods and possible calculation methods need to be described and data should preferably be presented in a data collection sheet. If the system under study has more than one product or system, allocation might be needed. In allocation, inputs and outputs of the system are divided between products or functions in accordance with the selected procedure. Allocation is not always needed, and it should be avoided if possible. (SFS-EN ISO 14044 2006.) In this study allocation is avoided using system expansion, the system expansion approach is described in more detail later in this paper.

In the third phase, LCIA is conducted. LCIA is relative to the functional unit and quantifies environmental impacts from each life cycle stage. LCIA includes three mandatory parts: the selection of impact categories, category indicators, and characterization models, classification, and characterization. The selected impact categories are in line with goal and scope of the study. Also, the identification of category endpoint(s), definition of the category indicator for a given category endpoint(s), and identification of the characterization model and the characterization factors (CFs) are necessary components for each impact category. In classification, LCI results are assigned to selected impact categories. It needs to be noted that some results may contribute to more than one impact category. In characterization, category indicator results are calculated. LCI results are converted to common units and aggregated within the same impact category. Conversion is done using CFs and the result is numerical indicator. After characterization, the inputs and outputs of the system being assessed are represented. The optional parts of LCIA are normalization, grouping, weighting, and data quality analysis. These optional parts are not conducted in this study. (SFS-EN ISO 14044 2006.)

The fourth phase, interpretation of the results, includes the identification of significant issues based on results from the LCI and LCIA phases, evaluation of completeness, sensitivity and consistency checks, conclusions, limitations, and recommendations. An assessment and a

sensitivity check of the significant inputs, outputs, and methodological choices are required to understand the uncertainty of the results. The interpretation should also consider the appropriateness of the definitions of the system functions, the functional unit, system boundary, limitations identified by the data quality assessment and the sensitivity analysis in relation to the goal of the study. (SFS-EN ISO 14044 2006.)

4.2 Comparative LCA

Comparative LCA studies has requirements set in the ISO standards to ensure comparability of the studied systems. Scope of the study needs to be set in consideration of the comparability of the studied systems. For example, performance, system boundary, data quality, and allocation procedures need to be equivalent. Comparison between different systems must be done based on the same functions and functional unit. Possible differences need to be identified and reported. The equivalence of systems under study needs to be evaluated before interpreting the results. (SFS-EN ISO 14044 2006.)

4.3 LCA of recycling processes

In LCA of recycling processes, the need for allocation is typical. Allocation is performed in the third phase of LCA, and general principles and procedures of allocation described in ISO 14044 apply for reuse and recycling. Allocation can be avoided using two different methods: the distribution of processes into sub-processes or expansion of the product system. If allocation cannot be avoided, allocation should preferably be carried out using physical relationships between the products, for example, mass. If physical relationships cannot be used, the allocation is performed using other relationships. Some allocation requirements for reuse and recycling are described: changes in the material properties must be considered and the system boundary for recovery processes between the original and following product system should be identified and explained. In addition to general allocation procedures, the allocation can be done based on later use of the recycled materials. (SFS-EN ISO 14044 2006.)

In ISO 14044, recycling processes are divided into open-loop recycling and closed loop-recycling. In open-loop recycling, the product system using recycled material differs from

the original product system and the recycled material often encounters negative effects on its material properties. In closed-loop recycling, the recycling process produces virgin-like material, and the material can be recycled time after time. (SFS-EN ISO 14044 2006.) However, the allocation method described in the standard is not comprehensive enough and leaves the implementation of some requirements to the LCA practitioner, especially in the case of more complex open-loop recycling. Several methods have been developed to address this problem. (Ekvall et al., 2020.)

Recycling processes face an allocation problem in LCA because the same material is used at least in two different products. A suitable method to model material recycling has been discussed since the 90's but no agreement has been reached. The selection of a modelling method for the recycling process can have a large impact on the LCA results. (Ekvall et al., 2020.) Ekvall et al. (2020) studied the modelling of recycling in LCA and assessed 12 different methods for open-loop recycling. The report comprehensively presents information and assesses the methods under study. The assessed methods both deal differently with the allocation problem and define the problem differently. The allocation methods also vary in the incentives they give.

4.4 Assessment of biodiversity in LCA

Environmental burden of a product or service on biodiversity can be examined through LCA. The impact on biodiversity can be quantified in the third phase of LCA. In LCIA, inventory flows are converted using CFs into environmental impact score(s), which is a cumulative score of impact factors. Biodiversity is an endpoint indicator presenting the final measure of the environmental impact caused by emissions or resource use. (Manders, 2022.) LCIA consists of two different indicator types, midpoint and endpoint. Midpoint indicators can be considered as links between the cause-effect chain of an impact category before endpoints, reflecting the importance of emissions or extraction. Typically, midpoint level is at the point where an environmental mechanism is equal to all flows assigned to the same impact category. Examples of midpoints are global warming potential and ozone depletion. Endpoint indicators reflect the final cause to, for example, human health or ecosystems as a result of the midpoint indicators. The selected biodiversity impact assessment method determines how CFs are derived, and which indicators are included. (Huijbregts et al., 2017.)

In LCA, several methods for biodiversity impact assessment have been developed. Damiani et al. (2023) analyzed methods and models for biodiversity impact assessment in LCA and beyond including 64 different methods. 23 of these methods were selected for more detailed analysis. The analysis covered goal and scope, data use and needs, and impact assessment characteristics aiming to find strengths and weaknesses. Overall, the study revealed that none of the existing methods takes well into account the whole variety of pressures on biodiversity, ecosystems, taxonomic groups, essential biodiversity variable (EBV) classes, and the fundamental aspects to consider in a BIA. The study showed need to develop BIA methods especially in the fields of coverage of drivers of biodiversity loss, ecosystem and taxonomic coverage, assessment of ecosystem services, and indicators to analyze essential biodiversity aspects. In this study, three biodiversity impact assessment methods are selected to perform LCIA: ReCiPe 2016, LC-impact, and Impact World +. All three of the methods are embedded in the SimaPro software.

4.4.1 ReCiPe 2016

ReCiPe 2016, "RE" later in the text, is a life cycle impact assessment method both at midpoint and endpoint level. In the method, 17 midpoint indicators and 3 endpoint indicators are included. The endpoint indicators are human health, ecosystem quality, and resource scarcity. RE provides CFs that are representative for global scale and several impact categories to implement CFs also at a continental and country scale. The endpoint indicator for ecosystem quality reflects the impacts on biodiversity and is the focus of assessment in this study. The unit for ecosystem quality is local relative species loss in terrestrial, freshwater, and marine ecosystems, respectively, integrated over space and time: the potentially disappeared fraction of species·m^2·y or potentially disappeared fraction of species·m^3·y. To integrate terrestrial, freshwater, and marine ecosystems into one unit, species densities in these areas are utilized. The final unit for the endpoint level damage to ecosystem quality is species·y (time-integrated species loss). Endpoint CFs are derived from midpoint CFs using constant mid-to-endpoint factor per impact category. (Huijbregts et al., 2017.) Figure 6 presents the midpoint indicators contributing to endpoint indicator ecosystem quality and damage pathways in between. Regionalized impact categories are highlighted with red.

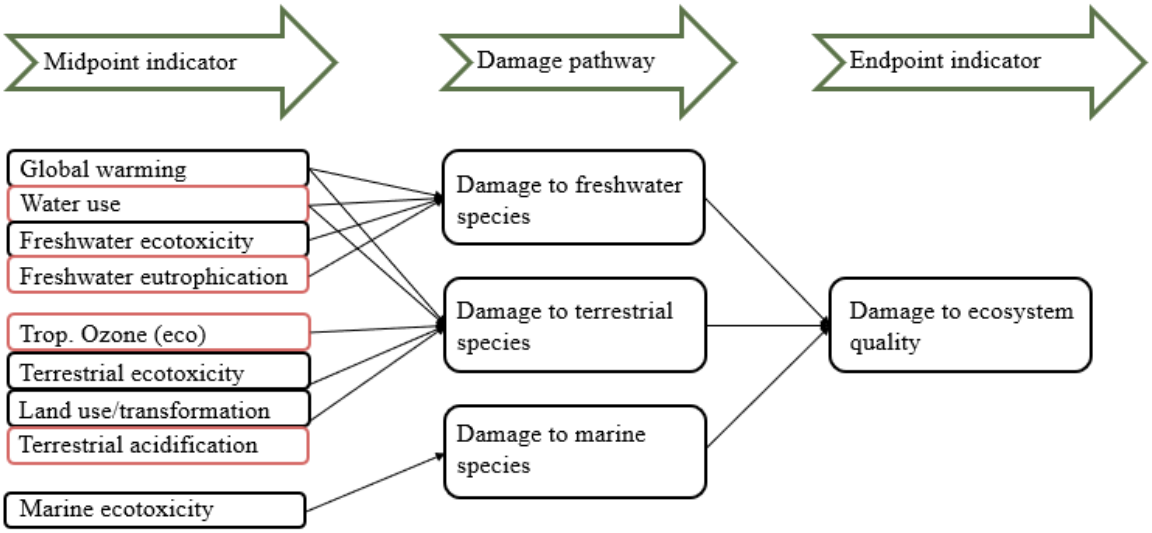

Figure 6. Midpoint indicators contributing to endpoint indicator ecosystem quality and damage pathway in between, modified from the original picture (Huijbregts et al., 2017). Regionalized impact categories are marked with red.

RE provides three sets of midpoint and endpoint CFs. The method uses a cultural perspective approach which includes individualist, hierarchist, and egalitarian perspectives. The perspectives have different value choices in the derivation of CFs and a different time horizon. The time horizon for the individualist perspective has 20 years, for the hierarchist 100 years, and for the egalitarian 1000 years - infinite. In this study, the hierarchist perspective is utilized as it is based on a scientific consensus and takes into consideration the time frame and plausibility of impact mechanisms. (Huijbregts et al., 2016.)

The article states that several improvements can still be made to the method. It mentions that further enhancements include advancements in modelling impact pathways, regionalization of more impact categories, change from local species loss to global species loss, and the incorporation of more impact pathways. In terms of regionalization, the most significant impact categories for delivering regionalized results are land use and toxicity. Currently, global species loss can be considered for the impact categories of water use and land use, but further research is required for other impact categories. (Huijbregts et al., 2016.)

4.4.2 LC-Impact

LC-Impact, “LC” later in the text, is a regionalized life cycle impact assessment method. Regionalization can be seen as an asset because large differences occur in environmental conditions through space. The regionalized CFs are provided for four spatial levels: the

original “native”, country averages, continental averages, and global averages. LC includes 11 midpoint indicators and 3 endpoint indicators. The endpoint indicators are human health, ecosystem quality, and natural resources. The endpoint indicator ecosystem quality is separated into three different ecosystem types: terrestrial, freshwater, and marine. The unit for ecosystem quality is the global potentially disappeared fractions of species over time (PDF·y). Combining the results of different ecosystem types would require a method to evaluate the importance of each ecosystem. For this reason, it is not recommended to combine ecosystem types, as a generally accepted weighting scheme does not exist. However, as a part of El Geneidy’s (2024) dissertation an equation to merge terrestrial, freshwater, and marine ecosystem results from LC-Impact method into one result was developed. The conversion is done using weights that are estimated based on share of all plant and animal species existing in each habitat type. Equation 1 demonstrates how to merge ecosystem results into a single result.

$$PDF_{total} = PDF_{terrestrial} * 0.801 + PDF_{freshwater} * 0.096 + PDF_{marine} * 0.102 \quad (1)$$

In LC, the result for ecosystem quality quantifies the global potential species loss for reason, as global species loss is irreversible. At the regional level, there is possibility for the recovery of extinct species. This leads to a lower numerical value of the impact compared to methods using area-specific results, but it does not imply that the impact is smaller. (Verones et al., 2020.) Figure 7 presents the midpoint indicators contributing to sub-categories of ecosystem quality in the LC method. Regionalized impact categories are highlighted with red.

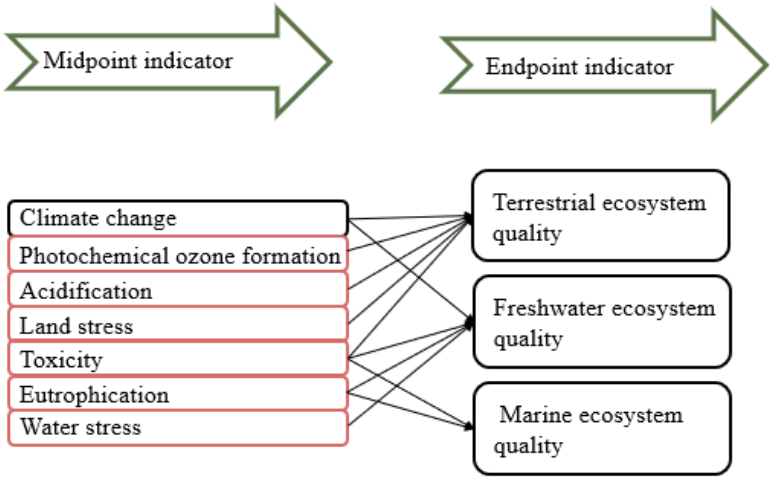

Figure 7. Midpoint indicators contributing to sub-categories of endpoint indicator ecosystem quality in LC-Impact method, modified from the original picture (Verones et al., 2020). Regionalized impact categories are marked with red.

The midpoint indicator climate change contains only global CFs due the global nature of the impact. Other midpoint indicators describe impacts limited to regional or local scale. The CFs of midpoint indicators may or may not inherently include a time dimension. For example, the CF for water stress is expressed in units of PDF·y/m^3, while the CF for land occupation is expressed in units of PDF/m^2. In these cases, a common unit (PDF·y) is achieved by multiplying the CFs by the inventory flows (m^3 in the case of water stress and m^2·y in the case of land occupation). The CF for the endpoint indicator of ecosystem quality quantifies relative global species loss by putting the regional species loss in the perspective of the global species pool. This is done across impact categories for one or more taxa, depending on data availability. (Verones et al., 2020.) The basic equation for the CFs for endpoint indicator of ecosystem quality is as follows:

$$CF_{ecosystem\ quality} = FF * XF * EF * VS \quad (2)$$

Where:

FF = fate factor
XF = exposure factor
EF = effect factor
VS = vulnerability score

LC-Impact provides four sets of CFs which are built in a modular way, allowing the user to add or neglect impacts that are farther away in a time perspective and less likely caused by a specific impact category. The four sets are: all impacts 100 years, all impacts long-term, certain impacts 100 years, and certain impacts long-term, where long-term sets have a time horizon between 1000 years and infinity. (Verones et al, 2020.) In this study, all impacts 100 years is selected, as it provides a high level of certainty regarding the effects of a specific intervention and aligns with the timeline chosen for the ReCiPe 2016 method.

In LC, the results of impact category ecotoxicity is assessed in different unit than results from other impact categories. In the study by Sanyé-Mengual et al. (2023) different LCIA methods were compared in the context of biodiversity impacts. To be able to assess contribution of all impact categories to the overall biodiversity impact a method to convert ecotoxicity into same unit as other impact categories were developed following the recommendations of the LC method developers. The biodiversity impact result for

ecotoxicity is presented in PDF·m^3·d while other impact categories in PDF·y. In the conversion process characterization factors of ecotoxicity are converted to PDF·y/kg. The conversion steps are presented in equations 3,4,5,6, and 7.

First, damage-level CFs are calculated for continental and global scale:

$$CF_{e,cont} = FF_{e,cont} * XF_e * EF_{e,cont} \quad (3)$$

$$CF_{e,glob} = FF_{e,glob} * XF_e * EF_{e,glob} \quad (4)$$

Second, the spatial specific CFs are converted from PDF·m^3·d/kg to PDF·y/kg using volume of the relevant exposure compartment, which are presented in Table 1:

$$CF'_{e,cont} = \frac{CF_{e,cont}}{V_{e,cont} * f_{\frac{d}{y}}} \quad (5)$$

$$CF'_{e,glob} = \frac{CF_{e,glob}}{V_{e,glob} * f_{\frac{d}{y}}} \quad (6)$$

Where:

V = Volume

f = generic factor to account 365.25 days per year

In the final step, calculated CFs in PDF·y/kg can be summed up to total CF for given emissions scenario that account effects in the exposure compartment both at continental and global scale:

$$CF'_e = CF'_{e,cont} + CF'_{e,glob} \quad (7)$$

Table 1. Ecotoxicity exposure compartment volumes at continental and global scale. (Sanyé-Mengual et al., 2023)

Compartment	Volume, continental scale [m^3]	Volume, global scale [m^3]
Freshwater	$6.76 \cdot 10^{11}$	$9.90 \cdot 10^{12}$
Seawater/Ocean	$9.87 \cdot 10^{13}$	$6.56 \cdot 10^{16}$
Natural soil	$4.37 \cdot 10^{11}$	$6.40 \cdot 10^{12}$

In this study, this same conversion method is used, and as a base CFs the methods CF set for all impacts 100 years is utilized. This is in accordance with the modelling choice for other impact categories. The set includes CFs for 27 metal ions.

4.4.3 IMPACT World+

IMPACT World+, "IW" later in the text, is a globally regionalised LCIA method that analyses global potential damages and provides CFs both at midpoint and endpoint levels. Most of the regional impact categories are spatially resolved and CFs are provided in global, continental, country, and native (non-aggregated) resolutions. The method includes 18 midpoint indicators, 21 damage indicators, and endpoint indicators from two different viewpoints. In the first one, the impacts are aggregated at the damage level to three areas of protection (AoP) that are human health, ecosystem quality, and resource and ecosystem services. The second viewpoint aggregate impacts to three areas of concern (AoC), which are water-related damages, carbon-related damages, and the rest of damages to human health and ecosystem quality. These are further divided into six sub-categories, including carbon human health, water human health, rest of human health, carbon ecosystem quality, water ecosystem quality, and rest of ecosystem quality. (Bulle et al., 2017.) In this study, the former viewpoint is utilized, as it is more comparable with other LCIA methods chosen for the comparative analysis. Figure 8 presents the impact categories and damage level indicators contributing to endpoint indicator ecosystem quality. Climate change short term, climate change long term, and water scarcity are proxies and this way not directly on the cause-effect chain leading to damages. Yellow arrows are presenting emissions, and resource uses contributing straight to damage pathways. Regionalized impact categories are highlighted with red.

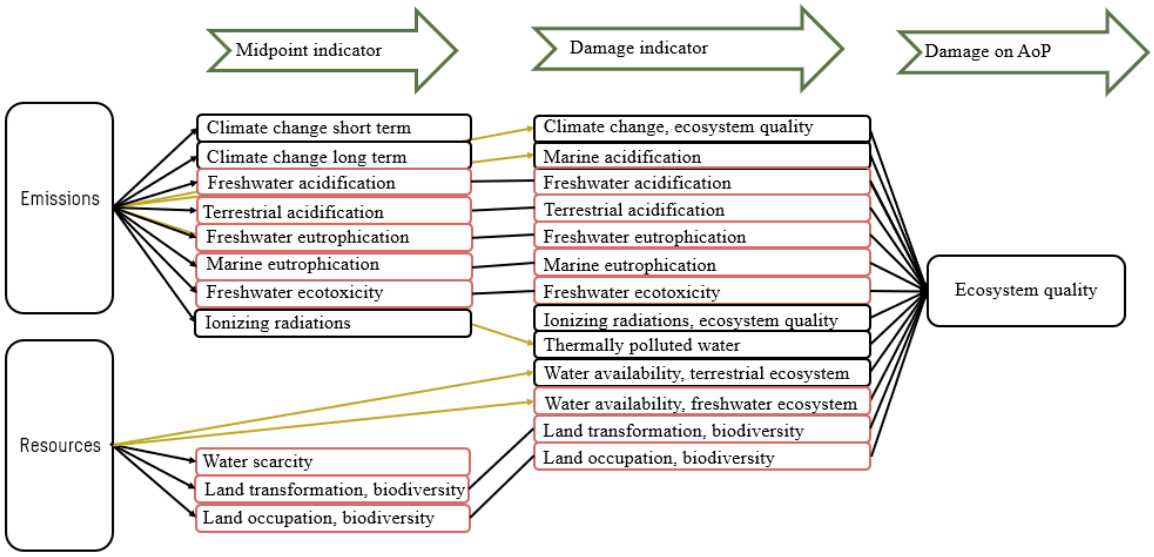

Figure 8. Impact World + LCIA framework related to endpoint indicator ecosystem quality from AoP viewpoint, modified from the original picture (Bulle et al., 2020). Yellow arrows present emissions and resources contributing stair to damage pathways and regionalized impact categories are marked with red.

At the midpoint and damage levels, impact scores for an impact category in an emitting region are calculated based on elementary flows (resource or emission), emitting compartments, and specific CFs for each elementary flow and impact category within a given emitting compartment and region. The sum of the impact scores provides the overall impact score at either the midpoint or damage level for the impact category. CFs are first calculated at the native level and then aggregated to the regional level (country, continent, or global). To derive the overall damage on the AoP, impact categories at damage level are summed into ecosystem quality. For resource-related impact categories CF are derived using extraction-consumption-competition-adaptation approach. The unit for ecosystem quality is the potentially disappeared fraction of species over a given area during predefined time ($PDF \cdot m^2 \cdot y$). (Bulle et al., 2020.)

At midpoint level the timeline is infinity for all indicators except climate change where both shorter-term indicator (GWP100) and long-term indicator (GTP100) are considered. At damage level, defined time horizons are used, and shorter-term impacts (100 years) are considered in all impact categories, in addition long-term impacts are available for climate change (500 years), freshwater ecotoxicity (infinity), and marine acidification (500 years). (Bulle et al., 2020.)

4.4.4 Review of selected methods

The study by Damiani et al. (2023) focused on the number of pressures, ecosystem types, taxonomic group coverage, EBV's, and fundamental biodiversity aspects when comparing different biodiversity impact assessment methods. It particularly emphasizes the ability to assess impacts on biodiversity in the context of LCA.

The methods selected in this study consider three out of five main drivers of biodiversity loss: land use and change of use, climate change, and pollution, excluding overexploitation and invasive species from consideration. These impact assessment methods also cover the largest number of pressures among those assessed in the Damiani et al. (2023) study. All three impact assessment methods consider land use, water use, GHG emissions, toxic emissions, acidifying emissions, other air emissions, and nutrient emissions, while excluding invasive species and overexploitation from the pressures included in the assessment. Currently, invasive species and overexploitation are not captured in any of the operational LCIA methods.

All selected methods result in the potentially disappeared fraction of species and consider impacts on terrestrial, freshwater, and marine ecosystems. In the case of acidification, IW considers impacts on both terrestrial and freshwater acidification, while RE and LC consider only terrestrial acidification. For eutrophication, RE considers all three ecosystem types, while LC and IW focus only on freshwater and marine ecosystems.

There are differences in taxonomic group coverage among molluscs, cnidarians, annelids, echinoderms, chlorophyta, rhodophyta, heterokontophyte, and fungi. Among EBV's, all three methods consider only community composition, leaving out genetic composition, species populations, species traits, ecosystem function, and ecosystem structure. Table 2 combines the coverage of biodiversity drivers and associated LCIA categories, taxonomic groups, and EBV classes by LCIA methods under assessment.

Table 2. Coverage of biodiversity drivers and associated LCIA impact categories, taxonomic groups, and EBV classes by LCIA method (Damiani et al., 2023; Sanyé-Mengual et al., 2023)

		ReCiPe 2016	LC-Impact	Impact World+
Driver	**Land use and change of use**			
	Land occupation	x	x	x
	Land transformation	x	x	x
	Climate change			
	Climate change	x	x	x
	Environmental Pollution			
	Ecotoxicity, terrestrial	x	x	
	Ecotoxicity, freshwater	x	x	x
	Ecotoxicity, marine	x	x	
	Acidification, terrestrial	x	x	x
	Acidification, freshwater			x
	Photochemical ozone formation	x	x	
	Eutrophication, terrestrial		x	
	Eutrophication, freshwater	x	x	x
	Eutrophication, marine	x	x	x
	Thermal pollution			x
	Ionizing radiation			x
	Other			
	Water use	x	x	x
Taxonomic groups	Mammals	x	x	x
	Birds	x	x	x
	Amphibians	x	x	x
	Reptiles	x	x	x
	Arthropods	x	x	x
	Bony fishes	x	x	x
	Cartilaginous fishes			
	Plants	x	x	x
	Molluscs		x	x
	Cnidarians		x	x
	Annelids		x	x
	Echinoderms		x	x
	Chlorophyta			x
	Rhodophyta			x
	Heterokontophyte			x
	Fungi	x		
	Others	x	x	x
EBV classes	Genetic composition			
	Species populations			
	Species traits			
	Community composition	x	x	x
	Ecosystem function			
	Ecosystem structure			

All the three methods selected in this study were also compared in the study by Sanyé-Mengual et al. (2023) where biodiversity impacts of EU consumption were assessed. As part

of the study, the contribution of impact categories to the overall biodiversity impact was analysed. It was noted that in LC, ecotoxicity dominates the overall result, covering 95.2% of the total impact on biodiversity. However, it is important to mention that the ecotoxicity categories were converted to PDF·y as presented in Chapter 4.4.2, for evaluation alongside the other impact categories, which may introduce uncertainty into the comparison. If ecotoxicity is not counted, climate change would have the highest contribution, covering 55%, followed by land use at 22%, and water use at 15%. In RE and IW ecotoxicity accounted for only 2% and 0.5%, respectively. In RE, climate change is the main contributor, covering 43%, followed by land use at 32% and acidification at 17%. In IW, land use is the primary contributor, and covers 61% of the overall biodiversity impact, followed by climate change at 24% and acidification at 13%.

5 LCA based biodiversity impact assessment of HDPE granulate production and comparison of EoL treatment options

In this chapter, a comparative LCA based BIA is conducted to assess impacts on biodiversity, and more specifically the potentially disappeared fraction of species due granulate production and EoL treatment of HDPE. Three scenarios are formed, and each assess the raw material acquisition and processing to produce HDPE granulates and one EoL treatment method. The following EoL treatment options are compared: combination of incineration with energy recovery and landfilling, mechanical recycling, and chemical recycling. The LCA follows ISO 14040 and 14044 standards. This chapter is divided into four sub-chapters: goal and scope definition, life cycle inventory analysis, biodiversity impact assessment, and result interpretation.

5.1 Goal and scope definition

In this BIA, the goal is to quantify the biodiversity impact of virgin HDPE granulate production and compare EoL treatment options by forming three scenarios for comparison. Raw material extraction and processing phase is similar in each scenario, but three different EoL treatment methods for separately collected post-consumer HDPE plastic waste is conducted, one in each scenario. While post-consumer plastic waste consists of various plastic types, this study focuses exclusively on the HDPE waste stream, assuming that the collected waste is entirely HDPE. The EoL treatment methods under assessment are combination of incineration with energy recovery and landfill, mechanical recycling, and chemical recycling. The study combines the raw material extraction and processing and EoL phases, referred to as cradle-to-gate with EoL. The processing of HDPE granulates into plastic products and the use phase of those products are excluded from this study. This limitation is made to concentrate on the production of plastic granulates and their EoL treatment, rather than on plastic products and their usage. It can be assumed that the use phase of HDPE granulates is similar across all scenarios, allowing for a valid comparison.

LCA-based BIA is an LCA study that quantifies impacts on the endpoint indicator (also referred as area of protection) ecosystem quality. The BIA conducted in this study is based

on secondary data from the Ecoinvent (v3.10) LCI database and scientific literature, with data limited to the EU region. The assessment is performed using SimaPro software (v9.6.0.1), and LCIA is conducted using three different impact assessment methods: ReCiPe 2016, LC-Impact, and IMPACT World +. As previously stated, an internationally agreed BIA method does not exist. Therefore, three different methods are implemented, and the factors influencing the results are compared. The methodologies of the LCIA methods used were detailed in the sub-chapters under Chapter 4.4.

In this study, the recycling processes are considered as EoL treatment options. In addition to producing recycled granulates, both recycling systems under assessment generate valuable by-products. This creates an allocation problem, which involves determining how to divide the environmental burden between the product and the by-product, as well as how to allocate the environmental burden across the different life cycles of the cycled plastic material. This allocation problem is addressed by limiting the study to one life cycle and applying a system expansion approach to incorporate the by-products into the assessed systems. The functional units in this study are the amount of HDPE granulates, electricity, and heat, with the reference flow being 1 kg of virgin HDPE granulates. All functional units must be produced in each scenario; for example, in Scenario 1, virgin HDPE granulates need to be produced at the EoL to fulfill the functional units. Similar system expansion approach was used in the study by Uusitalo et al. (2023), to ensures the comparability of the scenarios.

Each of the three scenarios has a similar production phase to produce virgin HDPE granulates using conventional methods from fossil-based raw materials. Virgin HDPE granulates are produced through the polymerization of ethylene. Ethylene is derived from naphtha and natural gas using the steam cracking process, with naphtha being the primary raw material. Naphtha, which consist mainly of C_5-C_{12} olefins, is derived from crude oil. (PlasticsEurope, 2014.) For the production phase, the eco-profile of HDPE granulate production provided by Plastics Europe is utilized. This eco-profile offers a cradle-to-gate model for HDPE production, limiting the life cycle to the phase where HDPE granulates are produced. The eco-profiles are based on average data from Europe.

After use, HDPE is discarded as post-consumer plastic waste, collected, and transported to the EoL treatment. It is assumed that no losses occur in the use phase and that 1 kg of HDPE plastic waste re-enters the system under assessment. In Scenario 1, HDPE plastic waste is divided between incineration and landfilling, with 77% transported to an incineration plant

for energy recovery and 23% sent to landfill. This division is based on data from Plastics Europe's (2022) plastic fact sheet regarding the shares of different treatment methods used for separately collected post-consumer plastic waste in the EU region. In Scenario 2, plastic waste is treated using mechanical recycling to produce virgin-like HDPE granulates. In scenario 3, plastic waste is treated using chemical recycling to produce virgin like HDPE granulates and heat from the by-products. Assumptions and limitations related to each scenario are presented in the inventory analysis phase. Figure 9 present the system boundary under assessment.

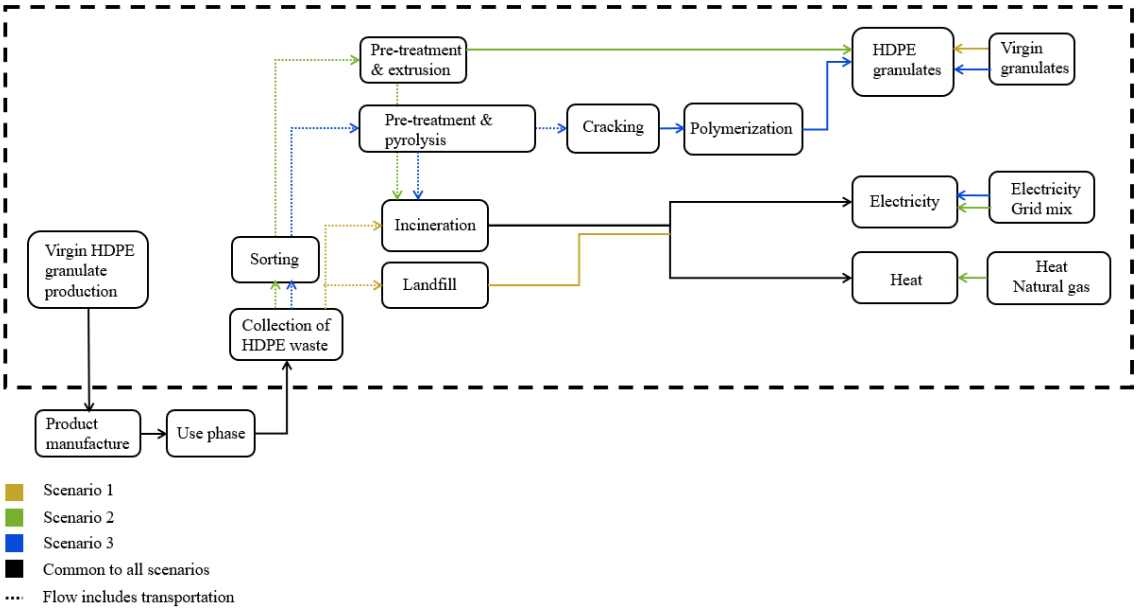

Figure 9. System boundary of the study.

5.2 Life cycle inventory analysis

In this chapter, inventory data for each unit process under assessment is presented. This chapter also outlines the possible calculations to derive the data and discusses limitations and assumptions related to the processes.

5.2.1 Virgin material production

Virgin material production is accounted using Plastics Europe's Eco-profile for HDPE virgin granulate production. Eco-profile datasets are available in the Ecoinvent database and

dataset “Polyethylene, high density, granulate {RER}| polyethylene production, high density, granulate | Cut-off, U“ is used. Eco-profiles are designed to serve as cradle-to-gate building blocks for LCA studies and comply with the requirements of ISO 14040 and 14044 standards. (PlasticsEurope, 2014.)

The cradle-to-gate system boundary encompasses the following processes: extraction of non-renewable resources, cultivation and harvesting of renewable sources, beneficiation or refining, transfer and storage, recycling of waste or secondary materials for use in production, energy conversion, all relevant transportations processes, production processes, and management of production waste streams and associated emissions. Capital goods, such as the construction of plants and equipment, as well as the maintenance of plants, vehicles, and machinery, are outside the LCI system boundaries. (PlasticsEurope, 2014.) However, the used Ecoinvent dataset includes construction of the chemical factory, which is estimated based on literature data by the industry. The construction activity includes materials, land occupation, land transformation, and decommission.

For production processes and on-site energy supply, primary data were collected. The LCI data reflects the production mix of technology utilized in the specific region as provided by the participating producers. Out of Europe’s 90 polyethylene production plants, 52 provided data, with 21 specifically contributing data for HDPE production. The collected primary data is supplemented with secondary data from background processes, such as grid electricity supply. (PlasticsEurope, 2014.)

Due to the multifunctional nature of the chemical and plastics industries, allocation procedures are used in the Eco-profile. Production sites that produce more than one polymer type with the same installation, allocate expenses and emissions to different polymer types based on mass. (PlasticsEurope, 2014.)

Steam cracking of liquid or gaseous feeds results in several products. In this case, the feedstock input is allocated by mass to all products leaving the cracking plant, and all the other inputs and outputs, such as energy input and emissions, are allocated by mass to the High Value Chemicals (HVCs), including ethene, propene, and purified hydrogen. Internal recycling of the cracking process is modelled as a closed-loop and does not lead to additional products. (PlasticsEurope, 2014.)

Oil refineries also have many multi-output processes and products. In this context, allocation is done using the energy content of products. Since many of the products are used for energy applications, allocation by calorific value is preferred. Intention of the process is considered, and the allocation between products are made based it. For example, mainstream to the process is allocated for all products, but catalysts, energy etc. are allocated only to the desired products. Residues do not receive any burden. (PlasticsEurope, 2014.)

Yielded waste is treated within the system and this way environmental burden from EoL management is allocated to the system. Process waste leaving the system (<0.5 wt.%) with recycling potential is allocated using a cut-off approach, meaning it does not receive any burdens or credits. (PlasticsEurope, 2014.)

Inventory data for primary energy demand for 1 kg of HDPE is presented in Table 3. Energy content in polymer is gross calorific value and process energy is derived by calculating difference between primary energy demand and energy content of polymer.

Table 3. Primary energy demand per 1 kg of HDPE (PlasticsEurope, 2014).

Primary energy demand	**Quantity [MJ]**
Energy content in HDPE	46.2
Process energy	34.0
Total primary energy demand	80.2

Since the produced HDPE is derived from fossil-based raw materials, 99% of the primary energy resources used are non-renewable. A small fraction of 1% comes from renewable resources, and primarily from the national electricity grid mixes. Table 4 presents the contribution of each primary energy resource to the total primary energy demand for production of 1 kg of HDPE.

Table 4. Primary energy resources per 1 kg of HDPE (PlasticsEurope, 2014).

Primary energy resource input	**Total energy input [MJ]**	**Total mass input [MJ]**	**Feedstock energy input [MJ]**	**Fuel energy input [MJ]**
Coal	1.25	0.063		1.25
Oil	59.28	1.294	40.27	19.01
Natural gas	15.65	0.339	7.57	0.08
Lignite	0.64	0.060		0.64
Nuclear	2.57			2.57
Solar	0.37			0.37
Geothermic	0.26			0.26
Waves	0.03			0.03
Wind	0.11			0.11
Sub-total renewables	0.8	0.0	0.0	0.8
Sub-total non-renewables	79.4	1.8	47.8	31.5
Total	80.2	1.8	47.8	32.3

The production of HDPE requires water withdrawals. Table 5 presents the water consumption for producing 1 kg of HDPE. Water use are withdrawals from natural resources and water consumption refers to the amount of water that is not returned to the original water source.

Table 5. Gross water resources used to produce 1 kg of HDPE (PlasticsEurope, 2014).

Water balance	**Quantity [kg]**
Process water	1.6
Non-contact cooling water	72.1
Water use	**73.6**
Water discharge to same waterbody	41.70
Water consumption	**31.9**

Generated wastewater contains dissolved and suspended organic compounds, which are represented as biochemical oxygen demand (BOD) measured over five days (BOD_5). BOD measures biodegradable organic material, and a high BOD indicates a high content of organic matter in the wastewater, which can lead to oxygen depletion in water. The BOD_5 value to produce 1 kg of HDPE is 3.58×10^{-6} kg. Chemical oxygen demand (COD) which

refers to the number of oxidizable substances in wastewater, is 3.51 x 10^{-5} kg to produce 1 kg of HDPE. The total organic carbon (TOC) in the wastewater is 6.48 x 10^{-6} kg. The TOC value refers to the total amount of carbon bound to organic compounds.

The production of HDPE also generates solid wastes. In Table 6, generated amounts of solid waste are categorized under hazardous and non-hazardous waste and placed under end-of-life treatment method.

Table 6. Waste generation from production of 1 kg of HDPE (PlasticsEurope, 2014).

	Incineration [kg]	Landfilling [kg]	Recovery [kg]	Unspecified [kg]	Total [kg]
Hazardous	4.26 x 10^{-4}	1.50 x 10^{-5}	3,68 x 10^{-4}	1.21 x 10^{-4}	9.30 x 10^{-4}
Non-hazardous	5.49 x 10^{-6}	1.48 x 10^{-4}	7,95 x 10^{-4}	3.34 x 10^{-4}	1.28 x 10^{-3}

5.2.2 Incineration and landfilling

In the Scenario 1, the EoL treatment combines incineration with energy recovery and sanitary landfilling. 73% of the collected HDPE waste is treated in municipal waste incineration plant with energy recovery and 23% is treated in sanitary landfill as part of communal waste. Incineration is accounted for using Ecoinvent dataset “Waste polyethylene {CH}| treatment of waste polyethylene, municipal incineration FAE | Cut-off, U”. Selected incineration process is controlled combustion with fly ash extraction (FAE). The data set includes infrastructure, waste-specific energy demand, and net energy generation. Collection and transportation of the waste are excluded. The dataset has been modelled for Switzerland, and it is assumed to describe the average municipal incineration process in Europe in an adequate level. Net energy production from 1 kg of incinerated polyethylene plastic waste is 4.98 MJ electric energy and 9.62 MJ thermal energy.

Transportation is estimated based on data from Finland. Tilastokeskus (2023) provides data on the average transportation distances and loading rates in domestic traffic by commodity type. Average transportation distance for household waste is 93 km. Transportation is accounted for using Ecoinvent dataset “Transport, freight, lorry >32 metric ton, EURO5 {RER}| market for transport, freight, lorry >32 metric ton, EURO5 | Cut-off, U”. Key inputs and outputs of incineration are presented in Table 7.

Table 7. Inventory data of HDPE waste incineration. Reference flow 1 kg virgin HDPE granulates.

Incineration	Quantity	Unit	Source
Inputs			
HDPE waste	0.77	kg	
Heat	0.17	MJ	Ecoinvent
Transportation	0.072	tkm	(Tilastokeskus, 2023)
Outputs			
Heat	7.41	MJ	Ecoinvent
Electricity	3.83	MJ	Ecoinvent

Landfilling is accounted for using Ecoinvent dataset "Waste polyethylene {CH}| treatment of waste polyethylene, sanitary landfill | Cut-off, U". It includes landfill facilities and release of waste-specific emissions. The emissions comprise air emissions from landfill gas and water emissions due leachate, and further to ground water. Collection and transportation of the waste are excluded from the dataset. For 1 kg of polyethylene landfilled, utilization of landfill gases generates 0.023 MJ of electric energy and 0.012 MJ of thermal energy. The overall degradability of waste during 100 years is 1%, and amount of waste remaining in the landfill after long time period is 0.0018 kg. Transportation is accounted for using same distance and dataset as in incineration. Table 8 presents key inputs and outputs for landfilling.

Table 8. Inventory data of HDPE waste landfilling. Reference flow 1 kg virgin HDPE granulates.

Landfill	Quantity	Unit	Source
Inputs			
HDPE waste	0.23	kg	
Transportation	0.021	tkm	(Tilastokeskus, 2023)
Outputs			
Heat	0.0026	MJ	Ecoinvent
Electricity	0.0053	MJ	Ecoinvent
Remaining waste mass	0.00044	kg	Ecoinvent

5.2.3 Mechanical recycling

In the Scenario 2, EoL treatment of HDPE waste is done using mechanical recycling. The EoL treatment includes collection and transportation of the waste, sorting, pre-treatment, and thermal extrusion. Sorting of waste is accounted for using Ecoinvent dataset "Waste polyethylene, for recycling, sorted {Europe without Switzerland}| treatment of waste polyethylene, for recycling, unsorted, sorting | Cut-off, U" which includes transportation, waste containers, sorting facilities, and energy production. Remaining material residue is incinerated with energy recovery and accounted for using Ecoinvent dataset "Waste polyethylene {Europe without Switzerland}| treatment of waste polyethylene, municipal incineration | Cut-off, U". Transportation distance and type for material residue from sorting plant to incineration is assumed to be same than in the case of landfilling and incineration in Scenario 1. Key inputs and outputs of waste sorting is presented in Table 9.

Table 9. Inventory data of HDPE waste sorting. Reference flow 1 kg virgin HDPE granulates.

Sorting	Quantity	Unit	Source
Inputs			
HDPE waste	1	kg	Ecoinvent
Electricity	0.016	kWh	Ecoinvent
Diesel	0.062	MJ	Ecoinvent
Propane	0.016	MJ	Ecoinvent
Outputs			
HDPE waste (sorted)	0.905	kg	Ecoinvent
Material residue	0.095	kg	Ecoinvent

After sorting, the waste is transported to recycling facility where pre-treatment and mechanical recycling take place. Transportation between sorting facility and recycling facility is estimated based on data from Finland. According to Tilastokeskus (2023) average transportation distance for waste to recycling is 160 km. Transportation is accounted for using Ecoinvent dataset "Transport, freight, lorry >32 metric ton, EURO5 {RER}| market for transport, freight, lorry >32 metric ton, EURO5 | Cut-off, U" and is accounted for in the pre-treatment process.

Sorted waste arrives to the pre-treatment facility in compacted bales and requires shredding, separation, and washing. Pre-treatment is modelled based on electricity demand, diesel

consumption, and water consumption. Electricity demand and diesel consumption are estimated based on Martín-Lara et al. (2022) study, where life cycle assessment of mechanical recycling of post-consumer polyethylene films was conducted. The electricity demand covers bale opening, drying, pre-crushing, optical separation, compaction, crushing, washing, and drying. Electricity demand is accounted for using Ecoinvent dataset "Electricity, medium voltage {Europe without Switzerland}| market group for electricity, medium voltage | Cut-off, U." Diesel is consumed by bale holder and is accounted for using Ecoinvent dataset "Diesel, burned in building machine {GLO}| market for diesel, burned in building machine | Cut-off, U".

Water demand for washing process is estimated based on literature review, and 2 m^3/t of waste is selected (Zou et al., 2023; Martín-Lara et al. 2022). Water consumption is accounted for using Ecoinvent dataset "Tap water {Europe without Switzerland} | market for tap water | Cut-off, U" and for wastewater treatment using "Wastewater, average {Europe without Switzerland}| market for wastewater, average | Cut-off, U". The wastewater dataset is selected in accordance with WEEE (Waste Electrical and Electronic Equipment) dataset for wastewater in pre-treatment of mechanical recycling (Bleu Safran, 2020). WEEE is French life cycle inventory database and the referred LCI report is for production of mechanically recycled plastics (PP, PS, ABS) from WEEE plastics managed in France and regenerated in Europe. Due lack of data, it is assumed that all water used for washing is returned as wastewater, which is not correct in real case due evaporation.

Material residues is incinerated and accounted for using Ecoinvent dataset "Waste polyethylene {CH}| treatment of waste polyethylene, municipal incineration FAE | Cut-off, U". Transportation is accounted as in the case of material residue from sorting. Table 10 presents the inventory data of pre-treatment.

Table 10. Inventory data of pre-treatment for chemical recycling. Reference flow: 1 kg virgin HDPE granulates.

Pre-treatment	Quantity	Unit	Source
Inputs			
HDPE plastic waste (sorted)	0.91	kg	
Electricity	0.42	kWh	(Martín-Lara et al., 2022)
Tap water	1.81	kg	(Zou et al., 2023)
Diesel (Bale holder)	0.043	MJ	(Martín-Lara et al., 2022)
Transportation	0.084	tkm	(Tilastokeskus, 2023)
Outputs			
Wastewater	1.81	kg	
HDPE plastic waste (treated)	0.60	kg	(Jeswani et al., 2021)
Material residue	0.30	kg	(Jeswani et al., 2021)

In the next and final step, pre-treated HDPE waste is processed into HDPE granulates through thermal extrusion. Thermal extrusion is modelled based on electricity demand presented in Martín-Lara et al. (2022) study. Due lack of data, it is assumed that all material losses during pre-treatment and thermal extrusion occurred during the pre-treatment phase. Table 11 presents the inventory data of thermal extrusion.

Table 11. Inventory data of thermal extrusion. Reference flow 1 kg virgin HDPE granulates.

Thermal extrusion	Quantity	Unit	Source
Inputs			
HDPE plastic waste (treated)	0.60	kg	
Electricity	0.16	kWh	(Martín-Lara et al., 2022)
Outputs			
Recycled HDPE granulates	0.60	kg	(Jeswani et al., 2021)

5.2.4 Chemical recycling

In the scenario 3, treatment of HDPE waste is done using chemical recycling. As a process for chemical recycling, thermal pyrolysis is selected. As noted in the theory section, pyrolysis of plastic waste in commercial scale is still rare, and technology is constantly developing. For this reason, collected data is mainly from laboratory scale experiments

scaled up to a commercial size plant. In this study, chemical treatment process includes waste collection, sorting, pre-treatment, thermal pyrolysis, steam cracking, and polymerization.

Sorting phase is assumed to be same in both mechanical recycling and chemical recycling. In pre-treatment, electricity demand, diesel consumption, transportation, and quantity of washing water are assumed to be same as in mechanical recycling scenario, but larger share of the treated waste can be used as a feedstock for pyrolysis than in thermal extrusion. Material losses in pre-treatment are estimated based on relation between waste material input and sorted output in the study by Jeswani et al. (2021). The study examined life cycle impacts of chemical recycling via pyrolysis in comparison with mechanical recycling and energy recovery. Waste feedstock in the study were sorted mixed plastic (polyethylene, polypropylene, polystyrene) from household waste. It is assumed that this adequately reflects the case in this study. Inventory data of pre-treatment of plastic waste for thermal pyrolysis is presented in Table 12.

Table 12. Inventory data of pre-treatment for chemical recycling. Reference flow: 1 kg virgin HDPE granulates.

Pre-treatment	Quantity	Unit	Source
Inputs			
HDPE plastic waste (sorted)	0.91	kg	
Electricity	0.42	kWh	(Martín-Lara et al., 2022)
Tap water	1.81	kg	(Zou et al., 2023)
Diesel (Bale holder)	0.043	MJ	(Martín-Lara et al., 2022)
Transportation	0.084	tkm	(Tilastokeskus, 2023)
Outputs			
Waste water	1.81	kg	
HDPE plastic waste (treated)	0.82	kg	(Jeswani et al., 2021)
Material residue	0.09	kg	(Jeswani et al., 2021)

After pre-treatment, plastic waste is fed into the thermal pyrolysis process. At a temperature of 350 °C, the plastic waste is converted into pyrolysis oil, gas, and char. According to a pyrolysis study by Ahmad et al. (2015), thermal pyrolysis of HDPE waste at this temperature yields products comprising 80.88% liquid, 17.24% gas, and 1.88% solid residue (char) from the total yield. The pyrolysis experiment was conducted at the laboratory scale, out in a steel microreactor under a nitrogen environment. Nitrogen environment increases liquid yield and decreases char production. Used nitrogen flow were not found from the Ahmad et al. (2015)

study and for this reason the nitrogen flow is estimated based on study by Pan et al. (2021). The study investigated pyrolysis of waste polyethylene to produce liquid fuel. In the study, three nitrogen flow rates were tested: 0.02 l/min, 0.06 l/min, and 0.1 l/min. Gas flow rate of 0.02 l/min were noted to have best impact on liquid yield. Based on this, nitrogen flow rate of 0.02 l/min is selected for this study. Required nitrogen gas demand is calculated based on yearly operating time and waste processing capacity. Key parameters for pyrolysis process are presented in the Table 13.

Table 13. Key parameters for chemical recycling.

Technical parameters		
Yield pyrolysis oil	80.88%	(Ahmad et al., 2015)
Yield pyrolysis gas	17.24%	(Ahmad et al., 2015)
Yield char	1.88%	(Ahmad et al., 2015)
Pyrolysis temperature	350 °C	(Ahmad et al., 2015)
Nitrogen flow	0.02 l/min	(Pan et al., 2021)
Density, nitrogen gas	1.2506 g/l (0 °C, 1013 mbar)	
Operational parameters		
Operating time	7500 h/y	(Stallkamp et al., 2024)
Capacity	100 kt PW/y	(Jeswani et al., 2021)

The electricity demand in the pyrolysis process is estimated based on a study by Jeswani et al. (2021), using the relationship between reported electricity demand and plastic waste input. Use of nitrogen gas is accounted for using Ecoinvent dataset "Nitrogen, liquid {RER}| industrial gases production, cryogenic air separation | Cut-off, U". Inventory data of thermal pyrolysis is presented in Table 14.

Table 14. Inventory data of thermal pyrolysis process. Reference flow: 1 kg virgin HDPE granulates.

Thermal pyrolysis	**Quantity**	**Unit**	**Data source**
Inputs			
HDPE plastic waste (treated)	0.82	kg	
Nitrogen, gas	$9.18 \cdot 10^{-8}$	kg	(Aladin et al., 2021)
Electricity	0.52	kWh	(Jeswani et al. 2021)
Outputs			
Liquid fraction	0.66	kg	(Ahmad et al., 2015)
Gas fraction	0.14	kg	(Ahmad et al., 2015)
Char	0.02	kg	(Ahmad et al., 2015)

Analysis of the pyrolysates in the study by Ahmad et al. (2015) showed that the liquid fraction from HDPE waste meets the fuel-grade qualities and is suggested to be a blend of gasoline and diesel-range hydrocarbons. This liquid fraction can be used to replace naphtha in the production of ethylene, which is required for manufacturing HDPE granulates.

Derived gas fraction and char is used to produce heat through combustion to meet the thermal energy demand of the pyrolysis process. Due lack of data, emissions from combustion are excluded from the study. It is also assumed that produced heat covers the thermal energy demand of the pyrolysis process, and no excess energy is produced. Lower heating value (LHV) for char is around 10 MJ/kg (Jeswani et al., 2021). LHV for gas fraction is typically between 22-30 MJ/m^3 (Al-Salem et al., 2009). Based on literature, excess heat is often produced, and it can be provided to a district heating network. In a study by Fivga and Dimitrou (2017), 7 kg of char and 7.85 kg of gas generated 156.76 kW of thermal energy through combustion from which 41.16 kW were used in the pyrolysis process and 115.6 kW were excess energy.

In the next step, liquid fraction from pyrolysis process is fed into a steam cracker to produce ethylene. Based on literature review, purification process is not required for liquid yield from pyrolysis of HDPE waste, as based on feedstock composition analysis no chlorine or oxygen enters to the pyrolysis process (Das et al., 2022; Kumar and Singh, 2011; Benavides et al., 2017). Based on this, it is assumed that the produced liquid fraction meets the quality requirements for steam cracker strait from the pyrolysis process. Relation between pyrolysis oil feedstock and amount of produced ethylene is based on study by Jeswani et al. (2021) and determines the amount of ethylene produced in this study.

Steam cracker process is accounted for using Ecoinvent dataset for conventional ethylene production “Ethylene {RER}| unsaturated hydrocarbons production, steam cracking operation, average | Cut-off, U”, which is modified to describe the process in this study. The production of conventional naphtha feedstock is subtracted from the process. The conventional steam cracking process includes also other possible feedstocks like butane and natural gas, which are still included in the process after modification. This is due several direct air emissions from the cracking process that could not be directed to a specific feedstock. However, the share of naphtha from the total feedstocks is around 64%, so it is assumed that the impact of other feedstocks is minor. Key inputs and outputs are presented in Table 15.

Table 15. Inventory data of ethylene production. Reference flow 1 kg virgin HDPE granulates.

Steam cracking	**Quantity**	**Unit**	**Data source**
Inputs			
Pyrolysis oil (naphtha like)	0.66	kg	
Sodium hydroxide	0.003	kg	Ecoinvent
Nitrogen, liquid	0.006	kg	Ecoinvent
Electricity	0.065	kWh	Ecoinvent
Outputs			
Ethylene	0.54*	kg	(Jeswani et al. 2021)

*The mass of outputs is lower than inputs due waste and direct emission generation

As a last step, ethylene is used as a feedstock for polymerization process to produce HDPE granulates. Relation between ethylene input and HDPE granulate output is based on study by Jeswani et al. (2021) and determines the amount of HDPE granulates produced in this study. The polymerization process is accounted for using Ecoinvent dataset for conventional HDPE polymerization process "Polyethylene, high density, granulate {RER}| polyethylene production, high density, granulate | Cut-off, U" which is modified to describe the process in this study. Production of conventional ethylene is subtracted from the process. Main inputs and outputs are presented in Table 16.

Table 16. Inventory data of HDPE polymerization.

Polymerization	**Quantity**	**Unit**	**Data source**
Inputs			
Ethylene	0.54	kg	(Jeswani et al. 2021)
Heat	0.59	MJ	Ecoinvent
Electricity	0.23	kWh	Ecoinvent
Outputs			
HDPE granulates	0.53	kg	(Jeswani et al. 2021)

5.2.5 EoL substitution

To fulfill the functional unit's, production of HDPE granulates, heat, and electricity is required to meet the same production quantity at the EoL in each scenario. Largest amount of heat and electricity were produced in Scenario 1, setting the functional unit for heat at 7.41 MJ and for electricity at 3.84 MJ. This means that shortage in heat and electricity

production needs to be fulfilled in Scenarios 2 and 3. Largest quantity of HDPE granulates were produced in Scenario 2 setting the functional unit for HDPE granulates at 0.60 kg. Shortage in granulate production in Scenarios 1 and 3 are fulfilled by producing virgin HDPE granulates as in production phase.

The additional heat demand is accounted for using Ecoinvent dataset “Heat, district or industrial, natural gas {Europe without Switzerland}| market for heat, district or industrial, natural gas | Cut-off, U”. This dataset is selected as according to Eurostat (2024) data on heat generation by fuel type in 2022, natural gas accounted the largest share. Electricity production is accounted for using Ecoinvent dataset “Electricity, medium voltage {Europe without Switzerland}| market group for electricity, medium voltage | Cut-off, U”. Figure 10, 11, and 12 presents the material and energy flows in each scenario. Functional units and functions to be fulfilled are marked with green.

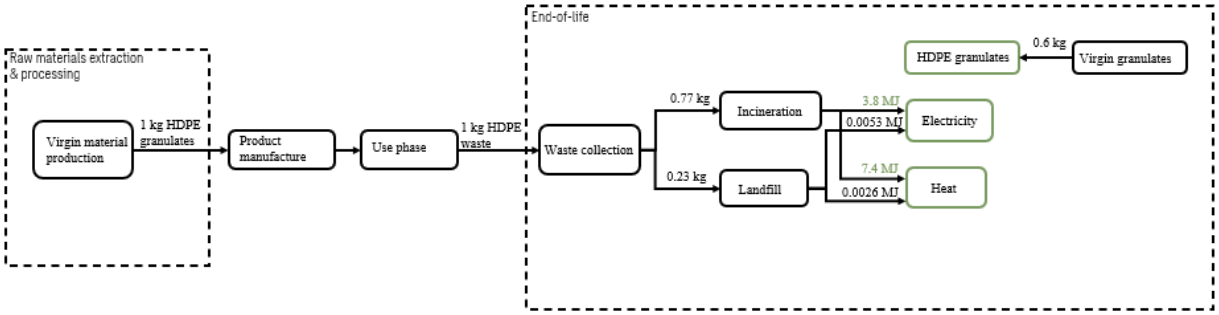

Figure 10. Material and energy flows in scenario 1. Functions to be fulfilled and functional units determined by this scenario are marked with green colour.

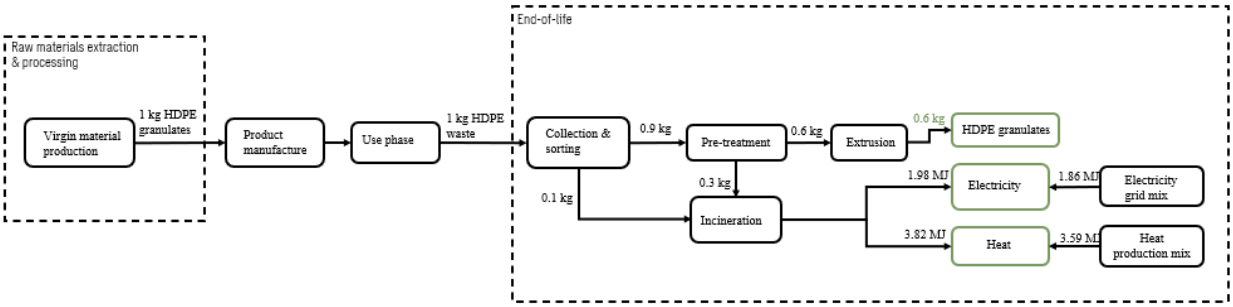

Figure 11. Material and energy flows in scenario 2. Functions to be fulfilled and functional unit determined by this scenario are marked with green colour.

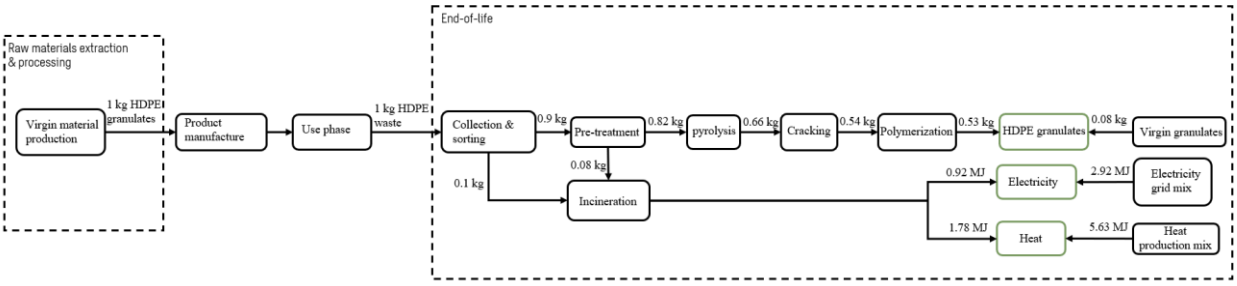

Figure 12. Material and energy flows in scenario 3. Functions to be fulfilled are marked with green colour.

5.3 Biodiversity impact assessment

This chapter presents the LCIA results separately for each method. The results open impacts of each scenario to potentially disappeared fraction of species. The results also examine the processes through which the impacts arise, and which impact categories contributes the most on the overall outcome.

5.3.1 ReCiPe 2016 results

In the ReCiPe method, the results directed to each ecosystem are aggregated into a single result within the method. The result for Scenario 1, where EoL treatment was divided between incineration with energy recovery and landfilling, is $2.54 \cdot 10^{-8}$ species·y and represents the scenario with the highest impact on potential species loss. Scenario 2, where mechanical recycling was selected as the EoL treatment option, shows the lowest impact, with result of $1.80 \cdot 10^{-8}$ species·y. Scenario 3, where chemical recycling was selected as the EoL treatment method, falls between the Scenarios 1 and 2, with a result of $2.24 \cdot 10^{-8}$ species·y. Figure 13 presents the results for each scenario and contribution of processes to the final results in accordance with the reference flow and functional units.

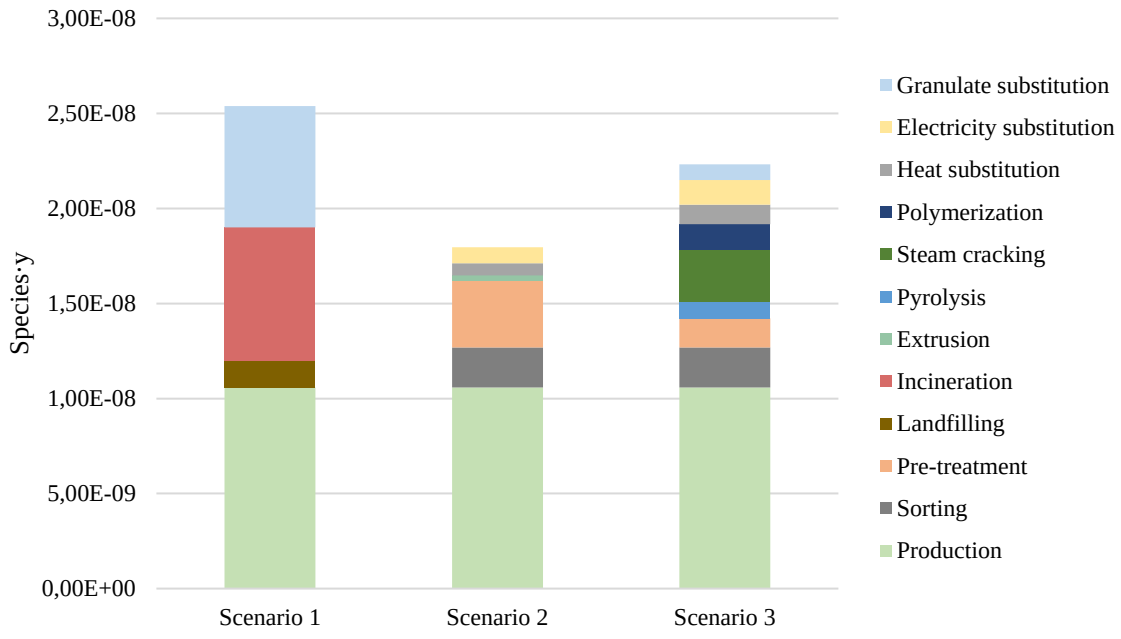

Figure 13. ReCiPe 2016 results for each scenario and contribution of each process to the result in accordance with reference flow and functional units.

In scenario 1, the life cycle was divided into four processes: production, where virgin HDPE granulates are produced; incineration with energy recovery; landfilling with landfill gas utilization; and granulate substitution to fulfil the functional units. Raw material extraction and processing, referred to as production, accounted for 41.7% of the total result, followed by incineration at 27.6%, granulate substitution at 25.2%, and landfilling at 5.5%. When divided into life cycle phases, production account for 41.7 %, followed by EoL treatment at 33.2%, and EoL substitution at 25.2%.

In Scenario 2, the life cycle was divided into six processes: production of virgin HDPE granulates; sorting of collected HDPE waste; pre-treatment of the waste to fulfil the quality requirements for extrusion; extrusion to produce recycled granulates; and heat and electricity substitution to fulfil the functional units. The share of production from the total result is 58.9%, followed by pre-treatment at 19.5%, sorting at 11.7%, electricity substitution at 4.7%, heat substitution at 3.6%, and extrusion at 1.6%. When divided into life cycle phases, the production account for 58.9%, followed by EoL treatment at 32.8%, and EoL substitution at 8.3%.

In Scenario 3, the life cycle was divided into nine processes: production of virgin HDPE granulates; sorting of collected HDPE waste; pre-treatment of the waste to fulfil the quality requirements for pyrolysis; pyrolysis to convert plastic waste into naphtha like liquid fraction; steam cracking to produce ethylene from naphtha like fraction; polymerization to produce recycled plastic granulates; and heat, electricity, and granulate substitution to fulfil the functional units. The share of production from the total result is 47.4%, followed by steam cracking at 12.3%, sorting at 9.4%, pre-treatment at 6.9%, polymerization at 6.1%, electricity substitution at 5.9%, heat substitution at 4.5%, pyrolysis at 3.8%, and granulate substitution at 3.7%. When divided into life cycle phases, the raw material extraction and processing account for 47.4%, followed by EoL treatment at 38.5%, and EoL substitution at 14.1%.

Behind the results are the impact categories contributing to the ecosystem quality. Figure 14 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and the contribution of processes in Scenario 1. It can be noted that the impacts of global warming to terrestrial ecosystems dominate the total impact on ecosystem quality and contribute the most to potential species loss with share of 71.5% of the total impact. The second largest contributors are freshwater eutrophication at 7.4% and terrestrial acidification

at 6.8%. The smallest contributors are water consumption impacts on freshwater ecosystems at 0.0001% and marine eutrophication at 0.001%. The impacts of water consumption on freshwater ecosystems arising from landfilling show negative results that indicate a positive impact on biodiversity. This may be due to the allocation procedure in the wastewater treatment of landfill leachate, which incorrectly suggests an increase in water due to the release of treated wastewater. Consequently, this process lowers the overall impact. Production, incineration, and granulate substitution contributes the most on global warming impact on terrestrial ecosystems, while landfilling have largest impact on freshwater eutrophication.

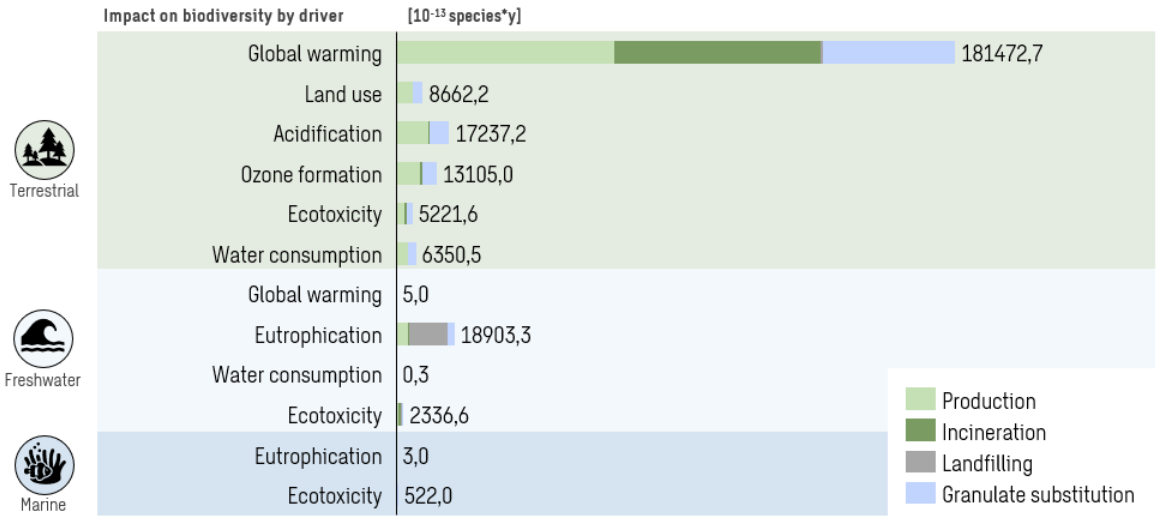

Figure 14. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 1.

Figure 15 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 2. It can be noted that impacts of global warming to terrestrial ecosystems dominate the total impact on ecosystem quality and contributes the most to potential species loss with share of 71.6% of the total impact. Second largest contributors are terrestrial acidification at 8.3% and ozone formation at 5.7%. Smallest contributors are water consumption impacts on freshwater ecosystems at 0.0001% and marine eutrophication at 0.001%. All processes contribute the most on global warming impacts on terrestrial ecosystems.

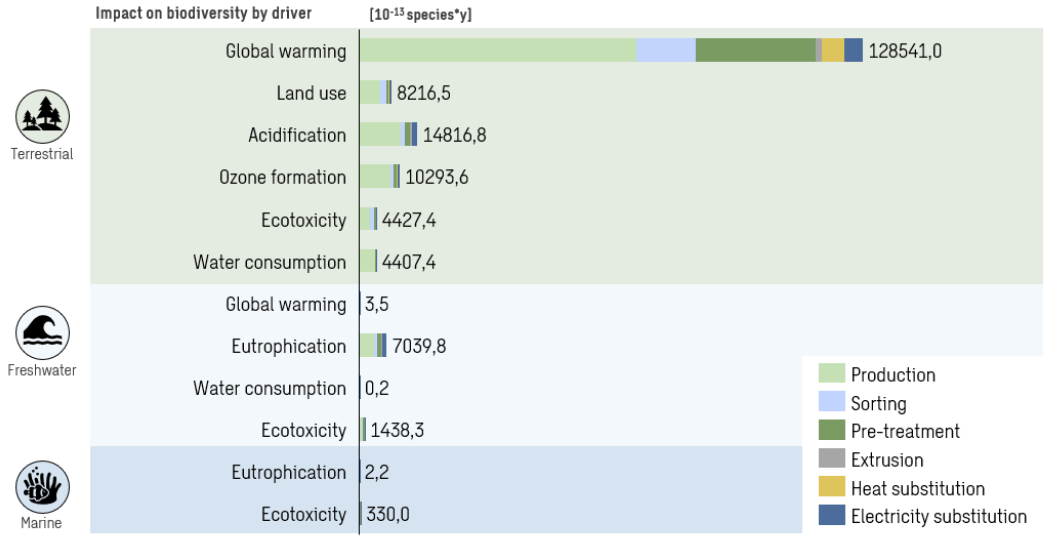

Figure 15. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 2.

Figure 16 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 3. It can be noted that impacts of global warming to terrestrial ecosystems dominates the total impact on ecosystem quality and contributes the most to potential species loss with share of 68.1% of total impacts. Second largest contributors are terrestrial acidification at 9.5% and ozone formation at 6.2%. Smallest contributors are water consumption impacts on freshwater ecosystems at 0.0002% and marine eutrophication at 0.001%. All processes contribute most on global warming impacts on terrestrial ecosystems.

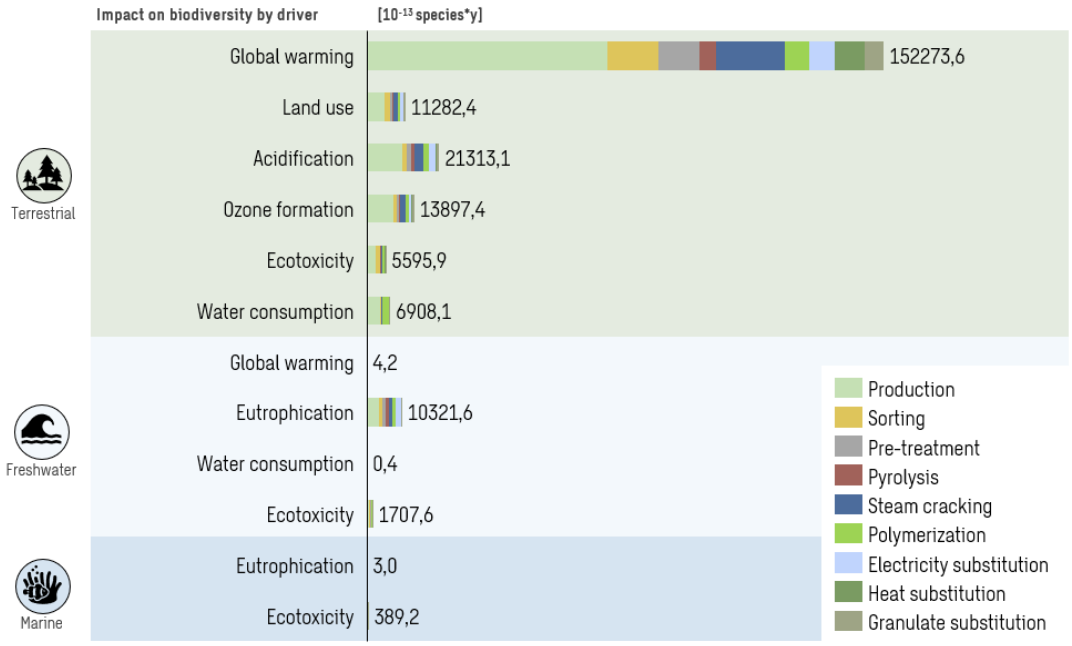

Figure 16. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 2.

5.3.2 LC-Impact results

In the LC-Impact method, the results are presented separately for terrestrial, freshwater, and marine ecosystems. As mentioned in the method description, results for ecotoxicity are presented in different unit than results for other impact categories. For this reason, ecotoxicity is calculated separately based on emission inventory derived from SimaPro. The CFs are converted to unit of PDF·y as presented in Equations 3, 4, 5, 6 and 7. Due uncertainty related to the conversion, and the previous knowledge about notably high contribution of ecotoxicity in LC method, results are presented with and without the impact category of ecotoxicity.

For Scenario 1, where EoL treatment was divided between incineration with energy recovery and landfilling, the results without ecotoxicity are $1.24 \cdot 10^{-14}$ PDF·y for terrestrial, $9.77 \cdot 10^{-15}$ PDF·y for freshwater, and $4.83 \cdot 10^{-19}$ PDF·y for marine ecosystems. This scenario represents the highest impact on potential species loss in terrestrial and freshwater ecosystems and the second highest in marine ecosystems. Scenario 2, where mechanical recycling was selected as the EoL treatment option, shows the lowest impact across all ecosystems, with results of $9.10 \cdot 10^{-15}$ PDF·y for terrestrial, $5.47 \cdot 10^{-15}$ PDF·y for freshwater, and $3.51 \cdot 10^{-19}$ PDF·y for marine ecosystem. Scenario 3, where chemical recycling was

selected as the EoL treatment method, falls between the Scenarios 1 and 2 for terrestrial and freshwater ecosystems and has the highest result for marine ecosystems, with results of $1.10 \cdot 10^{-14}$ PDF·y for terrestrial, $7.35 \cdot 10^{-15}$ PDF·y for freshwater, and $9.68 \cdot 10^{-19}$ PDF·y for marine.

When ecotoxicity is included, the results for Scenario 1 are $4.08 \cdot 10^{-11}$ PDF·y for terrestrial, $2.64 \cdot 10^{-12}$ PDF·y for freshwater, and $1.30 \cdot 10^{-11}$ PDF·y for marine. In Scenario 2, the results are $2.80 \cdot 10^{-11}$ PDF·y for terrestrial, $1.98 \cdot 10^{-12}$ PDF·y for freshwater, and $9.94 \cdot 10^{-12}$ PDF·y for marine. In Scenario 3, the results are $3.22 \cdot 10^{-11}$ PDF·y for terrestrial, $3.01 \cdot 10^{-12}$ PDF·y for freshwater, and $1.35 \cdot 10^{-11}$ PDF·y for marine. Now Scenario 1 has the highest impact on terrestrial ecosystems, and Scenario 3 on freshwater and marine ecosystems. Scenario 2 has the lowest impact on each ecosystem type. The impacts notably increased on every ecosystem type, especially on marine ecosystems and now lowest impacts in each scenario are assigned to freshwater ecosystems.

In LC-Impact, the result indicates the fraction of species that is in danger to go extinct if the pressure continues, for example, a result of 0.01 PDF implies that 1% of the global species pool is at risk to become extinct (Hujibregts et al., 2016). Figure 17 presents the results for each scenario in accordance with the reference flow and functional units. Results without ecotoxicity are presented on the left and with ecotoxicity on the right.

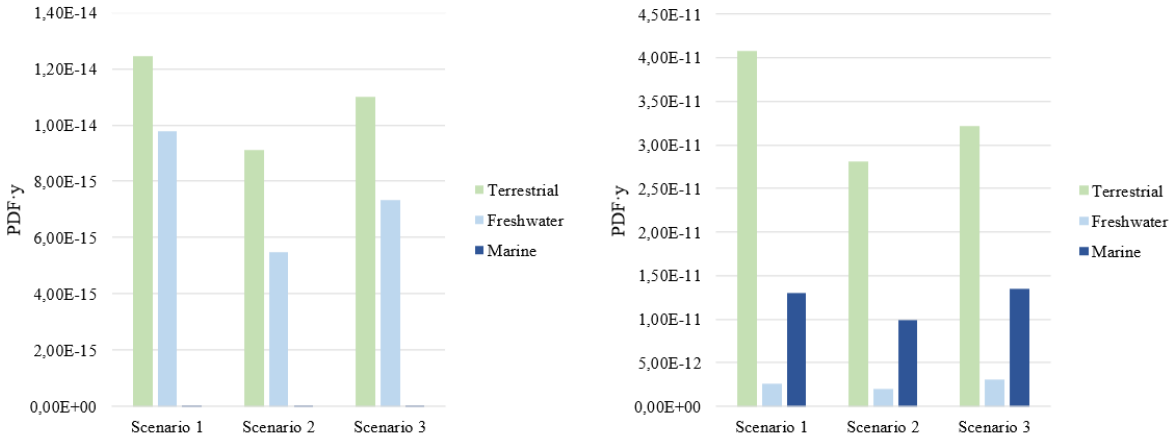

Figure 17. LC-Impact results for each Scenario in accordance with reference flow and functional units. Results without ecotoxicity on the left and with ecotoxicity on the right.

To obtain a single result for each scenario, Equation 1 is used to merge terrestrial, freshwater, and marine ecosystems. Figure 18 presents the merged results for each scenario and contribution of processes. Results without ecotoxicity are presented on the left and with

ecotoxicity on the right. Scenario 1 has the largest impact on ecosystem quality, followed by Scenario 3 and Scenario 2 has the lowest impact. Results without ecotoxicity are $1.09 \cdot 10^{-14}$ $PDF_{total} \cdot y$ for Scenario 1, $7.82 \cdot 10^{-15}$ $PDF_{total} \cdot y$ for Scenario 2, and $9.55 \cdot 10^{-15}$ $PDF_{total} \cdot y$ for Scenario 3. Results with ecotoxicity are $3.43 \cdot 10^{-11}$ $PDF_{total} \cdot y$ for Scenario 1, $2.37 \cdot 10^{-11}$ $PDF_{total} \cdot y$ for Scenario 2, and $2.74 \cdot 10^{-11}$ $PDF_{total} \cdot y$ for Scenario 3. Differences between results with and without ecotoxicity can be seen in process contribution, but differences between the overall results of scenarios are quite similar.

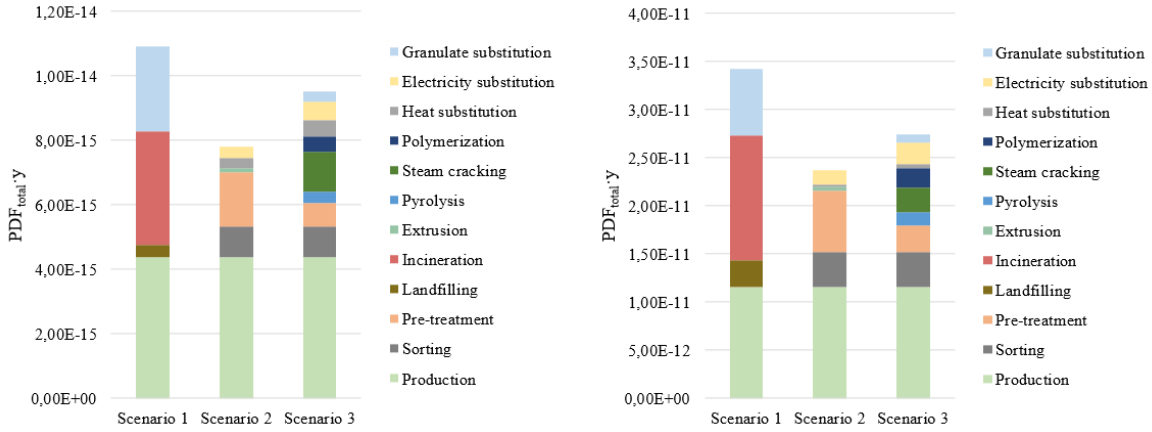

Figure 18.Merged LC-Impact results for each scenario and contribution of processes in accordance with reference flow and functional units. Results without ecotoxicity on the left and with ecotoxicity on the right.

In scenario 1, the life cycle was divided into four processes: production, where virgin HDPE granulates are produced; incineration with energy recovery; landfilling with landfill gas utilization; and granulate substitution to fulfil the functional units. The share of production from the total result without ecotoxicity is 40.1%, followed by incineration at 32.3%, granulate substitution at 24.2%, and landfilling at 3.4%. When divided into life cycle phases, raw material extraction and processing, referred to as production, account for 40.1 %, followed by EoL treatment at 35.7%, and EoL substitution at 24.2%. When ecotoxicity is included, incineration contributes the most with share of 37.9%, followed by production at 33.7%, granulate substitution at 20.3%, and landfilling at 8.1%. From the life cycle phases EoL treatment contributes the most with share of 46.0%, followed by production at 33.7%, and substitution at 20.3%.

In Scenario 2, the life cycle was divided into six processes: production of virgin HDPE granulates; sorting of collected HDPE waste, pre-treatment of the waste to fulfil the quality requirements for extrusion; extrusion to produce recycled granulates; and heat and electricity substitution to fulfil the functional units. The share of production from the total result is

55.9%, followed by pre-treatment at 21.9%, sorting at 12.0%, electricity substitution at 4.7%, heat substitution at 4.0%, and extrusion at 1.5%. When divided into life cycle phases, the raw material extraction and processing account for 55.9%, followed by EoL treatment at 35.4%, and EoL substitution at 8.6%. When ecotoxicity is included, production contributes the most with share of 48.7%, followed by pre-treatment at 26.8%, sorting at 15.3%, electricity substitution at 8.1%, heat substitution at 1.2%, and extrusion at 1.9%. From the life cycle phases production contributes the most with share of 48.7%, followed by EoL at 44.0%, and substitution at 7.2%.

In Scenario 3, the life cycle was divided into nine processes: production of virgin HDPE granulates; sorting of collected HDPE waste, pre-treatment of the waste to fulfil the quality requirements for pyrolysis; pyrolysis to convert plastic waste into naphtha like liquid fraction; steam cracking to produce ethylene from naphtha like fraction; polymerization to produce recycled plastic granulates; and heat, electricity, and granulate substitution to fulfil the functional units. The share of production from the total result is 45.8%, followed by steam cracking at 12.8%, sorting at 9.9%, pre-treatment at 7.8%, electricity substitution at 6.0 %, polymerization at 5.3%, heat substitution at 5.1%, pyrolysis at 3.9%, and granulate substitution at 3.6%. When divided into life cycle phases, the raw material extraction and processing account for 45.8%, followed by EoL treatment at 39.6%, and EoL substitution at 14.6%. When ecotoxicity is included, production contributes the most with share of 42.0%, followed by sorting at 13.2%, pre-treatment at 10.1%, steam cracking at 9.1%, electricity substitution at 8.1%, polymerization at 7.3%, pyrolysis at 5.2%, granulate substitution at 3.3%, and heat substitution at 1.6%. From the life cycle phases EoL treatment contributes the most with share of 44.9%, followed by production at 42.0%, and substitution at 13.0%.

Behind the results are the impact categories contributing to the ecosystem quality. Figure 19 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 1. The results are here considered by ecosystem type as defined in the methodology. Impacts on terrestrial ecosystems are mainly from climate change impacts which accounts 89.1%, followed by land stress at 6.3%, acidification at 4.5%, and ozone formation at 0.1%. Impacts on freshwater ecosystems are mainly from eutrophication with share of 51.6% and climate change at 35.3%, water stress accounts 13.1%. Impacts on marine ecosystems origin only from marine eutrophication impact category. The impacts of water stress arising from landfilling show negative results that

indicate a positive impact on biodiversity. This may be due to the allocation procedure in the wastewater treatment of landfill leachate, which incorrectly suggests an increase in water due to the release of treated wastewater. Consequently, this process lowers the overall impact. Production, incineration, and granulate substitution have largest impact on terrestrial ecosystems through global warming, while landfilling has largest impact on freshwater ecosystems through eutrophication.

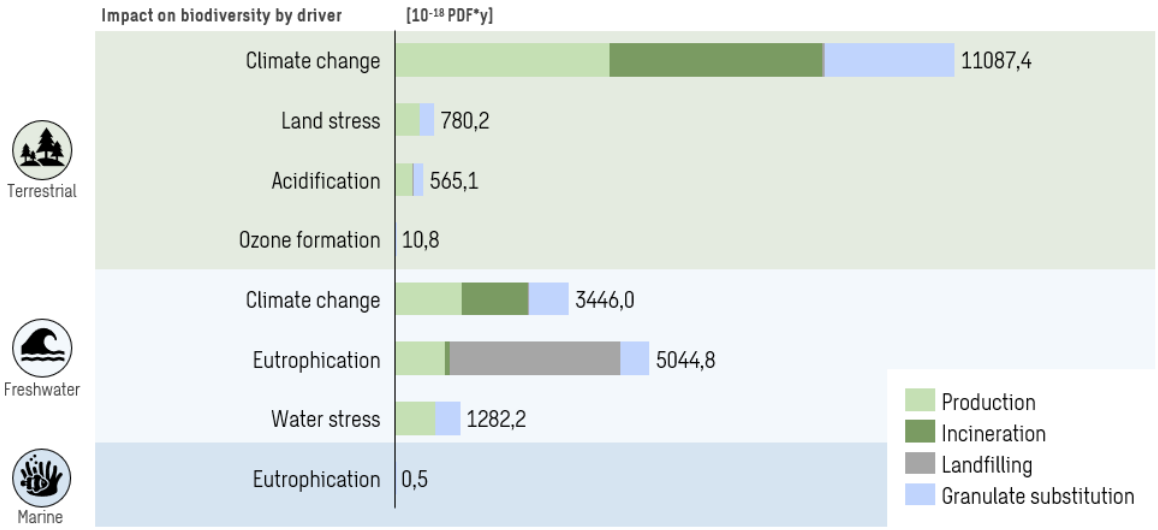

Figure 19. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 1.

When ecotoxicity is considered, it clearly dominates the contribution on each ecosystem type, and other impact categories has only a minor contribution. The contribution of ecotoxicity accounts 99.97% of the total impact assigned to terrestrial ecosystems, 99.63% of impacts on freshwater ecosystems, and almost full 100% on marine ecosystems. Figure 20 presents contribution in impact categories in Scenario 1, when ecotoxicity is included.

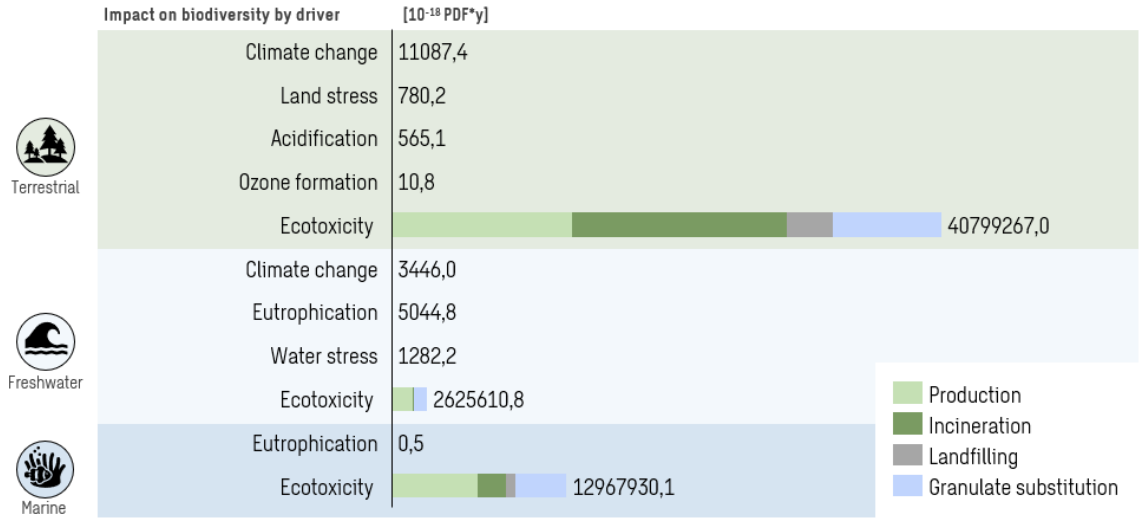

Figure 20. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 1 when ecotoxicity is considered.

Figure 21 presents all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 2. The results are here considered by ecosystem type as defined in the methodology. Impacts on terrestrial ecosystems are mainly from climate change impacts which accounts 84.4%, followed by land stress at 9.5%, acidification at 6.1%, and ozone formation at 0.1%. Climate change also accounts largest share of the impacts on freshwater ecosystems with share of 41.0%, but eutrophication is also a big contributor with share of 36.4%, and water stress accounts 22.6%. Impacts on marine ecosystems origin only from marine eutrophication impact category. All the processes have largest impact on terrestrial ecosystems through climate change.

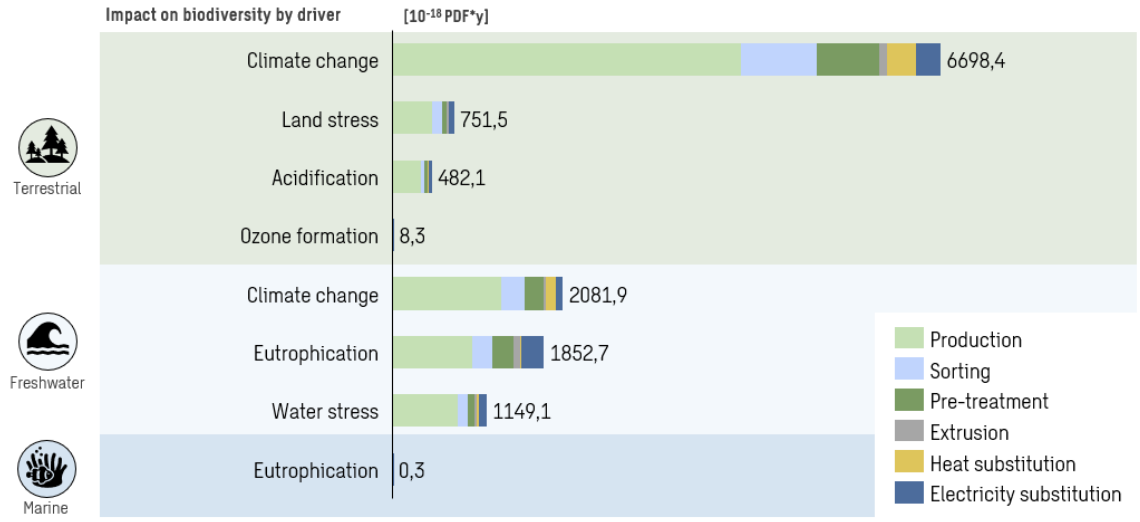

Figure 21. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 2.

When ecotoxicity is considered, it clearly dominates the contribution on each ecosystem type, and other impact categories has only a minor contribution. The contribution of ecotoxicity accounts 99.97% of the total impact assigned to terrestrial ecosystems, 99.74% of impacts on freshwater ecosystems, and almost full 100% on marine ecosystems. Figure 22 presents contribution in impact categories in Scenario 1, when ecotoxicity is included.

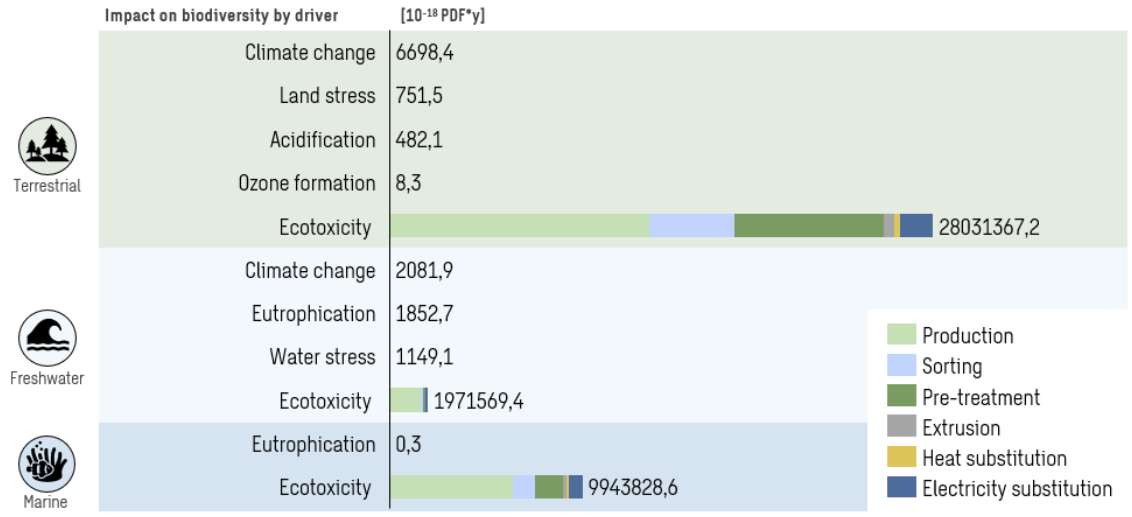

Figure 22. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 2 when ecotoxicity is considered.

Figure 23 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 3. The results are here considered by ecosystem type as defined in the methodology. Impacts on terrestrial ecosystems are mainly from climate change impacts which accounts 84.1%, followed by land stress at 9.5%,

acidification at 6.3%, and ozone formation at 0.1%. Climate change also accounts largest share of the impacts on freshwater ecosystems with share of 39.3%, but eutrophication is also a big contributor with share of 37.5%, and water stress accounts 23.2%. Impacts on marine ecosystems origin only from marine eutrophication impact category. All of the processes have largest impact on terrestrial ecosystems through climate change.

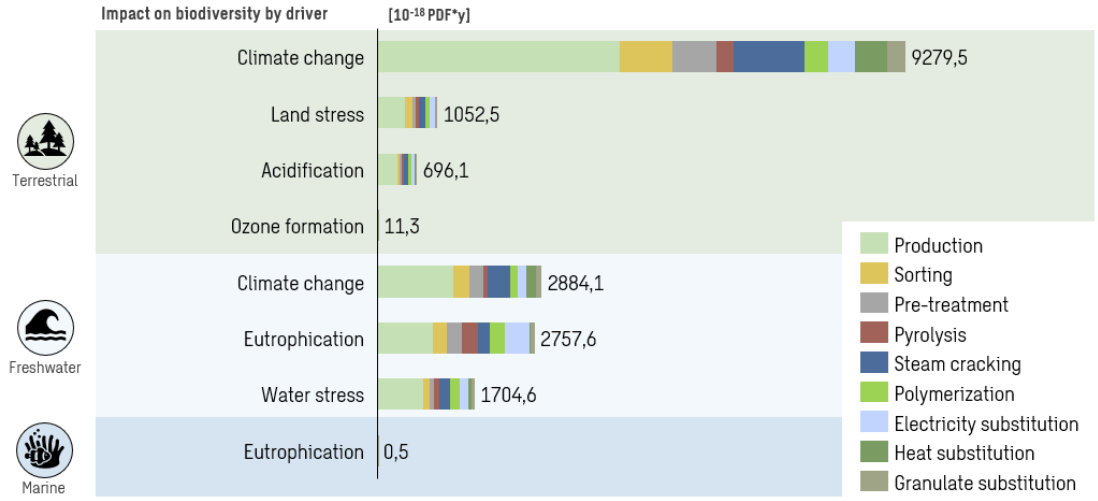

Figure 23. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 3

When ecotoxicity is considered, it clearly dominates the contribution on each ecosystem type, and other impact categories has only a minor contribution. The contribution of ecotoxicity accounts 99.97% of the total impact assigned to terrestrial ecosystems, 99.76% of impacts on freshwater ecosystems, and almost full 100% on marine ecosystems. Figure 24 presents contribution in impact categories in Scenario 1, when ecotoxicity is included.

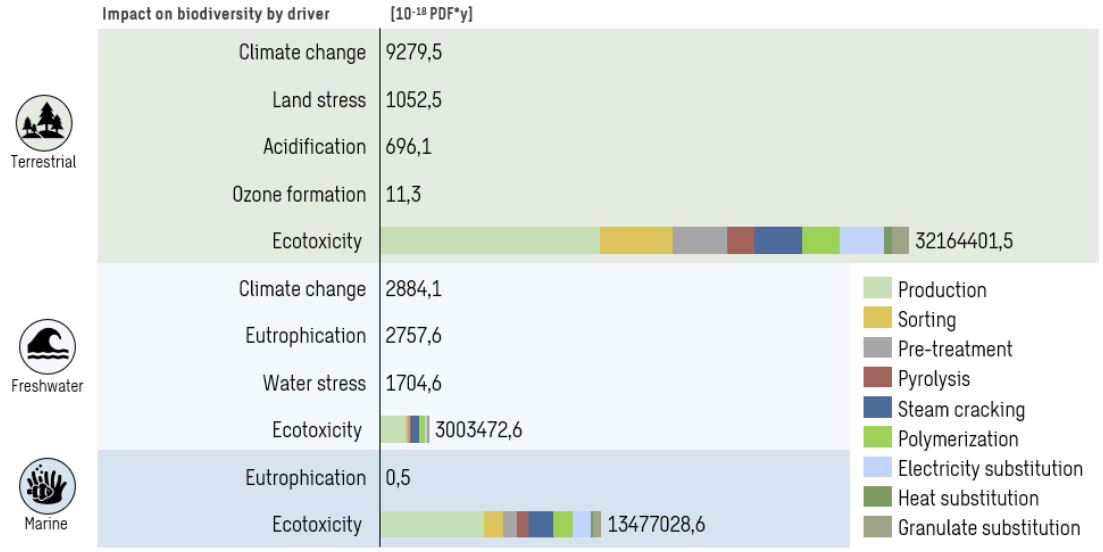

Figure 24. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 3 when ecotoxicity is considered.

5.3.3 IMPACT World+ results

In the IMPACT World+ method, the impact category results can be summed up to form the final endpoint result. The result for Scenario 1, where EoL treatment was divided between incineration with energy recovery and landfilling, is 1.70 PDF·m^2·y and represents the scenario with the highest impact on potential species loss. Scenario 2, where mechanical recycling was selected as the EoL treatment option, shows the lowest impact, with result of 1.26 PDF·m^2·y. Scenario 3, where chemical recycling was selected as the EoL treatment method, falls between the Scenarios 1 and 2, with a result of 1.56 PDF·m^2·y. Figure 25 presents the results for each scenario and contribution of processes to the final results in accordance with the reference flow and functional units.

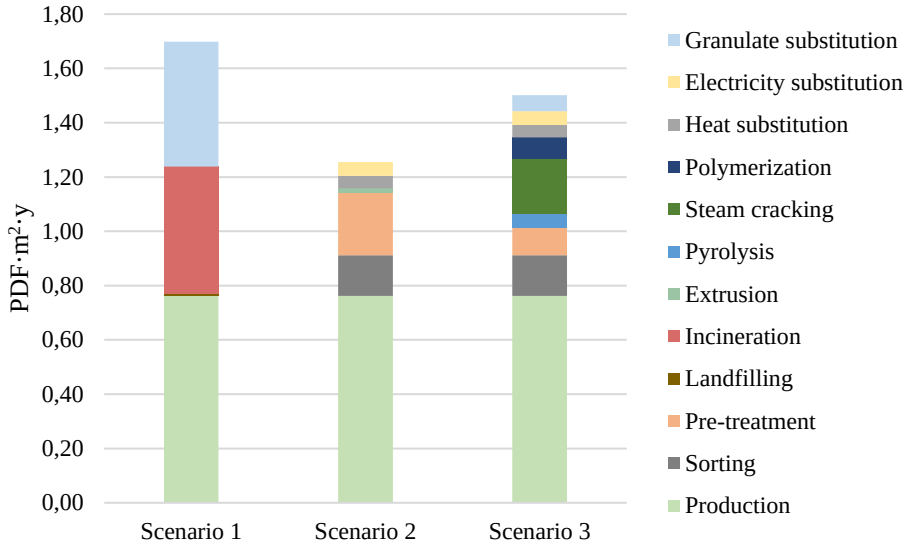

Figure 25. IMPACT World+ results for each scenario and contribution of each process to the result in accordance with reference flow and functional units.

In scenario 1, the life cycle was divided into four processes: production, where virgin HDPE granulates are produced; incineration with energy recovery; landfilling with landfill gas utilization; and granulate substitution to fulfill the functional units. The share of production from the total result is 44.9%, followed by granulate substitution at 27.1%, incineration at 27.5%, and landfilling at 0.4%. When divided into life cycle phases, raw material extraction and processing, referred to as production, account for 44.9 %, followed by EoL treatment at 28.0%, and EoL substitution at 27.1%.

In Scenario 2, the life cycle was divided into six processes: production of virgin HDPE granulates; sorting of collected HDPE waste, pre-treatment of the waste to fulfil the quality requirements for extrusion; extrusion to produce recycled granulates; and heat and electricity substitution to fulfil the functional units. The share of production from the total result is 60.8%, followed by pre-treatment at 18.4%, sorting at 11.9%, electricity substitution at 4.0%, heat substitution at 3.7%, and extrusion at 1.3%. When divided into life cycle phases, the raw material extraction and processing account for 60.8%, followed by EoL treatment at 31.5%, and EoL substitution at 7.7%.

In Scenario 3, the life cycle was divided into nine processes: production of virgin HDPE granulates; sorting of collected HDPE waste, pre-treatment of the waste to fulfil the quality requirements for pyrolysis; pyrolysis to convert plastic waste into naphtha like liquid fraction; steam cracking to produce ethylene from naphtha like fraction; polymerization to produce recycled plastic granulates; and heat, electricity, and granulate substitution to fulfil

the functional units. The share of production from the total result is 49.0%, followed by steam cracking at 13.1%, sorting at 9.6%, pre-treatment at 6.5%, electricity substitution at 5.1 %, polymerization at 5.0%, heat substitution at 4.7%, granulate substitution at 3.8%, and pyrolysis at 3.3%. When divided into life cycle phases, the raw material extraction and processing account for 49.0%, followed by EoL treatment at 37.4%, and EoL substitution at 13.6%.

Behind the results are the impact categories contributing to the ecosystem quality. Figure 26 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 1. Impact categories that were not assigned to one of the three ecosystems are placed directly under ecosystem quality. It can be noted that impacts of global warming dominate the total impact on ecosystem quality and contributes the most on potential species loss with share of 67.5% of the total result. Second largest contributors are freshwater ecotoxicity at 9.2%, terrestrial acidification at 7.3%, and land transformation 7.3%. Smallest contributors are ionizing radiation at 0.00000002% and thermally polluted water at 0.0008%. The impacts of water availability on freshwater ecosystems arising from landfilling show negative results that indicate a positive impact on biodiversity. This may be due to the allocation procedure in the wastewater treatment of landfill leachate, which incorrectly suggests an increase in water due to the release of treated wastewater. Consequently, this process lowers the overall impact. Production, incineration, and granulate substitution have largest contribution on global warming impacts on terrestrial ecosystems, while landfilling has largest impact on freshwater eutrophication.

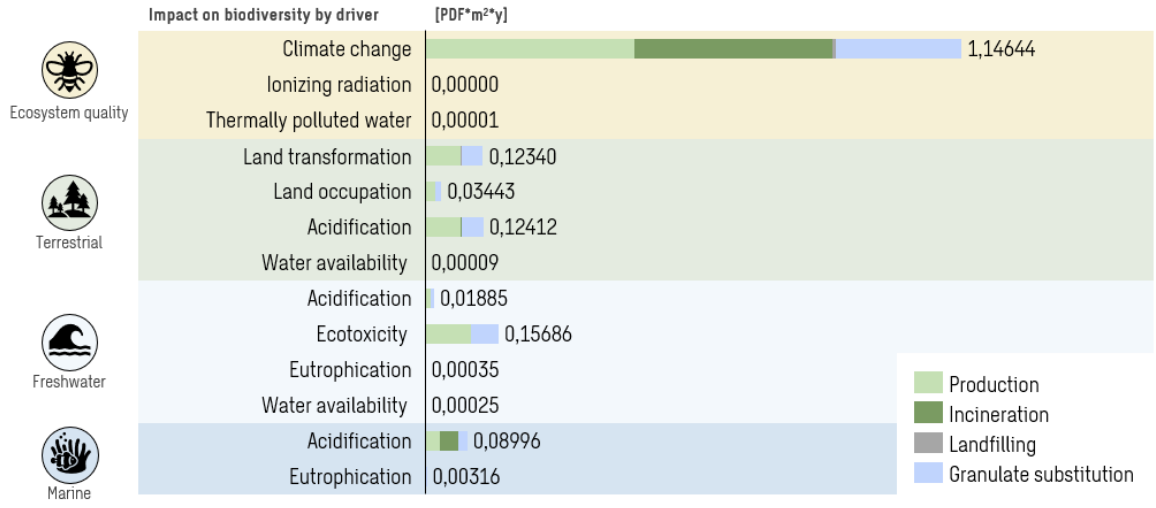

Figure 26. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 1.

Figure 27 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 2. Impact categories that were not assigned to one of the three ecosystems are placed directly under ecosystem quality. It can be noted that impacts of global warming dominate the total impact on ecosystem quality and contributes the most on potential species loss with share of 64.7% of the total result. Second largest contributors are freshwater ecotoxicity at 8.9%, terrestrial acidification at 8.5%, and land transformation 8.3%. Smallest contributors are ionizing radiation at 0.00000004 % and thermally polluted water at 0.001%. All the processes contribute the most on climate change impacts on terrestrial ecosystems.

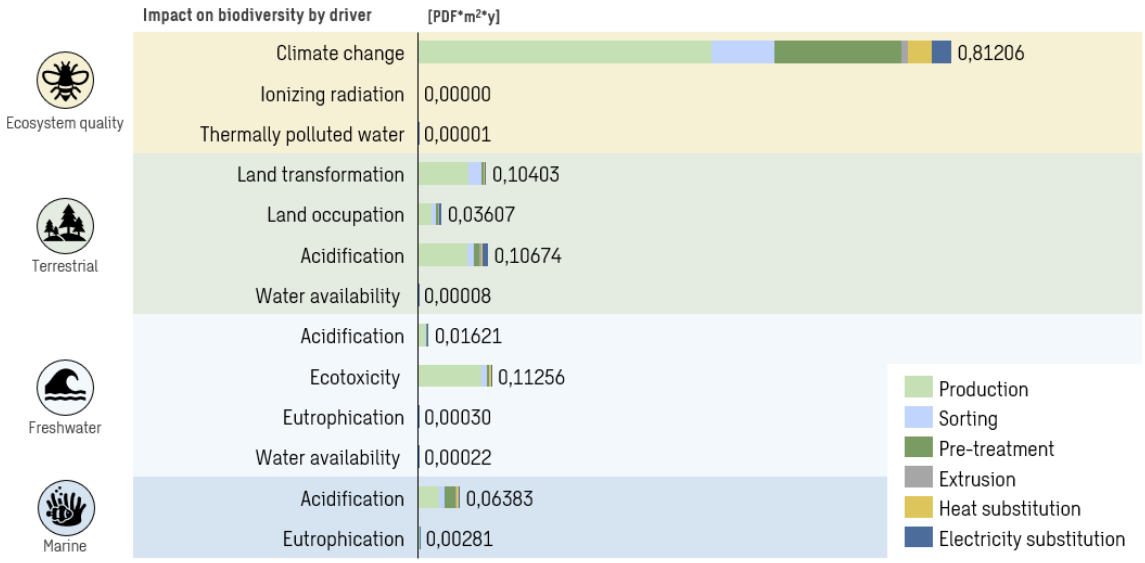

Figure 27. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 2.

Figure 28 present all impact categories/drivers of biodiversity loss contributing to ecosystem quality and contribution of processes in Scenario 3. Impact categories that were not assigned to one of the three ecosystems are placed directly under ecosystem quality. It can be noted that impacts of global warming dominate the total impact on ecosystem quality and contributes the most on potential species loss with share of 61.8% of the total result. Second largest contributors are freshwater ecotoxicity at 9.8%, terrestrial acidification at 9.8%, and land transformation 8.7%. Smallest contributors are ionizing radiation at 0.00000004 % and thermally polluted water at 0.001%. All the processes contribute the most on climate change impacts on terrestrial ecosystems.

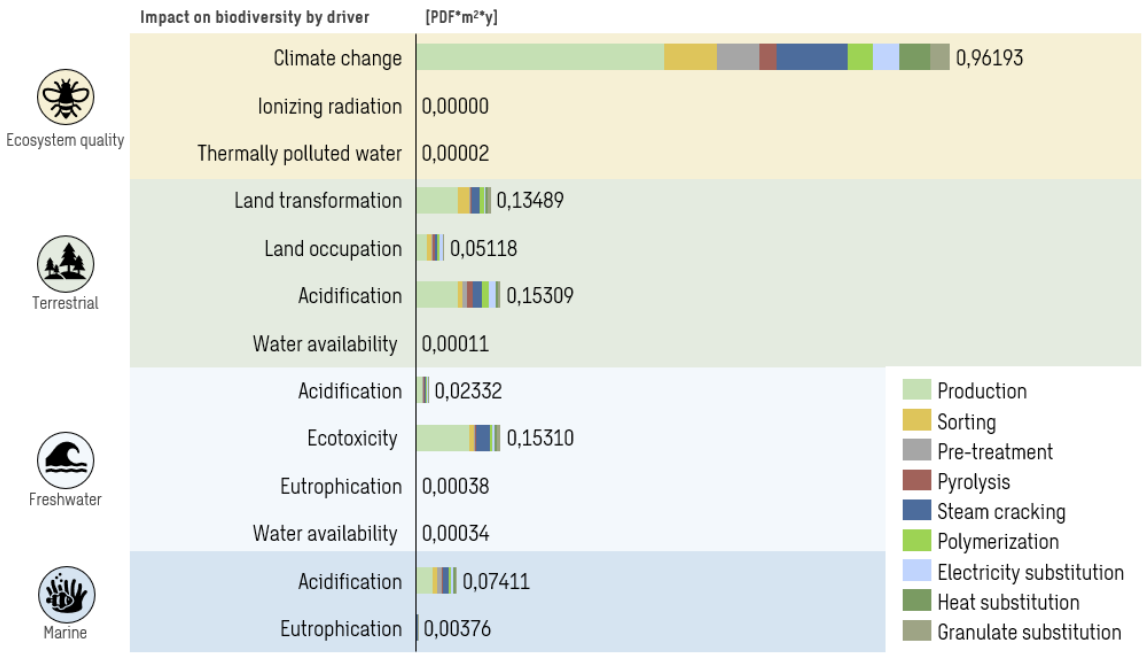

Figure 28. Contribution of each impact category to ecosystem quality and each process to impact categories in Scenario 3.

5.4 Result interpretation

The results show that all three LCIA methods placed the scenarios in the same order in terms of their total impact on ecosystem quality. Scenario 1 had the largest impact, followed by Scenario 3, while Scenario 2 had the lowest impact. Results of the study are combined in Table 17. LC-Impact results are presented without and (with) impact category of ecotoxicity.

Table 17. Impact of each scenario on endpoint indicator ecosystem quality by LCIA method. LC-Impact results are presented both without and (with) impact category of ecotoxicity.

LCIA method	Scenario		
	1	2	3
ReCiPe 2016 [species·y]	2.54E-08	1.80E-08	2.24E-08
LC-Impact [PDF_{total}·y]	1.09E-14 (3.34E-11)	7.82E-15 (2.73E-11)	9.55E-15 (2.74E-11)
IMPACT World+ [PDF·m^2·y]	1.70	1.25	1.50

The life cycles under assessment were divided into three phases: production which included raw material extraction and HDPE granulate production, EoL treatment, and EoL substitution. Granulate production had the largest impact in all scenarios, followed by EoL treatment and EoL substitution had the lowest impact in all the methods. However, differences between the methods occurred in process contribution and in the contribution of

impact categories. Figure 29 presents the contribution of each process by LCIA method for each scenario. LC-Impact results are here presented without ecotoxicity results.

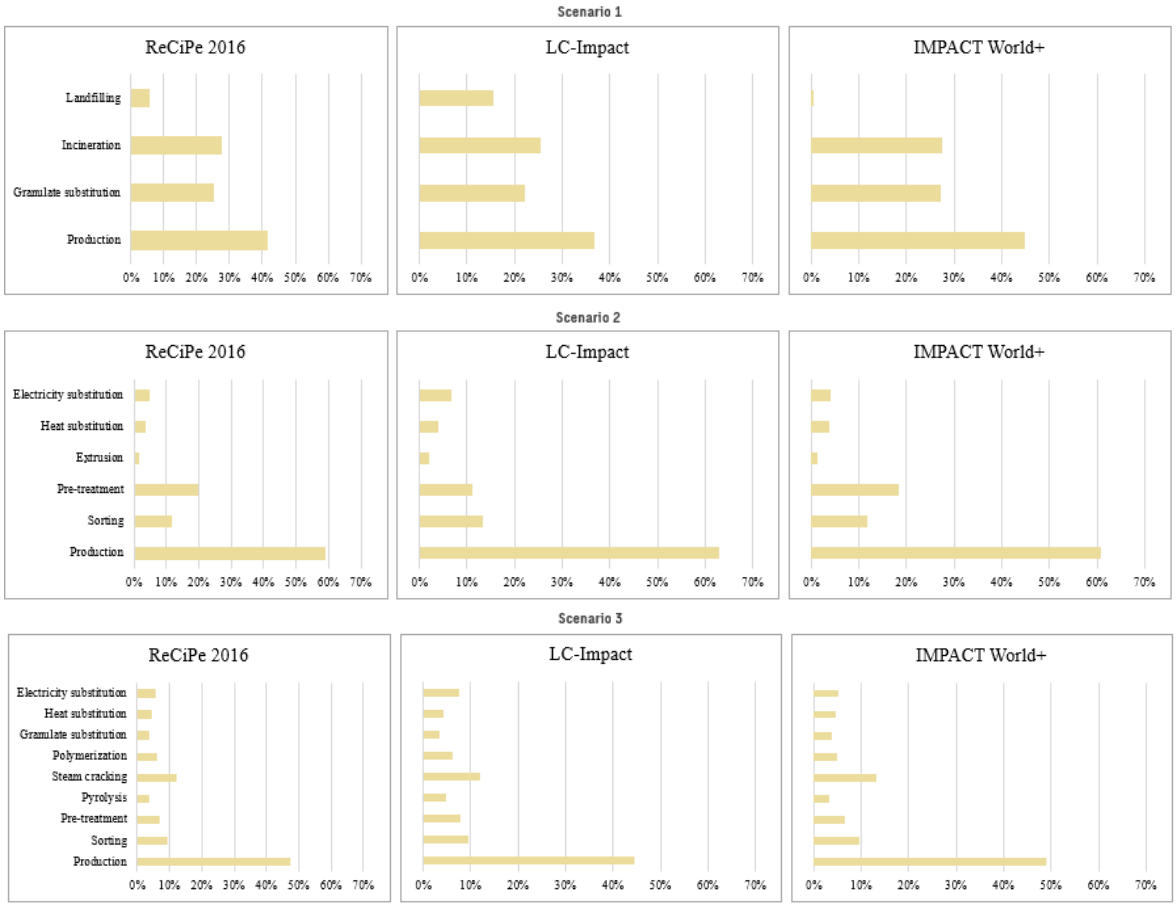

Figure 29. Contribution of processes in LCIA methods in each scenario.

The biggest difference between the methods occurs in Scenario 1 regarding the contribution of landfilling. The IW assigns a very minor contribution to landfilling, while the contribution is around 16% in LC. Impacts from landfilling contribute the most on impact category of freshwater eutrophication in RE and LC, which may explain why the IW gives such a low contribution to landfilling, as it also assigns a very low contribution to eutrophication in overall, as shown in Figure 30. Landfilling was also the only process where the methods yielded negative results, indicating a positive impact on biodiversity. The negative contribution in all methods arise from water use related impacts on freshwater ecosystems. This might be due to the allocation procedure in the water purification process of landfill leachate, which inaccurately gives impression that the amount of water would increase when it's released into a water body because the process does not consider the water intake. The water intake is also not considered in any other part of the landfilling process, because it

consists of multiple intakes during the manufacturing process and this way shows negative result for landfilling.

Production is the biggest contributor in each scenario and method. However, the fact that production is modelled as single process highlights the difference between the processes, as the EoL treatment and substitution phases are divided into smaller sub-processes. The process contribution in Scenarios 2 and 3 show quite similar results across the methods.

To compare the differences in impact category contributions, the impact categories were grouped into nine impact category groups, as performed in the study by Sanyé-Mengual et al. (2023). For example, in the case of the IW, the acidification group includes terrestrial, freshwater, and marine acidification. Figure 30 presents the contribution of each impact category in the LCIA methods for each scenario. LC-Impact results are here presented without ecotoxicity.

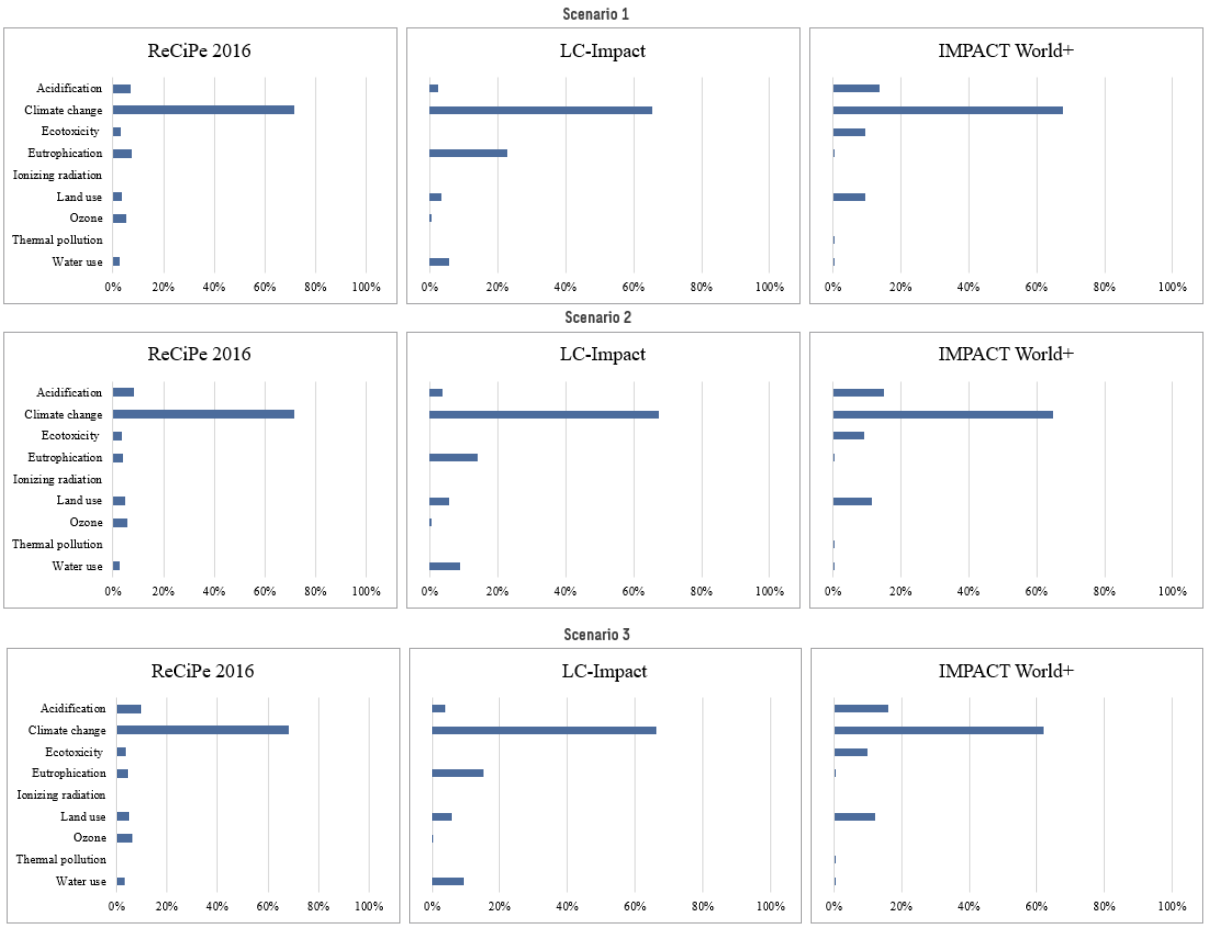

Figure 30.Contribution of impact categories in LCIA methods in each scenario.

It can be noted from Figure 30 that the share of impact categories in the total result is very similar within the method, regardless of the scenario. This observation is consistent across

all methods. The biggest difference is seen in LC method, where eutrophication has a larger share in Scenario 1 compared to Scenarios 2 and 3, which lowers the share of land use and water use.

Climate changes is the main contributor in each method, but the second-largest contributor varies. The significant contribution of climate change is likely due to the high dependency of plastics on fossil-based raw materials. Both RE and LC include impact category of climate change separately on terrestrial and freshwater ecosystems. However, it was noted from the results, that the share of climate change impacts on freshwater ecosystems were marginal in RE, while in LC, it is roughly estimated to be around one quarter of the total climate change impacts.

Acidification and land use have large contributions in each scenario after climate change. Ozone formation is assessed in both RE and LC, but RE gives a higher contribution to it. Eutrophication is evaluated in all the methods, but LC gives the highest contribution, even though all methods assess both freshwater and marine eutrophication. Water use is assessed in all the methods, with LC giving the highest contribution and IW the lowest, even though LC only assesses impact on freshwater ecosystems, while IW and RE also consider impacts on terrestrial ecosystems. Acidification is assessed in all the methods, but LC and RE evaluated only terrestrial acidification, while IW assesses impacts of acidification on all three ecosystem types. IW gives the highest contribution to acidification, followed by RE and LC. Ecotoxicity is shown here for RE and IW, but in IW it has higher contribution. Ionizing radiation and thermally polluted water are assessed only in IW, but they both have very minor contribution.

All the assessed methods included regionalized impact categories, meaning that origin of the emission or resource use has an impact on the biodiversity impact result. Due to the nature of the data used in this study, there may be a shortage of location data, creating uncertainty to the results. The used Ecoinvent datasets often combine available data from countries within Europe, forming an average. The datasets may also include global data inside the used processes. Some of the processes were modelled using country specific data if European average were not available. Data within European countries may vary and this way the result may not present the average process in Europe. For example, differences between landfilling and incineration data vary widely across European countries. Currently, there is no

possibility to examine the origin that SimaPro uses for the released emissions and resource extraction for spatially resolved impact categories.

Based on the results, mechanical recycling appears to be the best EoL treatment option in terms of biodiversity. When interpreting the results, all limitations need to be considered. When the recycling processes are compared, due to data limitations, the modeling of extrusion is the simplest and consist only of electricity use, which might not reflect the real situation. Steam cracking and polymerization are modeled based on conventional processes and may include emissions that would not be released if only pyrolysis oil is used as feedstock, or some emissions might not be considered.

Due lack of data, in both recycling scenarios it was assumed that used amount of washing water in pre-treatment is entirely returned as wastewater which is not accurate in real case due evaporation. This assumption may increase the impacts occurring from water purification process. Also, no specific data were found about the composition of released wastewater, so it was modelled as average wastewater treatment in accordance with the WEEE study.

Transportations that were not included in Ecoinvent processes are quite rough estimations and based on only data from one European country. These transportations include distance between waste collection and EoL treatment option in all the scenarios and transportations of material residues to incineration plant in Scenario 2 and 3. So, each scenario has some uncertainty related to transportation and the impact may be lower or higher in real case.

In the case of mechanical recycling, it is assumed that the recycling process produces virgin-like material, which might not be accurate. Mechanically recycled material can face material degradation even after first recycling, which affects its further applications and recyclability. The made assumption may have given mechanical recycling an inaccurate advantage. In Scenario 2, it was also assumed that all material losses occur in the pre-treatment phase, leaving extrusion burden free from impacts of material residue treatment. This assumption may increase the impact of pre-treatment and lower the impact of thermal extrusion.

Chemical recycling is still a new and constantly evolving field, with different technologies being developed. In the case of pyrolysis, the feedstock and process conditions have a notable impact on factors such as energy consumption and the yield of gas, oil, and char, and multiple setups can be used. For this reason, the modeled chemical recycling represents only

one method based on a literature review and secondary data, and therefore can only be considered an estimate. In the modeling of pre-treatment, no chemical recycling specific data were found, so the same data as in mechanical recycling were used, with the exception that a larger share of treated waste could be used as feedstock for the chemical recycling process. The impact on pre-treatment could be smaller or larger in real cases. In pyrolysis process pyrolysis gas and char were combusted to produce heat to maintain pyrolysis process, based on literature review it was noted that excess heat is often produced and can be used, for example, in district heating. This assumption may have affected the need for heat substitution in Scenario 3. However, due lack of data the emissions from pyrolysis gas and char combustion were limited out of this study.

To address the allocation problem, a system expansion approach was selected, considering only the first life cycle of the plastic. As noted in the Chapter 4.3 allocation procedures might have a notable impact on the results. In this case, the selected system expansion method increases the impact due to substitution but enables comparability between the scenarios. If multiple life cycles had been considered, the scenarios might have yielded different results based on the selected allocation procedure and further recycling potential.

Different LCIA methods all showed a high contribution of climate change. To mitigate the biodiversity impacts, emissions contributing to climate change should be the first target for potential reduction in each scenario. However, following climate change, there was variation between the methods regarding which impact category had the second largest contribution. Based on this study, the selection of impact assessment method to quantify biodiversity impacts may influence the planned actions based on the result. Additionally, variations in process contributions occurred, which may affect the process selected for further investigation into potential emission reduction. To be able to investigate deeper the emissions behind the results, emission inventory and their impact on overall results should be considered.

LC-Impact differed from other methods in that the results should primarily be interpreted per ecosystem type. To better compare results between the methods, the three ecosystem types were merged into a single result. The weighting method used is not widely accepted and includes uncertainty. It assigns the highest weight on terrestrial ecosystems, followed by freshwater ecosystems, with the smallest weight assigned to marine ecosystems. This was particularly noted in the results with ecotoxicity, where the high impact on marine

ecosystems compared to other ecosystems resulted in a relatively low total result. Therefore, after weighting, the scenarios maintained the same order based on the overall result, both with and without ecotoxicity. However, the ecotoxicity results themselves included uncertainty due to the calculation method and notably high contribution compared to the other methods. When ecotoxicity was considered in the assessment, its share of the total result for each ecosystem type was approximately 99%, and this was consistent across all scenarios. Some differences in process contribution were observed between the results without and with ecotoxicity. Figure 31 presents the process contribution in each scenario, both without and with ecotoxicity.

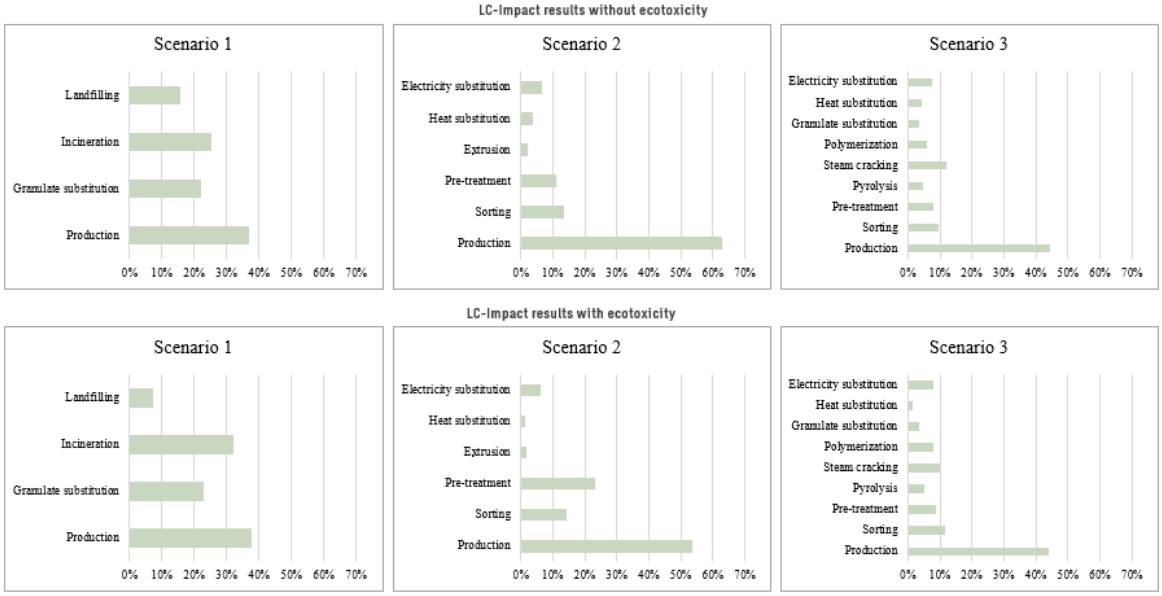

Figure 31. Process contribution in LC-Impact without and with impact category of ecotoxicity.

Figure 31 shows that in Scenario 1, the inclusion of ecotoxicity lowers the share of landfilling and increases the share of incineration. In Scenario 2, it increases the share of pre-treatment while lowering share of production, heat substitution, and extrusion. In Scenario 3, the differences are minor. However, steam cracking has larger share, while heat substitution shows a lower share.

6 Discussion

In the theory section, it was noted that the EU has several targets related to tackling biodiversity and plastic-related problems. These include, for example, higher recycling rates for plastic waste and setting the EU on a path to recover biodiversity. Based on this study, both mechanical and chemical recycling could be solutions to help achieve these targets. Both scenarios increase the recycling rate of plastic waste and have a lower impact on biodiversity in terms of potentially disappeared fraction of species than conventional plastic production, at least when HDPE is considered. Among the assessed recycling methods, mechanical recycling showed a lower impact on biodiversity. However, the assumptions made, and the limitations acknowledged may have affected the results, and more research is required to further investigate the impacts of these processes.

To improve the study, it could be beneficial to investigate the emissions and environmental mechanisms behind the impacts in more depth. This inclusion could enable more detailed suggestions to mitigate biodiversity loss. Differences between the methods could also be considered at this level. However, these were outside the scope of the study but could be a way to expand the research in future. Another suggested improvement is the inclusion of long-term emissions into the study. All methods under assessment provided, at least for some impact categories, the opportunity to select emissions over a longer timeline. Each method also suggested calculating results using more than one set of CFs. Long-term emissions do introduce higher uncertainty into the results, but emissions and resource use can have very long or infinite effects on the natural environment. Thus, the inclusion of long-term emissions could reveal important impacts that were not captured in the assessment using shorter-term emissions. For example, in LC-Impact, when considering all impacts over 100 years, 27 metal ions are accounted for in the ecotoxicity impact category, whereas in addition to the metal ions, long-term assessment also includes 2499 organic substances.

The current LCIA methos are not yet able to consider all drivers of biodiversity loss. The assessed methods exclude at least the impacts caused by invasive species and overexploitation, which are among the key drivers of biodiversity loss. However, it is unclear how these would affect the results in the context of plastic manufacturing and EoL treatment,

at least the overexploitation could be considered a thread to biodiversity arising from the life cycle of plastics.

One type of pollution causing biodiversity loss is plastic pollution. Incorporating plastic pollution into LCIA methods would enhance their effectiveness in evaluating biodiversity impacts. Both LC-Impact and ReCiPe 2016 recognize plastic pollution as a potential impact category. A study by Nessi et al. (2020) evaluated micro- and nanoplastic generation, considering total plastic loss, release into oceans, and release into terrestrial environments over the product life cycle. For conventional HDPE beverage bottles, depending on the selected method, between 0.01 and 0.02 kg of HDPE/kg produced was released into the ocean, while 0.005 to 0.009 kg of HDPE/kg produced was released into terrestrial environments. The findings were consistent for both conventional and mechanically recycled plastics. These results underscore the need to include plastic pollution in LCIA methods to effectively quantify its overall biodiversity impact.

The increase in recycling methods can reduce biodiversity impacts and enhance recycling rates. However, to further diminish the biodiversity impacts, eco-design may offer a viable solution. This approach could ensure safer recycling processes and improve reuse potential. This study noted that sorting and pre-treatment have a notable impact on the overall results of the recycling methods. To minimize the impact of these processes, material residues should be reduced, and eco-design could also serve as a tool for this optimization.

Plastics will remain an important material in the future, and recycling methods need further developments to meet the biodiversity and plastic recycling targets. As mentioned in the theory section, chemical recycling is currently considered the future of plastic recycling, and companies in the industry are committed to developing the technology. In addition to recycling methods, biobased materials, and the utilization of carbon capture in plastic manufacturing are also mentioned in Plastics Europe’s roadmap to circular and net-zero emissions plastics in Europe. These methods could also be potential scenarios to investigate concerning biodiversity impacts and compare with recycling methods and conventional plastics manufacturing.

7 Conclusions

The results indicated that mechanical recycling had the lowest overall impact on biodiversity, followed by chemical recycling, while conventional plastic production had the highest impact across the LCIA methods. Although the methods were not directly comparable, the contribution of processes to the overall results, as well as the contribution and coverage of impact categories, were analyzed.

When examining the contribution of life cycle phases, granulate production consistently had the highest contribution across all methods and scenarios, followed by EoL treatment, with substitution having the lowest impact. In terms of process contribution, production had the highest impact across the methods and scenarios, followed by incineration in Scenario 1, pre-treatment or sorting in Scenario 2, and steam cracking in scenario 3. In analyzing the contribution of impact categories, it was noted that the contributions were quite similar across the scenarios within each method, indicating that the methods may often assign high weighting to certain impact categories.

The ReCiPe 2016 method provided a single score for each scenario, resulting in $2.54 \cdot 10^{-8}$ species·y for Scenario 1, $1.80 \cdot 10^{-8}$ species·y for Scenario 2, and $2.24 \cdot 10^{-8}$ species·y for Scenario 3. The main drivers in Scenario 1 were impacts from global warming on terrestrial ecosystems, freshwater eutrophication, and terrestrial acidification. In Scenario 2 and 3, the primary impacts arose from global warming impacts on terrestrial ecosystems, terrestrial acidification, and ozone formation.

The LC-Impact method provided three results for each scenario, one for each ecosystems type, which should primarily be interpreted separately. However, to enable better comparison between the methods, ecosystem specific results were merged into a single result. The merged results were $1.09 \cdot 10^{-14}$ PDF·y for Scenario 1, $7.82 \cdot 10^{-15}$ PDF·y for Scenario 2, and $9.55 \cdot 10^{-14}$ PDF·y for Scenario 3. The main drivers in Scenario 1 were impacts from climate change on terrestrial ecosystems, freshwater eutrophication, and impacts of climate change on freshwater ecosystems. In Scenario 2 and 3, the main impacts arose from climate change impacts on terrestrial ecosystems, climate change impacts on freshwater ecosystems and freshwater eutrophication. The ecotoxicity impact category was examined separately, as it required conversion of CFs. It was noted that ecotoxicity has a

significantly high contribution, accounting for approximately 99% of the total result compared to other impact categories.

In the IMPACT World+ method, a single result for each scenario was provided, yielding 1.70 PDF·m^2·y for Scenario 1, 1.25 PDF·m^2·y for Scenario 2, and 1.50 PDF·m^2·y for Scenario 3. The main drivers behind the result were climate change, freshwater ecotoxicity, and terrestrial acidification in all scenarios. Notably, compared to other methods, landfilling had a low contribution, likely due to the method's minor weighting on eutrophication in overall.

The assessment demonstrated that the selected impact assessment methods can yield varying contributions for the assessed processes and highlight different impact categories. All methods assigned high contributions to impacts from climate change on terrestrial ecosystems, but the second highest contributor varied. Therefore, to reduce the negative impact on biodiversity, actions may be targeted to different impact categories or processes depending on the method used.

To further develop the study, it is suggested to include an assessment of long-term emissions and a deeper exploration of the emission inventory and weighting of emissions in the methods. The study also acknowledges uncertainties, and data gaps that may have affected the results. In conclusion, the results underscore the need for innovative recycling strategies to address the pressing challenges of biodiversity loss linked to plastic manufacturing and EoL treatment.

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