Industrial Ecology - master thesis
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Assessing the environmental performance of hydrogen-oxidising bacteria as feed alternative at pilot and commercial scale, using a combination of intermittent and baseload renewable energy sources

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Abstract

Aquacultures are rapidly increasing worldwide to meet the rising protein demand caused by population growth. However, commonly used fish feed products are known to harm the environment while depleting natural species. Various microbial protein forms are thus being investigated, such as those using hydrogen-oxidising bacteria, referred to as MP. This study investigates MP's environmental performance over conventional feed products as overall advantages and disadvantages remained unclear based on previous publications.

A four-phase explorative scenario framework for assessing novel technologies was used, further developed from recent studies. Based on a literature review of MP's environmental performance, the ammonia and heat source used needed further evaluation. At the same time, previous studies displayed limitations related to CO2 allocation, continuous energy and nutrient supply, energy grid stability, the readiness of the assessed technologies, the cost of production, and other influencing socio-economic factors. An attributional cradle-to-gate life cycle assessment (LCA) was conducted to determine an optimal system performance at a pilot-scale rooted in the literature findings. Subsequently, technical and socio-economic parameters were evaluated to influence MP's progressive environmental performance.

Consequently, an ex-ante LCA compared MP production scenarios over time, differing in energy sources and production scales, against soybean meal (SBM). Even at high technical development, comparative results closely aligned, yet MP using geothermal energy showed a slight overall advantage over SBM after normalisation, in contrast to hydro and bioenergy displaying no real benefit. When oxygen (O2) was recycled, the former showed an overall advantage over the incumbent at high production capacity. A total disadvantage would be expected for the latter, considering continuous H2 and NH3 supply. Further research should focus on integrating recycled phosphate, green H2 in the steel industry, differences in databases regarding water use for hydropower plants, and scenario analysis of the incumbent system. Besides these findings, the developed framework contributes toward the comparative assessment of novel technologies against established ones.

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AB: activity browser
AEL: alkaline electrolyte
BbP: biobased product
BCHP: biomass combined heat and power
BECCS: bioenergy carbon capture and storage
BG: biomass gasification
BR: Brazil
CA: contribution analysis
CCA: cross-consistency analysis
CCS: carbon capture and storage
CCU: carbon capture and utilisation
CF: characterisation factor
CH4: methane
CO2: carbon dioxide
DAC: direct air capture
DM: dry mass
EF: environmental footprint
E-TAC: electrochemical thermally activated chemical
GHG: greenhouse gas
GMA: general morphological analysis
H2: hydrogen
HOB: hydrogen-oxidising bacteria
IC: impact category
ILCD: International Reference Life Cycle Data System
KOH: potassium hydroxide solution
LCA: life cycle assessment
LCOH: levelized cost of hydrogen
LP: low pressure
LU: land use
LUC: land use change
MP: microbial protein using hydrogen-oxidising bacteria
MPL: market penetration level
MRL: manufacturing readiness level
N2: nitrogen
NH3: ammonia
O2: oxygen
PCC: post-combustion capture
PEF: product environmental footprint
PEM: polymer electrolyte membrane
PtF: power-to-food
PtX: power-to-X
RoW: rest of the world
SA: sensitivity analysis
SBM: soybean meal
SMR: steam methane reforming
TCH: total cost of hydrogen
TRL: technology readiness level

1 Introduction

Current food systems are a leading cause of environmental dilapidation, contributing to biodiversity loss, water scarcity, climate change and land degradation (Campbell et al., 2017). The current protein need for human consumption is unprecedented (Trostle, 2008). It is assumed to increase further over the coming decades amid the rising standard of living and world population, which is estimated to grow to 10 billion by 2050 (Ezeh et al., 2012). Ensuring food security under current production methods often poses a challenge, especially for developing countries, causing social and economic issues because of malnutrition (Gabriel et al., 2014).

Many countries have supported aquacultural practices to meet this growing demand for protein, which has grown at an average of 7.5% every year since 1990, much quicker than any other livestock sector (Troell et al., 2014). Fish have a much low feed conversion ratio of about 1.1-1.6kg, which is considerably less than other species, with that of ruminants being 3-6 times higher (Jones et al., 2020). Yet, it requires almost 1.5 times the amount of sea fish to produce 1kg of farmed fish (Jones et al., 2020), which has contributed to overfishing of the oceans, reducing the population of species, such as tuna and mackerel, to as little as 25% compared to their initial existence (Cohen et al., 2018). Despite a constant annual amount of wild catch since the early nineties of roughly 90 million tons (Jones et al., 2020), an estimated 60% of wild fishery stocks are already fully depleted, 30% are overfished, and less than 10% still have lasting capacity (Little et al., 2016).

Due to its high crude protein content of up to 60% and its provision of all essential amino acids, fishmeal and fish oil have been the dominant ingredients in aquaculture feeds (Jones et al., 2020). Although some animal by-products have similar nutritional levels as fishmeal, they can hardly compete due to the demand for high-quality protein, omega-three fatty acids, and the lipid requirements of carnivore fish species, such as salmon (Hasan, 2001). Even though plant-based feeds can be compatible with some fish species like salmonids, this also increases the price (Jones et al., 2020). Aquacultures can thus additionally deplete fishery stocks or have the opposite effect, depending on the source of fish feed. In addition, using fishmeal as a feed source leads to antibiotic resistance of bacteria in marine sediments (Han et al., 2017). Even with a drastic reduction in the proportion of fishmeal in fish feed, a substantial shortage of fishmeal of over 1 million metric tons can arise by 2050 (Jones et al., 2020).

Aside from the depletion of the oceans, feed production on land has caused numerous environmental problems (Bodirsky et al., 2015). Due to their outstanding share of protein and production efficiency, soybeans are the most relevant contributor to animal-free protein for both animals and humans (Sillman et al., 2019). Soybean meal (SBM) comprises two-thirds of all protein-concentrated feed in the EU (Spiller et al., 2020). The crop is expected to have a yearly growth of 1.6% until 2027, primarily due to China´s high need for animal feed (Spiller et al., 2020). This increase in production causes land use change (LUC); (Spiller et al., 2020), severely impacting climate change and damaging biodiversity (Castanheira & Freire, 2013; (Chaudhary & Kastner, 2016).

Besides these impacts, nutrient runoff due to the low nutrient absorbability of plants grown in open fields causes environmental pollution (Lassaletta et al., 2014). Therefore, low-priced protein sources with high nutrient conversion efficiencies are needed (Crab et al., 2007), which display lower environmental burdens than conventional protein crops (Spiller et al., 2020). As the European Union is almost exclusively dependent on foreign supply for SBM, the urgent need to replace imports with alternative protein sources has now been deemed a necessity by the members (Spiller et al., 2020). Innovative food technologies show promising prospects to guide this transition (Parodi et al., 2018), such as the upcoming field of cellular agriculture, which is generally perceived as a sustainable alternative to conventional protein sources (Tuomisto, 2019; (Rischer et al., 2020).

Microbial protein, also named single-cell protein, a subset of cellular agriculture, has increasingly been getting attention as a novel alternative to conventional protein sources (Pikaar et al., 2018). The production process achieves much higher yields than those achieved through traditional agricultural methods (Pikaar et al., 2018) due to the rapid growth of microbial protein (Ritala et al., 2017). The protein content of the retrieved biomass is usually between 50–83%, which generally is not entirely usable (Sillman et al., 2020). Since 1960, efforts to produce microbial protein using methanol, methane (CH4) or waste hydrocarbon streams have been pursued, aiming for more sustainable food production (Goldberg, 1985). However, because of the low costs of fishmeal and the enormous production costs of microbial protein in the past, it was only recently that developments in the field had gained momentum due to a five-fold increase in fishmeal prices since 1990 (Kupferschmidt, 2015). Single-cell proteins can be consumed by humans directly or produced for animal feed (Ritala et al., 2017).

1.1 Knowledge gap, research aim, and research questions

Because many single-cell protein production methods are still being developed, little is known about their environmental impacts from a system's perspective. Life cycle assessment (LCA) is the most comprehensive tool to evaluate a product or service system's environmental performance (Wolf et al., 2012). Only a few studies have examined the ecological impacts of single-cell proteins using the LCA methodology (Cumberlege et al., 2016); (Pikaar et al., 2018); (Sillman et al., 2020); (Spiller et al., 2020); (Järviö et al., 2021); these studies are based on evaluating single-cell proteins derived from bacterial sources. As the energy and material use of single-cell proteins differ substantially (Spiller et al., 2020), it is essential to assess the environmental impacts of each production technique individually, including scenarios of various system modifications, such as the use of different substrates or microorganisms (Sillman et al., 2020).

Several microbial protein forms from various microbial sources, such as bacteria, yeast, fungi, and microalgae, are being investigated for commercialisation, displaying different strengths and weaknesses for multiple products (Jones et al., 2020). Microbial protein can rapidly grow inside bioreactors through fermentation on organic substrates such as sugar from sugarcane (Pikaar et al., 2018). Autotrophic microbes are single-cell organisms that use organic or inorganic carbon sources as feedstocks utilising CH4 or carbon dioxide (CO2) streams, making organic substrates redundant (Ritala et al., 2017). This process, thus, decouples protein production from LU (LU) and agricultural pollution entirely (Pikaar et al., 2018). In addition to not being influenced by changes in climate or soil conditions, the use of herbicides and pesticides becomes redundant, which could otherwise impact biodiversity loss (Sillman et al., 2019).

Besides potentially providing an ecologically sound alternative to conventional feed for numerous animals, some microbial proteins display additional benefits (Jones et al., 2020). Microbial protein derived from yeast has been successfully proven to supplement 40% of fishmeal in salmon feed without displaying adverse side effects for the fish or yield capacity (Jones et al., 2020). *C.utilis*, a yeast-derived microbial protein, successfully supplemented 50% of fishmeal in shrimp feed while increasing their growth rate (Jones et al., 2020). The authors also note that several bacteria-derived microbial proteins have displayed even better results, with the product KnipBio Meal replacing fishmeal in shrimp production and up to 55% in salmon feed (Jones et al., 2020).

Microbial protein production through methanotrophic bacteria that utilise CH4 is already used to produce feed commercially. Yet, as natural gas is currently used as a cheap CH4 source, this method's long-term feasibility is questionable (Cumberlege et al., 2016); (Pikaar et al., 2018). For an optimal scenario, the CO2 footprint, water consumption level, and LU indicators of FeedKind, a methanotrophic bacteria, were 2.23kg CO2-eq, 10kg, and 0m2, respectively, per kg of protein produced (Cumberlege et al., 2016). These values are based on a scenario analysis assuming that the product was left in its powdered form, biogas was used instead of natural gas as a substrate, and fossil fuels were replaced with renewable energy sources throughout the supply chain (Cumberlege et al., 2016).

Due to the limited availability of biogas, it is debatable if it should be used to produce food instead of as an energy source. Even if all resources from available feedstocks, either sustainably-grown crops or recovered wastes, were utilised for biogas production, this would only cover 6-9% of global primary energy demand or 26-37% of natural gas demand (World Biogas Association, 2019). Furthermore, pelletising is a very energy-intensive process necessary for fish feed. Accounting for it is, therefore, essential for a comprehensive analysis. The default scenario of FeedKind used natural gas instead of biogas and electricity from the US grid instead of renewable sources while including pelletising in the production. The default scenario results were thus much worse than the optimal scenario with a footprint of 5.82kg CO2-eq, 18.98kg water use, and 0.034m2 LU per kg produced (Cumberlege et al., 2016). The impacts of 1kg of soybean produced, in comparison, are 0.53-3.74kg CO2-eq for climate change (Sillman et al., 2019); (Smetana et al., 2019), 5.24-6.04m2 for LU (Sillman et al., 2019), and 92-466kg for water use (Cumberlege et al., 2016). Thus, the water and land requirements are much less for FeedKind, yet the CO2 footprint is not competitive with SBM.

KnipBio Meal, another methanotroph, is a non-pathogenic plant epiphyte called *Methylobacterium extorquens* (Ochsner et al., 2015). It is known for utilising methanol, yet, it can also feed off other substances such as ethanol (Ochsner et al., 2015). There are two types of ethanol: petroleum-derived ethanol and bioethanol made from biomass fermentation (Tamers, 2006). KnipBio Meal uses condensed distillers soluble, a by-product of bioethanol production (SalmonBusiness, 2019).

Despite efforts to increase bioethanol production, the global share of ethanol is still dominated by fossil-based products (Rass-Hansen et al., 2007). The latter is a biofuel, mainly of first-generation origin, thus directly competing with arable land for food production (Phillips, 2022). Second- and third-generation biofuels were introduced to mitigate such competition (Phillips, 2022). Still, the OECD-FAO have projected that the input for bioethanol will remain in food crops (Hammond & Seth, 2013). In contrast, second-generation bioethanol is only expected to represent 7% of overall manufacturing, mainly produced in developing countries (Hammond & Seth, 2013).

Alternatives to methanotrophs are microbial proteins produced through hydrogen-oxidising bacteria (HOB). Suppose hydrogen (H2) is made from renewable energy sources through water electrolysis; this could decouple microbial protein production from arable land and fossil fuel inputs (Pikaar et al., 2018). The required CO2 is abundant, making it a cheap resource (Silmann et al., 2019). Microbial/single-cell protein using HOB will now be referred to as MP. The case of *Cupriavidus necator* (formerly *Ralstonia eutropha*) (Yu, 2014); (Liu et al., 2016), an example of a MP, can substitute conventional high protein feed products such as fishmeal and SBM, as well as animal-based proteins for human consumption. Compared to such feed and food products, *C. necator* matches the protein level, essential amino acids and nutritional value (Volova & Barashkov, 2010) that animals cannot create themselves (Sillman et al., 2020). Furthermore, MP has a high theoretical solar-to-biomass conversion rate of around 10% (Liu et al., 2016). Comparing this to a conversion efficiency of 0.55% for soybean seeds (Hu et al., 2020), MP shows a drastic productivity increase. In certain parts of the planet where food supply is insecure due to weather conditions, such production methods can play a crucial role in stabilising food security while utilising non-arable land (Sillman et al., 2019). However, the ways to produce MP for feed and food are still under development (Järviö et al., 2021).

There are only two LCA studies on single-cell proteins using HOB; (Sillman et al., 2020); (Järviö et al., 2021), of which only the latter is based on empirical data. The environmental benefits of MP using renewable energy sources over other animal-based proteins have been well documented (Järviö et al., 2021). Regardless, in the case of plant-based feed alternatives, the potential advantages and disadvantages over MP were less evident in the study (Järviö et al., 2021) and depended on various parameters. Additionally, the statements made in the publication (Järviö et al., 2021) regarding the comparison between feed alternatives and MP did not always match the information found in the supplementary information. More research was therefore required to gain tangible results for such a comparison. In addition, both studies (Sillman et al., 2020); (Järviö et al., 2021) displayed limitations and recommendations for further research. Based on these limitations and recommendations which are further discussed in chapter 3, the following research questions were defined:

> *What are the environmental impacts of producing 1kg of MP, progressing from pilot-scale to high commercial-scale while considering a combination of intermittent and baseload energy sources, and how do these results compare to SBM production?*

To comprehensively answer the main research question, several sub-research questions were formulated:

- *How to best allocate biogenic carbon from point sources for MP production?*
- *From an economic and environmental perspective, what is MP production's most efficient technological setup?*
- *What are not previously identified hotspots for improving this technology?*
- *What are the current methods for assessing the ecological footprint of emerging technologies, and how can they be improved?*

These questions will be answered in this study's conclusion. The target audience of this study is technology developers, experts, and policymakers. The research will be reviewed by supervisors from the University of Leiden and the TU Delft. In theory, the supervisors could be viewed as having the role of a steering committee. There is no commissioner to this study. It was a topic freely chosen as a master's thesis as part of the course Industrial Ecology hosted by the two Universities. The study is therefore free of diverging interests.

2 Methodology

2.1 Background of the methodology

2.1.1 The LCA methodology

Many research and policy institutions have progressively advocated avoiding accidental environmental externalities (Cooper & Gutowski, 2020). As previously mentioned, LCA is the most effective method to determine a product or service system's environmental footprint (EF) through physical measurements of material and energy flows (Wolf et al., 2012). A system boundary includes all upstream and downstream activities related to the product, aiming to include all environmental impacts associated with the system while preventing problem-shifting from one impact category (IC) to another (Guinée et al., 2002). Inside the system boundary, there are foreground and background systems. These systems are comprised of unit processes that link the so-called economic flows, namely material, waste, and energy flows, to all stages of the product's or service's system, from mining, production, and transportation to use and disposal (de Haes & Heijungs, 2009). The system boundary also defines the relationship between the economic and ecological subsystems. For example, in the case of agriculture, the proceeds belong to the economic system, whereas the soil belongs to the environmental system.

The LCA methodology is based on four steps: goal and scope definition, life cycle inventory (LCI), life cycle impact assessment (LCIA), and interpretation (Guinée et al., 2002). This framework to determine environmental impacts in an iterative progression between the four stages (Guinée et al., 2002) is demonstrated in figure 1.

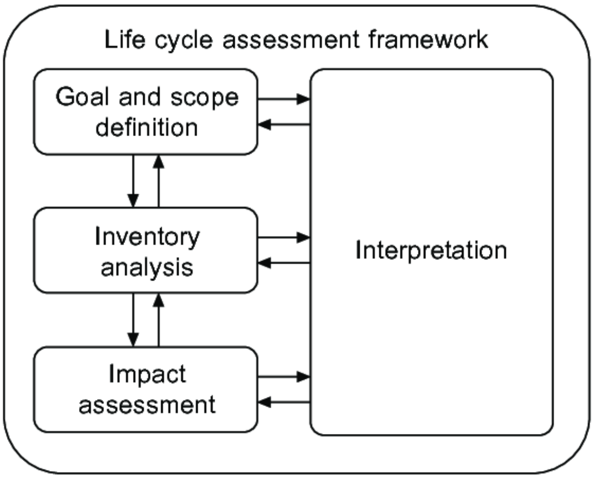


Figure 1. LCA framework diagram (Guinée et al., 2002)

The ISO 14040 and14044 standards give guidance to the LCA framework and the four distinct phases (Guinée et al., 2002):

- Goal and scope: in this phase, the plan for the LCA is laid out. The system boundary, function, functional unit (FU), and reference flow of the product system(s) under analysis are defined.
- LCI analysis: in this phase, the product system gets defined through the system boundary, using a flow chart diagram including all upstream and downstream economic and environmental flows. Data for the unit processes are gathered and quantified. Multifunctional processes are allocated, and cut-offs to and from the system boundary are defined. The primary outcome of this phase is an inventory table.
- LCIA: the inventory table is further evaluated in this step, assigning each environmental flow towards one or more ICs through CFs. These characterised results can be normalised by comparing the characterisation results to a global or regional reference value. The results can be additionally simplified by weighing, indicating a further preference for one system over another.
- Interpretation: in this phase, the robustness of the model is evaluated while conclusions and recommendations are made.

The four guiding steps of the LCA framework have historically been used in an ex-post way to assess the well-established, commercially available product and service systems with easy access to factual data (van der Giesen et al., 2020). However, the conventional LCA framework falls short of giving guidance for evaluating emerging technologies related to data availability issues, incorporation of technology development levels, and ways to assess the environmental performance of new materials (Cucurachi et al., 2018).

2.1.2 The ex-ante methodology

Prospective LCA incorporates predictions of a technology's environmental impacts at a particular time in the future, which can be applied to developing and fully developed technologies (van der Giesen et al., 2020). Ex-ante refers to when products and technologies have not yet reached manufacturing maturity (Tecchio et al., 2016). The ex-ante methodology is understood to evaluate the environmental performance of an early-stage technology, scaled to industrial levels, applying probable scenarios and comparing these results to a conventional system or technology (Cucurachi et al., 2018). Ex-ante LCA can therefore be seen as a sub-division of prospective LCA; yet, some also see prospective LCA as modelling future scenarios of technologies at an early stage of development (Arvidsson et al., 2018), making the boundary between the two methods somewhat blurry. The ISO 14040 and 14044 frameworks should equally be used for ex-ante LCA studies. Nevertheless, as a lack of available data in ex-ante studies leads to uncertainties, guidance beyond this framework is needed to ensure a comprehensive analysis (van der Giesen et al., 2020).

2.1.3 Scenario development in LCA studies

There are numerous outcomes when evaluating a future technology. It is, therefore, essential to think of changing parameters based on various assumptions that will likely influence the results. These parameters can be quantitative (technical/operational) or qualitative, surrounding parameters related to broader socio-economic aspects (Cucurachi & Steubing et al., 2021). Examples of surrounding parameters are human consumption patterns or policy decisions.

Scenarios can predict future outcomes of emerging technologies based on these changing parameters, paving the way for implementing a comprehensive ex-ante LCA (Delpierre et al., 2021). As emerging technologies are part of complex systems, scenarios are the only plausible way to assess future developments (Cucurachi & Steubing et al., 2021). The inclusion of scenarios has thus been used in environmental studies related to emerging technologies, e.g. (Ravikumar et al., 2016); (Tsoy et al., 2019); (Delpierre et al., 2021).

Given the enormous ambiguity concerning the development of emerging technologies, two types of scenarios are commonly used in ex-ante LCA: exploratory and normative (Delpierre et al., 2021). The former represents possible outlooks, whereas the latter represents desirable ones (Börjeson et al., 2006). Both are very effective in accounting for the ambiguous and complex nature and data uncertainties surrounding the broad implementation of a novel technology (Delpierre et al., 2021). Each scenario can deal with this ambiguity, yet their approach is different. Depending on the aim and scope of the research, either one of these two will be better suited. Normative scenarios are frequently related to backcasting, where a technology is modelled at a specific time (Guinée et al., 2018). In this case, an approach is developed to reach a particular goal in time (Quist, 2013). In the case of explorative scenarios, alternatives to the present are investigated, which can be done by asking questions based on "what if" (Cucurachi, Blanco, et al., 2021).

When building scenarios based on particular values for the parameters, there can be large possible combinations on the one hand and a mismatch of possibilities, such as an ambitious environmental policy leading to a minimal share of renewables, on the other hand (Cucurachi, Blanco, et al., 2021). The general morphological analysis (GMA), a method for exploratory scenario analysis, therefore, recommends a cross-consistency analysis (CCA) to evaluate if the identified parameters are aligned with each other (Ritchey, 2011). More precisely, the GMA "structures and investigates the total set of relationships contained in multi-dimensional, non-quantifiable problem complexes" (Ritchey, 2011; p 84).

Recently two frameworks were developed combining the ex-ante LCA methodology and scenario development based on the GMA. As emerging technologies are embedded into complex socio-technical systems, it is crucial to look beyond technical parameters to evaluate how these are influenced by the "big picture" or, in other words, by the surrounding, non-quantifiable parameters (Cucurachi, Blanco, et al., 2021). Therefore, the GMA is also very useful in guiding the approach to assessing emerging technologies.

The first of the two frameworks (Delpierre et al., 2021) combines technology and explorative scenario analysis to evaluate the performance of alkaline electrolyte (AEL) electrolysis and the proton exchange membrane or polymer electrolyte membrane (PEM) electrolysis at large-scale production. Consequently, the authors (Delpierre et al., 2021) developed a framework rooted in ex-ante LCA combined with the GMA. The second framework uses scenario development to guide the ex-ante LCA approach for biobased products (BbPs). This framework (Cucurachi, Steubing, et al., 2021) was developed between the Joint Research Center (JRC) of the European Commission and the Institute of Environmental Sciences at Leiden University.

2.1.4 Limitations of scenario development using the ex-ante LCA methodology

According to the authors (Cucurachi, Steubing, et al., 2021), scenario analysis could also be used for the LCIA phase to test parameters relating to the effects of novel chemicals or the impact assessment methods; yet their framework does not cover this. This lack of coverage could be due to numerous challenges posed by the LCIA phase for ex-ante LCA, for which scenario analysis might not be sufficient. LCIA, even for ex-post LCA studies, is very constricted by huge data gaps related to missing characterisation factors (CFs) for substances (Fantke, Aurisano, et al., 2018); (Fantke, Aylward, et al., 2018); (del Pilar Jiménez-Donaire et al., 2020). These data gaps are due to the high costs and workload of assessing individual chemicals (Hou et al., 2020). These limitations are further discussed under section 6.4.3.

In addition to current limitations for ex-post LCIA there are data gaps, and ignorance about future impacts related to CFs and currently undiscovered ICs referred to as "unknown unknowns" (van der Giesen et al., 2020). Some ICs might become redundant, while new impacts will arise based on novel insights and research. These "unknown unknowns" can lead to unforeseeable environmental impacts, disguising the actual environmental performance of an emerging product system versus the incumbent product system. To measure uncertainties, it is common practice to deal with "known unknowns" in ex-post LCA. Yet, the inherent ambiguity related to any future prediction of emerging technologies adds a new layer of complexity and uncertainty to the LCIA phase of ex-ante LCA (van der Giesen et al., 2020). Indeed, numerous researchers point toward the drawbacks of deficiency of CFs at the LCIA phase of ex-ante LCA studies (McKone et al., 2011); (Tufvesson et al., 2013); (Deng et al., 2017). In addition, the product system may cause unpredicted novel effects while CFs of the incumbent and emerging technology may transform in the future (van der Giesen et al., 2020). It is therefore recommended to evaluate such possibilities, while results of ex-ante LCA studies should inform discussion rather than be fixed statements, given the high levels of uncertainty.

Besides dealing with "unknown unknowns", it is crucial to account for future changes to the background systems when future product systems are assessed (van der Giesen et al., 2020). Furthermore, a mismatch between foreground and background data should be avoided when determining the impacts of novel technologies (Arvidsson et al., 2017). It should therefore be involved in the narrative to include future databases for the background processes for both the emerging technology and the incumbent (Tsoy et al., 2020); (Cucurachi, Steubing, et al., 2021). This inclusion is essential as background data used to make future predictions is often outdated, even for the current situation (van der Giesen et al., 2020). However, such databases only resemble the future energy mix, neglecting other factors.

2.2 Research approach

In this study, an explorative scenario approach was taken. The study comprises four phases, as depicted in figure 2, based on a recently developed framework for assessing novel technologies (Delpierre et al., 2021), which was further developed. A thorough literature review was conducted during the first phase, encompassing a critical evaluation of potential system improvements for the pilot-scale MP system. Consequently, specific technical parameters were evaluated, potentially improving the overall product system's performance. Based on phase one, the second phase evaluated an optimised technological setup at the pilot-scale through parameter testing using an attributional LCA. In the third phase, changing technical and socio-economic parameters were determined, influencing MP's performance as it develops from pilot-scale to large manufacturing scale. The third phase was based on the ex-ante LCA framework for BbPs (Cucurachi, Steubing, et al., 2021). Based on phase three, scenarios were built in the fourth and final phase. Subsequently, an ex-ante LCA compared MP production, progressing from pilot to large manufacturing scale to SBM production.

The framework used in this study had to be adjusted from the original framework (Delpierre et al., 2021) for several reasons: firstly, before performing a pilot-scale LCA, system improvements for the pilot-scale had to be evaluated based on shortcomings and recommendations of previous studies. As the original framework (Delpierre et al., 2021) assessed technologies whose technical development was much further than MP's, this additional step was necessary. Secondly, the original framework was used to compare production at the pilot-scale against the large-scale, neglecting progressive development (Delpierre, 2019). Thirdly, as no direct access to input from technology developers and other experts was available for this study, influencing parameters were determined based on a recommended alternative approach rooted in literature findings, a critical assessment of the current state-of-art product system, and process calculations (Cucurachi, Steubing, et al., 2021). Lastly, it seemed suitable to outline the scenarios during the goal and scope definition in both the LCA and ex-ante LCA study, as this is the most appropriate place (Pesonen et al., 2000). The scenario outline was thus placed between the goal definition and the scope definition, as the former dictated it, while the scenario outline informed the latter.

2.2.1 Phase one: literature review

As the first step, a thorough literature review was conducted assessing recommendations and limitations of previous MP studies, documented under section 3.0. Search engines such as the digital libraries of the University of Leiden and the Technical University of Delft and Google Scholar were used. Search words included: single-cell proteins, microbial proteins, and HOB, in combination with search words related to life cycle assessment, environmental assessment, environmental impact, and techno-economic assessment. The literature review outcomes determined numerous technical parameters that needed a further review to establish if they could increase the system's performance. This determination set the groundwork for the attributional LCA conducted in phase two.

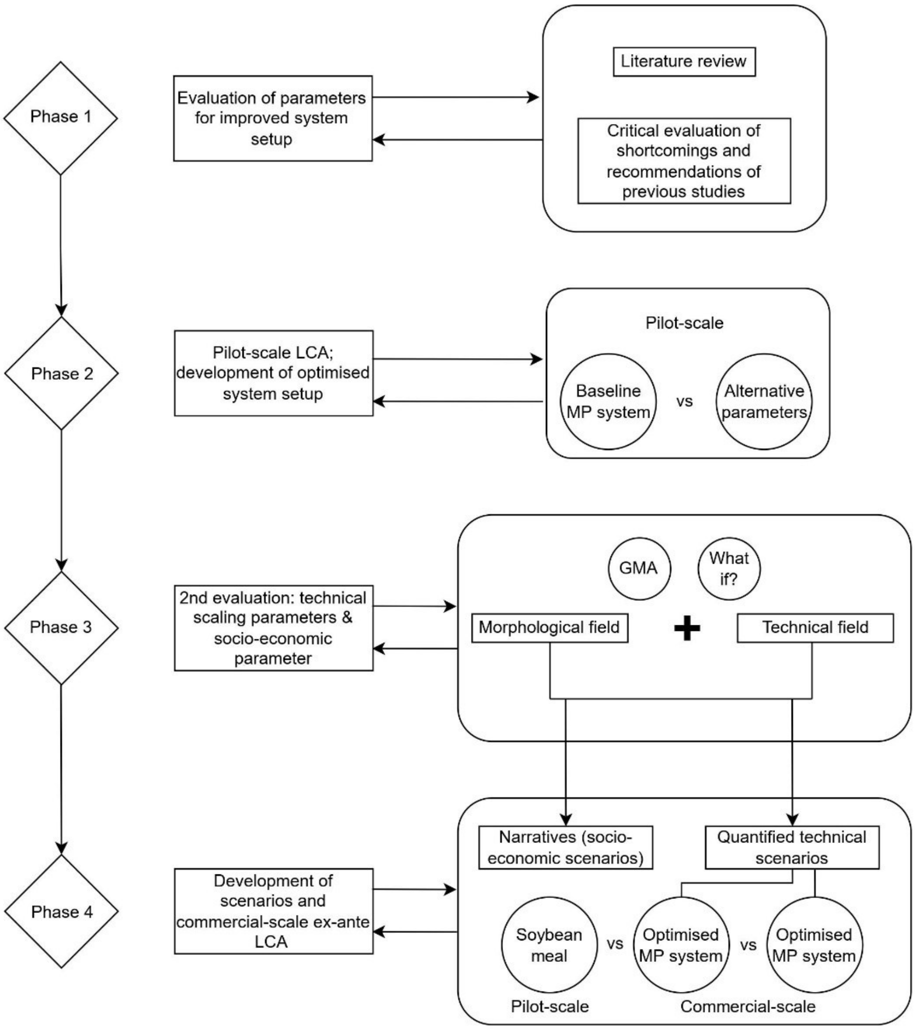


Figure 2. Methodological framework, based on (Delpierre et al., 2021)

2.2.2 Phase two: pilot-scale LCA

The second research phase aimed to determine the technological setup's "optimal" environmental performance at the pilot-scale based on currently available technologies. This evaluation was necessary before deciding on future scenarios, given the limitations and recommendations of previous studies highlighted in section 3.0.

Subsequently, an attributional LCA, defined as assessing a current product system to evaluate the associated fraction of the global burden (United Nations Environment Programme, 2011), was performed, where specific technical parameters were tested. Secondary data was used to assess the MP product system based on literature findings of previously published MP LCA studies. The most recent MP LCA study (Järviö et al., 2021) provided insight into an existing pilot-scale MP product system and its data. It was used to quantify economic and environmental inputs and outputs of unit processes, modelling the foreground system. Where data gaps were identified, older MP studies (Sillman et al., 2019); (Sillman et al., 2020) often gave valuable insight. Consequently, technical parameters were determined to test environmental improvements centred around recommendations and hotspots identified by the authors (Sillman et al., 2020); (Järviö et al., 2021), as well as shortcomings of these two studies.

The assessment was performed with an appropriate FU and system boundary. The background system, i.e., all upstream and downstream unit processes, were retrieved from the database (ecoinvent version 3.7, 2020). The 3.7 database was used as a default version, as this was the most recent one available. Copper production could not be found in the 3.7 versions. Thus, the 3.6 database was used instead for this product. The software activity browser (AB), an open-source LCA software based on the Brightway framework (Steubing et al., 2020), was used for the modelling process.

ICs can vary in their analysis approach through different characterisation models or category indicators. Therefore, the JRC (Joint Research Centre) has developed the International Reference Life Cycle Data System (ILCD) to unify LCA results while creating more transparency (Hauschild et al., 2011). This study used the ILCD 2.0 2018 midpoint ICs for both the LCA and ex-ante LCA.

The second phase of this study mainly looked at quantitative, i.e. technical parameters, and not at any surrounding qualitative ones, apart from the product price. Parameters were exchanged, one at a time, integrating sensitivity analysis (SA) into the LCI phase. Further SA and contribution analysis (CA) was performed comprehensively in the ex-ante LCA in phase four of the study and was therefore neglected at this stage. Grounded in the research of the first LCA, an optimal MP system setup was defined for the pilot-scale.

2.2.3 Phase three: parameter evaluation

Qualitative and quantitative parameters were assessed during the third phase of the research. At first, technical parameters were evaluated which would have the most influence on the system's performance as it matured over time. In the second part of this phase, the surrounding parameters responsible for the technologies' development were determined. Subsequently, the relationship between the technical and morphological fields was put into context. Further guidance from the ex-ante LCA framework for BbPs (Cucurachi, Steubing, et al., 2021) was used to integrate technical parameters into the LCI and determine their dependency on the surrounding parameters.

2.2.4 Phase four: scenario development and ex-ante LCA

Based on phase three, scenarios were developed incorporating both qualitative and quantitative parameters. Consequently, the system was scaled up to more mature manufacturing stages, and an ex-ante LCA was performed. Then, the results of the scaled-up MP product systems were compared to SBM production in a comparative assessment.

For an explorative assessment, a few diverse, so-called cornerstone scenarios resembling different ends of the spectrum should be used rather than many similar ones (Cucurachi, Blanco, et al., 2021). Yet, the ex-ante LCA performed in phase four had to take a slightly different approach as results were compared over time as the technology moved from a pilot plant production level to one with a high MPL. This approach also aligned with recommendations to evaluate the transitional development rather than just the outcome in 2050 (Delpierre, 2019). Comparing the technology's progression over time was somewhat different from having a good, medium and worst-case scenario, as recommended in the BbP environmental assessment framework (Cucurachi, Blanco, et al., 2021). However, if there are no considerable improvements in the production efficiency over time, the pilot-scale performance could also resemble the technical performance later. Therefore, the pilot-scale scenario could also be seen as a worst-case scenario, while the future scenario could be seen as a best-case scenario.

A framework (Tsoy et al., 2020) was used to guide the upscaling process, although reasonably detailed empirical data was available, potentially reducing uncertainty and increasing accuracy. Therefore, performing process simulations and using molecular structure models was unnecessary; instead, manual calculations, partly based on stoichiometry and proxies, were used, as recommended by the framework (Tsoy et al., 2020). Detailed information on these calculations can be found in appendix 1 & 2. Consequently, 12 scenarios resembling a progression over 30 years were determined. The development of the scenarios was not done linearly but rather in an iterative way, as recommended by the BbP assessment framework (Cucurachi, Blanco, et al., 2021).

The LCIA characterisation results were compared to the impacts of SBM production. Normalisation was performed using global normalisation factors (NFs) based on world reference data from 2010, applying the EF method (Sala et al., 2017). More updated models were available for ozone layer depletion and the three toxicity-related ICs. These toxicity-related ICs are carcinogenic and non-carcinogenic effects and freshwater ecotoxicity. The former was based on the WMO (WMO, 2014), and the USEtox model (Saouter et al., 2018) was applied for the latter three. USEtox is a widespread LCIA model that provides CFs for human and ecotoxicology impacts (Rosenbaum et al., 2008); (Frischknecht et al., 2016). However, the model has immense data gaps due to the high costs and workload associated with assessing individual chemicals (Hou et al., 2020). These data gaps lead to poor data quality regarding chemicals streams into the environment and unknown CFs (Hou et al., 2020).

Given these limitations, using scenario analysis in the LCIA was outside this study's scope. Instead, guidance in dealing with constraints related to the LCIA stage was based on recommendations for improved practice for assessing the environmental performance of emerging technologies (van der Giesen et al., 2020). Besides "known unknowns" and "unknown unknowns", the shortcomings of a temporal mismatch between foreground and background data systems were considered. Overcoming this challenge was especially relevant for the incumbent system, which was modelled exclusively using background data. As mentioned in section 2.1.4, future databases resemble the future energy mix, neglecting other influencing factors.

Nevertheless, such perspective databases were assumed irrelevant for MP production in the future, as all energy was modelled to come from renewable sources. For the incumbent system, energy demand is not the leading factor for agricultural production, as mentioned in section 3.6, which is why it seemed acceptable to neglect the future energy mix. Instead, a slightly different approach was taken, setting the groundwork for a new method to improve the uncertainty behind data mismatches between foreground and background data in ex-ante LCA studies.

At the core of this new method, a CA of the incumbent system was performed per IC. Process contributions were evaluated to determine to what degree changes towards those contributions could be expected in future and if this would influence the results of the comparative analysis. A CA was also performed for the emerging technology to identify hotspots and recommendations for future studies. The CA was primarily based on the AB's Sankey function. Contributions below 5% were cut off given their limited impact, and the calculation depth was set to 250. Sankey diagrams for each of the three reference flows are in appendix 1.

Through close contact with the supervisors of this study and experts at CML at Leiden University and the Department of Technology, Policy, and Management at the Technical University of Delft, further insights into the technological setup, modelling approaches, and scenario assessment were gathered. Access to this broad knowledge was precious when scaling the new product system and predicting future scenarios for the emerging technology. Reflections broadened the research approaches through various meetings and conversations with these experts from diverse industrial and academic backgrounds.

3 Phase one: literature review

This phase evaluated literature recommendations and their shortcomings. The analysis was focused on economic and environmental aspects related to the MP foreground system and its influencing technical parameters.

3.1 Continuous energy supply

Power-to-X (PtX) describes the process of hydrocarbon production using various sources of CO2, in addition to H2 generated through water electrolysis powered by renewable energy technologies (Sillman et al., 2020). Because many renewables rely on wind and sun, the electricity supply generated through these intermittent technologies has ebbs and overflows. As manufacturing costs are now at par with conventional energy production, increasing the share of renewables will inherently lead to increased supply instability (Sillman et al., 2019), which calls for solutions to utilise the occasional oversupply of power (Park et al., 2016). PtX is a possible solution to use excess energy (e.g. in the form of gas) when prices are low, making it a promising large-scale storage medium (Sillman et al., 2020). Many PtX applications have lower environmental impacts than their fossil counterparts. Extensive research is conducted on PtX technologies because of the wide-scale application of renewable technologies to overcome environmental pressures (Sillman et al., 2020). There is currently a lot of research on power-to-food (PtF); (Sillman et al., 2019). Besides using H2 to produce MP through water-splitting electrolysis, this concept also utilises Oxygen (O2) generated in the process (Sillman et al., 2020).

Even though MP production requires more electricity than SBM generation, the production can be aligned with the fluctuation in supply and demand, resulting in lower electricity prices (Sillman et al., 2020). Indeed, some bacteria, such as *Cupriavidus necator*, have lasted through the night despite H2 deprivation (Liu et al., 2016). Yet, relying on intermittent energy sources and changes in electricity supply and demand would likely have adverse effects on the stability of such a system. Additionally, due to the intensity of the capital costs for producing MP, a continuous supply of electricity and H2 is essential for this protein alternative to becoming economically competitive (Nappa et al., 2020). Therefore, the authors (Nappa et al., 2020) stress the importance of including storage solutions as part of the production process when assessing the viability of using MP as a protein feed and food alternative. Yet, both MP LCA reference studies (Sillman et al., 2020) and (Järviö et al., 2021) considered intermittent renewable energy technologies without including H2 storage solutions, thus neglecting continuous energy and nutrient supply.

Baseload energy supply was considered in the latest MP LCA study (Järviö et al., 2021), including Finish hydropower and nuclear power; both showed better results than those using wind and solar power. However, it is unlikely that baseload technologies, also called grid balancing technologies, will, or should, be readily available for any other function but to stabilise an increasingly volatile energy system. Using baseload technologies sensibly is especially important for biomass and hydropower plants affected by a supply shortage and water scarcity. An exception could be geothermal energy, as it is less dependent on natural resources and readily available in certain parts of the world.

Given these shortcomings, it was, therefore, essential to assess MP's environmental performance while considering a combination of intermittent and baseload energy sources for energy supply.

3.2 Distinguishing between biogenic and fossil CO2 sources

Carbon capture and storage (CCS) is an innovative technology that captures and permanently stores CO2, which can come from point sources, such as biomass combustion to produce energy (Kemper, 2015) or through direct air capture (DAC); (EASAC, 2018). Drax in the UK is one of a few bioenergy plants demonstrating CCS and carbon capture and utilisation (CCU) projects (Ricardo Energy & Environment, 2020). Globally, large sums of biomass are needed for bioenergy with CCS to keep an increased temperature below two-degree Celsius (Turkenburg et al., 2016). Nonetheless, biomass growth can affect eutrophication and cause LU-related issues, such as water utilisation, soil erosion, and biodiversity loss, which all must be considered carefully (Weiss et al., 2012).

CCU only temporarily stores CO2 in products. CCU applications are thus often falsely assumed to reduce or even lead to negative greenhouse gas (GHG) emissions, as can be the case for CCS (Von Der Assen et al., 2013). There is a lot of controversy about what negative emissions are, which is why this study defined them based on a literature review:

> "Upstream and downstream GHG emissions associated with the removal and storage process […] are comprehensively estimated and included in the emission balance. […] The total quantity of atmospheric GHG removed and permanently stored is greater than the total quantity of greenhouse gases emitted to the atmosphere" (Tanzer & Ramírez, 2019); (p.1216).

Based on this definition, it is crucial to further classify CCU into early and delayed GHG emissions (Von Der Assen et al., 2013), of which only the latter can mitigate climate change by reducing fossil resource depletion (Peters et al., 2011); (Quadrelli et al., 2011); (Romero & Steinfeld, 2012). It is usual to negate carbon absorbed by the body in food consumption LCA studies due to numerous necessary presumptions related to the absorption of carbon and the time it will take until it is released (Järviö et al., 2021). Therefore, in the case of MP, the authors (Järviö et al., 2021) presumed that the carbon assimilated by the microbes ends up in the atmosphere after consumption. This assumption thus classifies them as early GHG emissions. Yet, it is imperative to distinguish between biogenic and non-biogenic CO2 sources to assess CCU products.

In the case of biogenic carbon uptake by biomass, there are two ways of modelling these streams in the LCI (Arehart et al., 2021). The "-1/+1" approach accounts for both the uptake and the release of biogenic carbon from and into the atmosphere (Arehart et al., 2021). Yet this approach is problematic in a cradle-to-gate analysis, falsely implying a negative carbon uptake. On the other hand, the "0/0" method assumes carbon is taken from the atmosphere through plants after being re-released during combustion, accounting for zero net GHG emissions. In the case of MP, the gas input can be modelled as having no net GHG emissions, considering a biogenic source and the carbon to return to the atmosphere shortly after consumption.

However, despite its fossil origin, the latest MP LCA study (Järviö et al., 2021) has also used a "0/0" approach by neglecting to model CO2 as gas input. Therefore, modelling the gas with a "0/+1" approach would have been more appropriate. Figure 3 provides more clarity on the difference between using biogenic and fossil carbon sources.

Furthermore, as CCU turns CO2 from emission into feedstock, it is essential to correctly allocate the gas as it is always a by-product of another process such as electricity or ammonia (NH3) production (Von Der Assen et al., 2013). Yet, in the study (Järviö et al., 2021), such an approach seems to have been neglected, as the LCI only included "liquid carbon dioxide production out of waste gases from different chemical production processes". Nevertheless, as no upstream emissions are associated with waste, the production of liquid CO2 is merely one process that captures and cleans the gas. Therefore, this approach did not consider any upstream emissions caused by the co-production of CO2 and different chemical production processes. As the current market prices for CO2 in the UK range from 70-180£/t (Ricardo Energy & Environment, 2020), modelling it as a waste input seems inappropriate. Thus, allocation for the co-production of various chemical processes and liquid CO2 should be performed in such cases. Had the authors (Järviö et al., 2021) performed such an approach, net GHG emissions would have likely been higher.

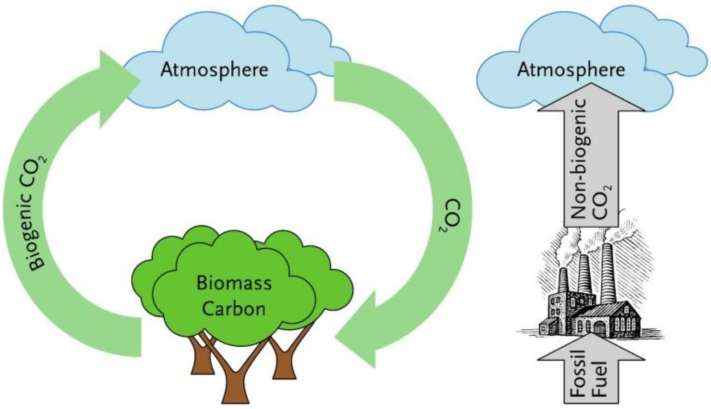


Figure 3. Fossil vs biogenic CO2 emissions (Technology Collaboration Programme, 2022)

Similarly, the other MP LCA study (Sillman et al., 2020) also modelled CO2 as a waste flow, assigning no upstream emissions associated with feedstock production. Given these modelling choices of the only two MP reference studies, comparing biogenic and fossil CO2 sources as input for MP production seemed necessary while allocating CO2 as a by-product in the latter case.

3.3 Heat Source

Apart from CO2, heat was previously identified as a hotspot of MP production (Järviö et al., 2021). When producing low-pressure (LP) steam on-site using renewable energy instead of supplying it through the chemical industry with the average energy mix, impacts on climate change and terrestrial acidification considerably decreased. Yet, values in all other ICs increased (Järviö et al., 2021). Therefore, evaluating sustainable solutions for heat supply for MP production was essential. One study (Sillman et al., 2020) has included industrial waste heat as a thermal energy source in one MP production scenario, assigning no emissions to it. Yet, the circular economy naturally designates an economical rate for all streams (Olofsson & Börjesson, 2018). Heat should thus be modelled accordingly.

3.4 NH3 source

Another central input for MP production is nitrate (N2) through NH3 or ammonium, which both reference studies (Sillman et al., 2020) and (Järviö et al., 2021) have identified as an essential contributor to various ICs. Based on the best-case scenario (Järviö et al., 2021), the production of NH3 contributed 30.6% towards water scarcity, 27.5% towards climate change, and 24.4% towards terrestrial acidification, and just below 10% towards freshwater and marine eutrophication as well as non-carcinogenic human toxicity. In both reference studies, NH3 is produced via the Haber-Bosch process, the most common production method worldwide. Yet, as this process uses natural gas as an H2 source, it is not only fossil dependent but displays high environmental impacts related to the process's energy requirements (Udvardi et al., 2015).

Therefore, assessing the effects of using more sustainable NH3 sources for MP production was essential.

3.5 Assessing the challenges of early-stage technologies for MP production

To comprehensively assess novel technologies, looking at their stage of development is crucial, as it takes considerable time until they operate in a comparable way to the incumbents (van der Giesen et al., 2020). As environmental burdens caused by emerging technologies at a low maturity level are probably not linearly scalable, the results of such an analysis must be presented concerning their scale of operation (Gavankar et al., 2014). The technology readiness level (TRL) and manufacturing readiness level (MRL) are frameworks to measure the developmental stage of a technology at a certain point in time. The indicators range from the lowest level, namely conceptual development (TRL/MRL 1-4), to the highest level, namely small-scale production (TRL/MRL 8-9) and mass production (MRL 10); (Gavankar et al., 2014).

In contrast to the TRL, the MRL goes beyond assessing the technology's functional readiness, including evaluating elements or subsystems needed to reach manufacturing maturity (Gavankar et al., 2014). In addition to these two concepts, current studies suggest including the broader market environment and market penetration dynamics expressed through the market penetration level (MPL); (Bergerson et al., 2020); (Hulst et al., 2020). It is necessary to include the MPL, as economies of scale increase productivity, resulting in further energy and material efficiency. Once technologies reach a maximum MRL of 10 (MPL 0-5), the MPL determines whether a technology is at an early industrial production level (MPL 0-50) or a mature production level (MPL 50-100; figure 4); (Hulst et al., 2020). It usually takes technologies around 25 years from early developments, with TRL/MRL 0, to compete with the incumbents, even though this period appears to decline for more recent innovations (Hirooka, 2006). Once the novel product or technology enters the market, reaching full maturity will take a comparable time frame (Kramer & Haigh, 2009).

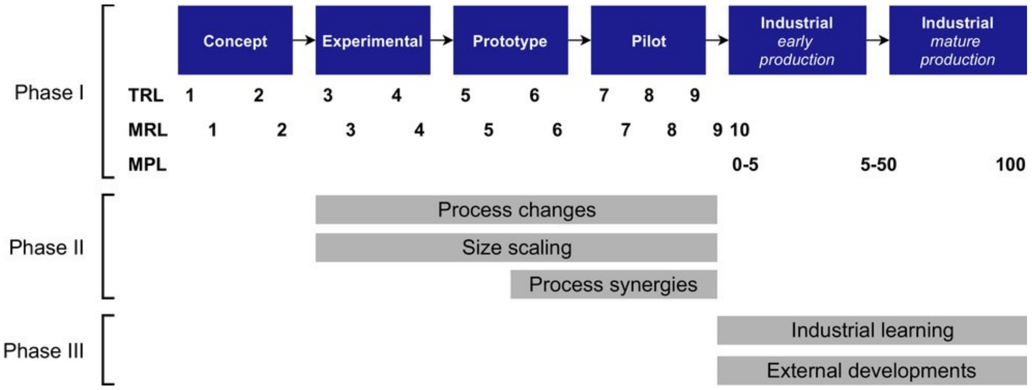


Figure 4. Development stages of emerging technologies (Hulst et al., 2020)

Poor-quality data is often used for some LCA studies on emerging technologies with a low TRL and MRL (Gavankar et al., 2014). Yet, waiting for these technologies to reach manufacturing maturity (i.e., TRL of 9 and MRL of 9-10) would defeat the purpose of early-stage environmental assessment, which is used to stir the decision-making process. Consequently, early-stage analysis is essential to avoid otherwise unforeseen burdens on the environment and lock-in effects (van der Giesen et al., 2020). Yet, several studies assessing mass production scenarios (i.e., MRL of 9 or 10) are based on pilot-scale data with a TRL of 6 or 7 (Gavankar et al., 2014), not representing operational scales (Arvidsson et al., 2014). Such a data mismatch is also the case for the first reference study (Sillman et al., 2020) that was primarily based on estimations, neglecting the TRL entirely.

While influencing a technology's efficiency, it will only reach a high MPL if the cost of production is economically competitive. Excluding the production price in the analysis might lead to technologies with a high TRL/MRL not getting to a high MPL by failing to penetrate the market. As previously mentioned, wind and solar-powered water electrolysis for MP production showed promising environmental results (Sillman et al., 2020); (Järviö et al., 2021), at least compared to animal-based proteins. Yet, water electrolysis powered by renewable energy sources for H2 generation could only become competitive with their fossil counterparts if capital expenditure and electricity production costs were reduced substantially (Al-Qahtani et al., 2021). The total cost of H2 (TCH) production is made up of the levelized cost of H2 (LCOH), which is primarily connected to the readiness of some of the assessed technologies and the monetised environmental impacts (Al-Qahtani et al., 2021).

Currently, the most implemented technology for commercial electrical water splitting worldwide is the AEL electrolyser (Ursua et al., 2012), followed by the PEM electrolyser (Carmo et al., 2013); (Delpierre et al., 2021). Even though these so-called green H2 production technologies (Al-Qahtani et al., 2021) are fully developed and commercialised, they face challenges for global integration linked to their productivity (Dotan et al., 2019). Indeed, poor efficiency levels of electrolyte electrolysis drive high production costs and hamper competition with conventional, fossil fuel-based H2 generating technologies (Al-Qahtani et al., 2021).

Under optimised circumstances, where volumetric productivity is at its maximum, while H2 and electricity are continuously supplied through cheap energy sources, the current price of MP is estimated to be 2.1€/kg, and that of soybeans is 0.27€/kg (Nappa et al., 2020). As at least 0.28kg H2 are needed per kg MP (see appendix 1), and given the current green H2 production price between 6-10€/kg (Squadrito et al., 2021), 2.1€/kg MP produced presumes a drastic reduction in green H2 price. Additionally, it should be noted that comparing MP to soybeans, the former displays a higher protein content per kg produced (Nappa et al., 2020). Even when considering the protein content, a drastic price difference remains, which shows the importance of considering the costs of technologies or technological components when performing an LCA on developing novel products to guide the decision-making process. Nonetheless, the TCH has been neglected by both previous MP LCA studies (Sillman et al., 2020); (Järviö et al., 2021).

As production costs and environmental emissions should ideally decrease over time, it is necessary to clearly define an LCA study's temporal scope when assessing a technology's future performance. The temporal scope defines when economic and environmental results can be expected. Still, this definition and thus the likelihood of specific scenarios accruing at a certain point in time has been neglected by the reference studies. Water electrolysis consumes around 75% of the electricity needed for the foreground system; thus, the latter is crucial in determining the environmental performance (Järviö et al., 2021) and also the overall costs of MP production. Two critical parameters for MP production are thus nutrient utilisation and electrolysis efficiency (Sillman et al., 2020); (Järviö et al., 2021). In the study's best-case scenario (Järviö et al., 2021), the percentage of in situ electrolysis efficiency and nutrient utilisation have been modelled at 79% and 99%, respectively, while the pilot data was measured at 60% and 85-90%, respectively. Even though 100% nutrient utilisation is theoretically possible through a closed system design (Lee, 2015), these values seem arbitrary without indicating when and under what circumstances to expect such a performance.

A full techno-economic assessment of the various technical setups for MP production would have exceeded the scope of this study. Yet, as the aim was to assess MP production at a high

MPL, assuming a competitive production price, this assessment incorporated an economic feasibility assessment. Therefore, it was essential to look at alternative, sustainable, price-competitive H2 sources for MP production, besides PEM and AEL, while including a realistic outlook on when these technologies will be ready to penetrate the market.

3.6 Expanded assessment of ICs

The industrial energy demand for agricultural products is not the primary driver for GHG emissions (Poore & Nemecek, 2018), as is the case for MP production, making the source of electricity the leading cause of environmental impacts (Sillman et al., 2020); (Järviö et al., 2021). Despite the high energy demands for MP production, both reference studies showed relatively low GHG emissions for MP production due to incorporating renewable energy systems in the analysis. Nevertheless, a recent publication on various H2 production technologies (Al-Qahtani et al., 2021) showed the importance of holistic assessment in LCA through a vast inclusion of ICs, especially when assessing bioenergy. For example, biomass gasification (BG) with CCS has the potential of sequestering 14.63kg CO2/kg H2 produced, which solar and wind energy technologies could not achieve.

Yet, when the authors (Al-Qahtani et al., 2021) assessed endpoint indicators based on human health, ecosystem quality, and resource depletion, BG with CCS performed much worse than solar and wind energy in all three. In addition, BG with CCS also performed worse on the first two indicators due to the high water and LUC linked to the biomass plantation phase compared to LP steam methane reforming (SMR) with CCS (Al-Qahtani et al., 2021). SMR is currently the standard method for producing H2 globally. Furthermore, due to its low H2 yield, generating the global annual demand for H2 through biomass would require almost half of all available cropland in the US (National Research Council and National Academy of Engineering, 2004).

Despite the importance of a holistic environmental assessment, the first MP LCA (Sillman et al., 2020) only included ICs related to conventional agriculture: LU, water scarcity, climate change, acidification, and eutrophication. The second study (Järviö et al., 2021) has expanded the assessment, including ozone depletion and human non-carcinogenic toxicity. However, only six ICs were considered in the publication (Järviö et al., 2021). Both studies still fall short of including other ICs related to human health, ecosystem quality, and resource depletion, other than water and LU, such as minerals and metals and fossil use. Including resource use, however, is essential, given the reliance of wind turbines on rare earth metals. Therefore, a holistic assessment of MP production based on previously neglected ICs was necessary.

3.7 Summary of literature gap

Table 1 summarises the recommendations and limitations of previous studies, discussed in sections 3.1 – 3.6.

Table 1. Summary of research aim based on previous studies

Origin of issue	Issue
Renewable energy	continuous energy production & H2 storage was not considered
CO2	not allocated; no differentiation between biogenic & fossil CO2
Heat	not allocated; identified as a hotspot
NH3	identified as a hotspot
Technology development	manufacturing maturity & timescales have been neglected
Material & resource efficiency	the scaling parameters were not considered
Surrounding parameters	were not considered (e.g. cost of production)
ICs	not comprehensive enough

4 Phase two: the pilot-scale LCA

4.1 Goal and scope definition of the pilot-scale LCA

At first, the goal of the LCA study was made explicit. The scope was set concerning temporal, technological and geographical coverage.

4.1.1 Goal

This attributional LCA study intends to evaluate MP production's best environmental system performance while including shortcomings related to modelling choices of previous MP LCA studies discussed in phase 1. Thus, the aim was to assess MP production using a combination of intermittent and baseload energy sources while determining economic feasibility. This goal is aligned with the second sub-research question defined in section 1.1. Only biomass combustion is considered as baseload energy supply at this stage, as other solutions are assessed in the study's fourth phase. The reason for determining an optimal system performance was to assess if MP could compete with SBM, which was evaluated in phase four.

Furthermore, the results of this second phase aimed to inform parameter choices for the ex-ante LCA study conducted in phase four, where the MP system is scaled from a pilot-scale to a commercial scale. In addition, the goal is to determine suitable biogenic CO2 allocation methods for the feed alternative's production, in line with the first sub-research question. Bioenergy using point sources is thus at the heart of this analysis.

4.1.2 Scenario development for optimal system performance at pilot-scale

The following parameters were tested (table 2), aiming to find an optimal system setup to improve the overall environmental performance of MP production at a pilot-scale while including shortcomings of previous studies. The parameter choices were based on the review conducted in phase 1.

Table 2. Technological parameters that were tested in the analysis

Parameters to be tested	Baseline-scenario	Alternative-scenario
Source of LP steam	Supplied through the chemical industry	1) Natural gas; produced on-site through 2) BCHP or 3) geothermal energy
Source of NH3	Supplied through the chemical industry	Produced on-site using renewable H2
Wind turbine	1MW	3MW
H2 source	AEL electrolysis	SMR with CCS
CO2 source	Supplied through the chemical industry with fossil-origin	Biogenic origin: 1) point source – PCC from BCHP plant; 2) DAC

Three alternatives were assessed instead of LP steam from the chemical industry: heat from natural gas, heat from a biomass combined heat and power (BCHP) plant using combustion, and heat through geothermal energy.

As the environmental impacts of the MP product system primarily depend on the H2 source (Sillman et al., 2020); (Järviö et al., 2021), it should be of renewable origin. However, as extensively elaborated in section 3.5, other price-competitive solutions needed to be evaluated to reduce the price of MP production. SMR with CCS has a lower TCH caused by a lower LCOH than its renewable counterparts (Al-Qahtani et al., 2021).

SMR with CCS, a so-called blue H2 production method, was chosen as an alternative to water electrolysis, despite being a fossil feedstock. This choice was justified as CCS could be an essential method leading toward a clean H2 economy (Simbeck, 2004), as greener technologies could gradually replace less sustainable production methods over time while the infrastructure is mainly available (Meadowcroft, 2009). Yet, even though CCS improves the environmental performance of SMR in the short term, this is no viable long-term solution for H2 production, as reservoirs to store CO2, which is converted from fossil sources through SMR, have limited capacity (Al-Qahtani et al., 2021). Additionally, though being the most cost-effective economically speaking at present, SMR with CCS cannot compete with nuclear or wind-powered water electrolysis when comparing the monetised environmental impacts (Al-Qahtani et al., 2021). The ecological effects of blue H2 generation to produce MP have not been evaluated. These technologies, therefore, need careful determination to assess if this is a viable solution for MP production in the short term.

Besides SMR with CCS, CH4 pyrolysis is another blue H2 production method, bridging the gap between the conventional, fossil-based grey production and the green methods using water electrolysis through renewable energy sources (Sánchez-Bastardo et al., 2021). Comparing the two blue H2 production methods, CH4 has the clear advantage of producing solid carbon as the sole by-product (Sánchez-Bastardo et al., 2021). Solid carbon can be used in economically large-sale applications such as soil amendment and environmental remediation, building and construction materials, and electricity production by direct carbon fuel cells (Muradov & Veziroğlu, 2005). On top of reducing CO2 emissions compared to grey production methods, solid carbon from catalytic CH4 splitting in cement and metallurgical plants would further reduce these GHG emissions (Muradov & Veziroğlu, 2005). Levels of carbon in the soil lost through erosion, LUC, and tillage, are responsible for the soil's quality and fertility (Anderson et al., 2018). Adding carbonaceous products to the ground can thus increase plant growth and harvest (Muradov & Veziroğlu, 2005).

The feasibility of such applications for solid carbon still needs further assessment (Muradov & Veziroğlu, 2005); regardless, storing solid carbon is cheaper than sequestration of CO2 derived from carbon-capturing processes (Amin et al., 2011); (Kang et al., 2020). Given the exhaustion of natural gas reserves, this is no long-term solution, yet probably the cheapest blue H2 production method, bridging the gap toward a clean H2 economy (Machhammer et al., 2016); (Parkinson et al., 2018). Still, in another study, the H2 production costs for SMR with CCS were lower than CH4 pyrolysis, even though the former results were more uncertain due to transport and long-term CO2 storage costs (Timmerberg et al., 2020). In addition, a low H2 price and GHG emission level for CH4 pyrolysis depend on the solid carbon market (Timmerberg et al., 2020). CH4 pyrolysis has a TRL of 3-5 (Al-Qahtani et al., 2021), increasing uncertainty. Given these limitations, only SMR with CCS was chosen as a cheaper alternative to green H2 production methods, given a higher TRL of 7-8 and more data availability.

Two options were tested as an alternative to CO2 from the chemical industry. The first option was to produce CO2 through a small-scale, 4.000 t/year DAC unit, and the second option was to retrieve the gas through a post-combustion capturing (PCC) unit in combination with a BCHP. On-site, green NH3 production was tested for a small-scale production unit of 20.000t/year as an alternative for supply from the chemical industry. N2 was provided from the air through an air separation unit, and H2 was provided through water electrolysis, besides CCS with SMR. In addition, different size windmills were also tested to guide the early decision-making process.

Some authors (Matassa et al., 2015) have suggested using phosphate and sulphur from waste streams. Regardless, using wastewater for nutrient supply had little effect on the overall MP product system's EF (Sillman et al., 2020). Using such waste streams was thus not further considered.

4.1.3 Scope

This assessment was a detailed LCA study, following the ISO 14040 and 14044 standards, which usually require between 20-200 person work days (Guinée et al., 2002). The scope was a cradle-to-gate analysis, even though a cradle-to-grave analysis is the most comprehensive method of analysing a product system from a life cycle perspective. However, in the case of feed and food products, a cradle-to-gate analysis can be adequate as proposed by the product environmental footprint (PEF); (Zampori & Pant, 2019) or some GHG accounting systems (Cucurachi & Steubing et al., 2021). Additionally, when comparing two systems that share identical downstream life-cycle stages, these stages may be left out of the analysis (Guinée et al., 2002). As this is true for the reference flows used for this comparative analysis and the later ex-ante stage, a cradle-to-gate assessment seemed sufficient.

Besides the "0/0" modelling approach for biogenic carbon absorbed by BbPs, some characterisation models also use the "-1/+1" method, as discussed in section 3.2. Yet, given the latter's limitations for a cradle-to-gate analysis, the PEF recommends simply reporting the biogenic carbon flows while using a "0/0" modelling approach (Cucurachi & Steubing et al., 2021), which was the method used in this study (figure 5). Yet, fossil CO2 sources were considered for the baseline scenario, calling for a different CO2 modelling approach. In this case, fossil CO2 emissions were modelled as direct emissions into the air, despite not including the consumption phase. Using a "0/+1" approach discussed in section 3.2 seemed appropriate, as a negative carbon footprint would otherwise skew the results.

Even though MP production through industrial methods presumably has adverse effects on biodiversity, regardless of the energy source (Järviö et al., 2021), these effects would be difficult to assess through the LCA methodology, given its limitation in this respect (Notarnicola et al., 2017). Impacts on the loss of biodiversity were thus not further considered in this study.

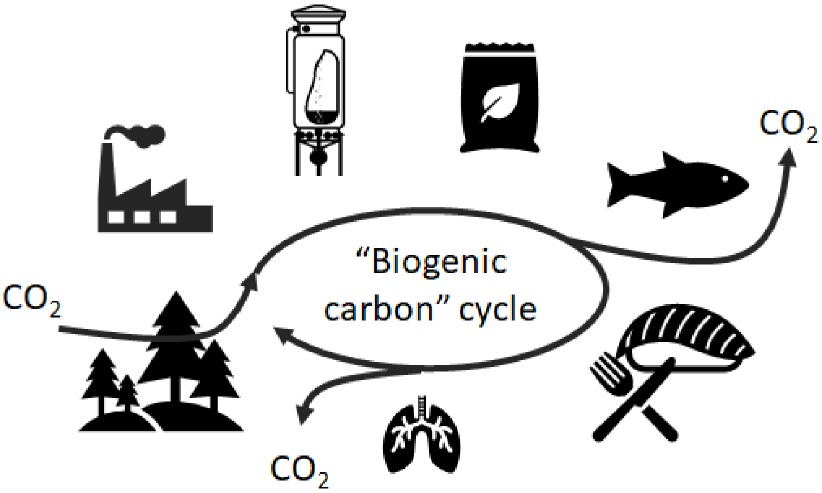


Figure 5. Biogenic carbon cycle from biomass plantation to MP consumption

4.1.3.1 Technological scope

The assessment included most upscale and downscale processes related to the MP product system, with some exceptions discussed under section 4.2.3. All operations associated with MP production are based on LCI data of a pilot-scale unit (Järviö et al., 2021), with a TRL assumed to be in the range of 5-7. Wind-based water electrolysis has a TRL of 9, yet a low MPL due to poor efficiency levels leading to high costs (Al-Qahtani et al., 2021), as discussed in section 3.5. While AEL is fully developed, PEM has a TRL of 5-7 (The Royal Society, 2018). Given its higher TRL, AEL is the most widely used electrolysis production method, yet, PEM is gaining market momentum due to its greater efficiency related to higher current density production (Carmo et al., 2013). Yet, minimal differences were examined between the two technologies at large-scale commercial production, using the ex-ante LCA methodology (Delpierre et al., 2021). Variations between the two systems for MP production were thus not evaluated in this study, as such a comparison would have had limited effects on the results.

Therefore, one technology had to be chosen over another. This way, priority was given to the detailed assessment of more crucial system parameters. For the future performance of the two electrolytic H2 production technologies, all ICs in scenario A of the comparative study (Delpierre et al., 2021) performed better for AEL except for ozone depletion and acidification. In another scenario (scenario B), differences between the two technologies were negligible apart from acidification, where AEL performed worse (Delpierre et al., 2021). Analysing these results further, the planetary boundaries for acidification and ozone layer depletion have not yet exceeded the safety zone (figure 6). This containment within the safety zone is contrary to other ICs, such as climate change and eutrophication-related biogeochemical flows, where the safe zones have been exceeded. Thus, preference was given to AEL over PEM in this study, given its better data availability and slightly better environmental performance according to the planetary boundaries.

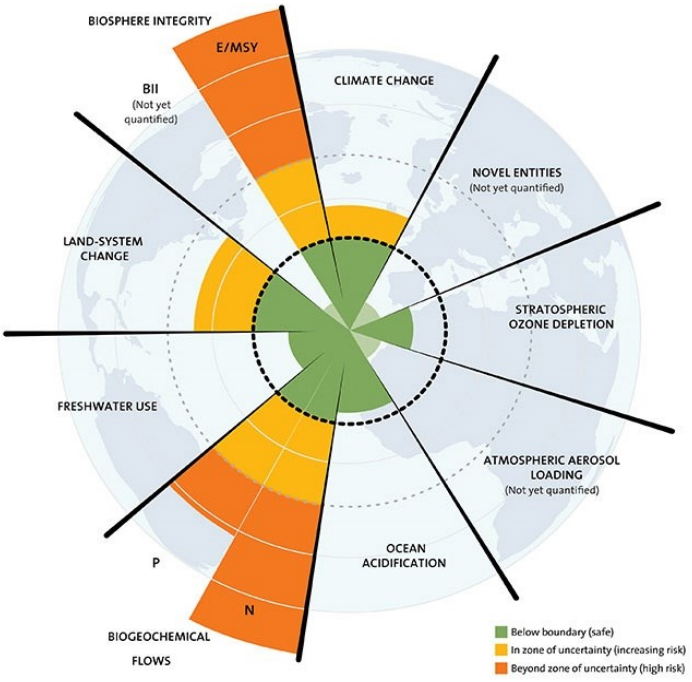


Figure 6. Planetary boundaries (Steffen et al., 2015)

DAC systems are mainly categorised as no-temperature, low-temperature, and high-temperature production. Only low-temperature has reached a TRL of 9, with the other two methods still developing with TRLs below 6 (Viebahn et al., 2019). Climewors is the widest-known low-temperature DAC company (Fasihi et al., 2019), which is currently one of the biggest commercially available producers (Viebahn et al., 2019). Indeed, Climeworks is a frontrunner in DAC in Iceland, whose commercial plants in Hinwil and Hellisheiði already display efficiency levels of 85.4% and 93.1%, respectively.

Large-scale commercial facilities using variable renewable energy sources, such as wind and solar, for NH3 production through electrolytic H2 are currently under development with a TRL of 8 (IEA, 2021). Cachimayo, a small 25 MW electrolyser facility in Peru powered through hydropower, is the only plant worldwide producing NH3 through electrolytic H2 (IEA, 2021).

4.1.3.2 Temporal Scope

The temporal scope of this assessment was based on the current stage of MP market development. The data for all foreground processes was retrieved from recently published sources, mainly from 2021. The oldest publication used for the LCI data was from 2001 (Spath & Mann, 2001). However, this was used for SMR, which has been and remains the incumbent technology for H2 production. Therefore, improvements to this technology over the last twenty years seemed minor and negligible compared to the benefit of providing the most comprehensive data.

4.1.3.3 Geographical Scope

The technological setup was assumed to be based in the UK, where Drax, a frontrunner in biomass combustion with CCS and CCU applications, is situated (Ricardo Energy & Environment, 2020). All background data was modelled accordingly. All necessary transportation to this plant was considered where applicable.

4.1.4 Function, FU, and reference flow

The function: the production of MP.

The FU: the production of 1kg of MP before packaging, produced under continuous energy supply with 65% protein and 5% moisture content.

The reference flow: the production of 1kg of MP, before packaging, produced under continuous energy supply with 65% protein and 5% moisture content using biomass combustion as a baseload energy solution.

Suggestions have been made to establish FUs based on nutritional indexes (Saarinen et al., 2017); (Sonesson et al., 2019). Nevertheless, the main reason for choosing the FU based on the product's weight was to make this study easily comparable to the only previously published LCA study on MP production grounded in empirical, first-hand data (Järviö et al., 2021).

4.2 LCI of the pilot-scale LCA

First, the product system was defined based on the system boundary. The study aimed to assess all upstream and downstream flows within the system boundary, to be as comprehensive as possible. Through the LCI, these inputs and outputs were quantified. Yet, due to data gaps, this was not always possible. In addition to known data gaps discussed in section 4.2.3, it is likely, that there was unawareness of certain missing information in some cases that led to gaps in the LCI model. Therefore, all decisions and assumptions were reflected upon and documented precisely and transparently to reduce the likelihood of missing information. Based on a comprehensive analysis, it was assessed if cut-offs were necessary or if using a proxy would lead to more accurate results.

4.2.1 Economic/environmental system boundary

Flows to and from the system boundary were classified into economic or environmental flows. The former flows are artificial, even though they can be traced back to numerous environmental flows. The latter, on the other hand, are direct flows to or from the environment without previous human intervention.

In the case of biomass, the boundary between the economy and the environment might be less precise. This study considered biomass an economic flow with limited resource capacity. The assumption was that it came from sustainably managed forests. Despite being of biogenic origin, the CO2 gas was also considered an economic flow, as the biomass needs to undergo a row of production processes to make the gas available from the biomass.

Wind for H2 production or N2 from the air for NH3 production was considered an unlimited and readily available environmental flow. Water for the electrolysis process was regarded as an economic flow. If salt water were used instead for this process, which is endlessly available, this would be considered an environmental flow; yet, the produced H2 and O2 gas would still be regarded as economic flows, for the same reason mentioned for the biogenic CO2.

4.2.2 Flow chart

A flow chart was created, as depicted in figure 7, showing all inputs and outputs to and from the product system. At the same time, the system boundary between the economic and environmental systems was determined. Environmental flows are usually not depicted in a flow chart; however, this was done to emphasise the relation between the economic and environmental system boundary.

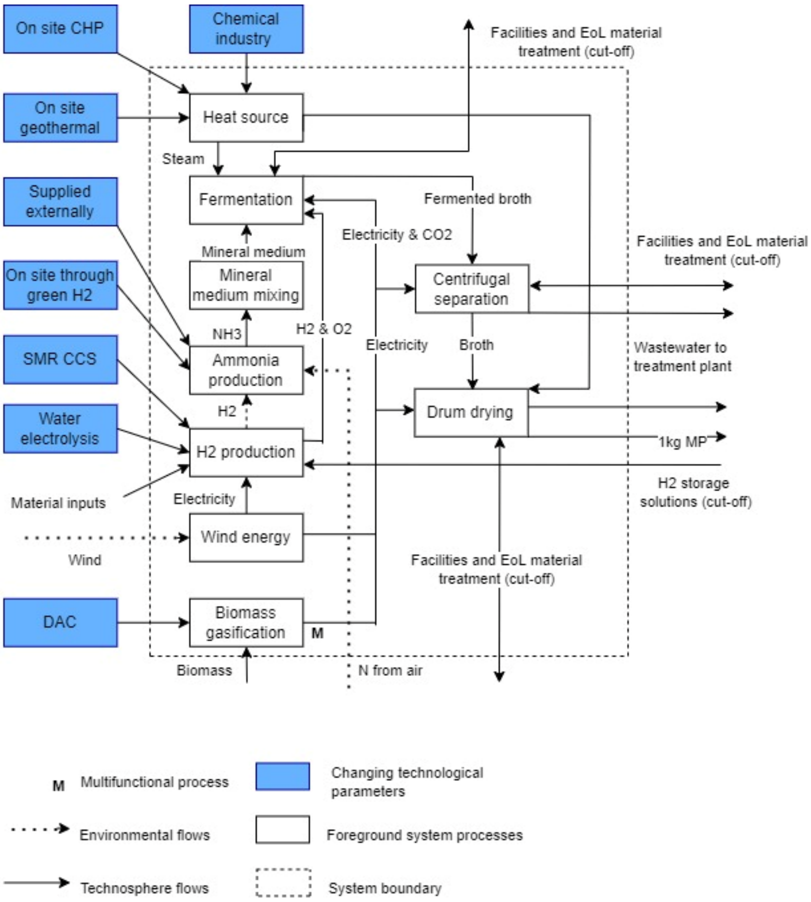


Figure 7.Flow chart of MP production system with varying technological parameters

The first MP production step takes place in a continuous stirred-tank bioreactor where HOB are augmented to a manufacturing volume of 200m3; simultaneously, the growth process happens under stable circumstances (Järviö et al., 2021). The primary feedstocks in the bioreactor are CO2 gas as a carbon source, H2 gas as a chemical energy source, and O2 gas. The latter two gases are derived through electrical water splitting. Other inputs into the fermentation process are inorganic salts such as phosphorus, sulphur, and NH3 as an N2 source (Järviö et al., 2021). A detailed list of nutrient inputs and all other foreground processes is in appendix 2. Even though the nutrients are supplied to the bioreactor directly, they were modelled as a separate process. This modelling choice was due to LCI data availability and transparency reasons. After fermentation, the broth is pasteurized using low-pressure (LP) steam at 120°C.

Consequently, the broth is centrifuged to separate the water from the biomass. After the centrifugal separation, the cell effluent is dried to a powder with 5% moisture and 65% protein content (Järviö et al., 2021). The equipment needs to be cleaned every three months using sodium hydroxide, nitric acid solutions and water (Eide et al., 2003), which was also modelled as a separate process.

As no emissions are released into the environment directly during H2 production, the emissions caused are strongly interlinked to the material inputs of the electrolysis plant and energy inputs (Koj et al., 2017). For the AEL electrolysis system, many components such as tanks, heat exchangers, pumps, electronics, and filters were thus considered. Furthermore, the following materials were included: materials needed for the membrane, aramid fibres, gasket manufacturing, and various metals to produce cathodes and anodes. Other material inputs were related to the creation of cells and cell stacks, considering their lifetime.

Additionally, the following resources were included: water as a means of cooling, deionised water for the water-splitting process, and a potassium hydroxide solution (KOH) used as an electrolyte for the AEL. The latter was assumed to have a life expectancy of 10 years. A KOH steel filter of 145kg was also considered. Due to chemical parameters based on stoichiometry, 9kg of water is needed to produce 1kg of H2; this value is currently 10kg/kg H2 created (Delpierre et al., 2021). A 20-year life expectancy for the electrolysis plant and a stack lifetime of 10 years were considered. Furthermore, the plant was assumed to be operational for 95% of the year (8300h/year), while five run-ups were considered where steam is used for heating purposes, and N2 is applied for cleaning.

For all scenarios, based on varying technical parameters, wind energy was chosen in this study as the only intermittent source of energy to supply H2 and O2, as it outperformed solar energy for MP production in various locations across Europe, especially in more northern regions (Järviö et al., 2021). As the production occurs during oversupply, the H2 and O2 gases must be stored. Electricity needed for the foreground systems is supplied through wind energy and biomass combustion when the wind is unavailable. This availability ratio was taken as 35% to 65%, respectively (Delpierre, 2019).

4.2.3 Cut-off

Facilities for the core MP production process were excluded, except for impacts related to LU for the MP production unit, as no data could be retrieved. Additionally, all process equipment for MP production related to fermentation, centrifuge, and drying, was also negated due to data availability issues. As no process equipment was considered, end-of-life treatment was also excluded. Yet most of this equipment is assumed to be stainless steel for the fermentation tanks, which has a high life expectancy and can be recycled at the end-of-use stage. Therefore, this equipment's overall impact was deemed negligible over the facility's lifespan. The end-of-life treatment for the electrolysers was also neglected due to missing data. Due to data availability, very detailed electronic equipment was also not considered for the foreground system. Facilities and all process equipment were also ignored for the green NH3 production plant as no data could be found. As steam was assumed to be kept in a closed loop, its overall impacts seemed negligible. Steam used for the fermentation and drying process was thus modelled as heat, ignoring the necessary water input.

Additionally, purification, compression or storage solutions were not considered for H2, O2, and NH3 gases, which have also been neglected in previous studies. An appropriate assessment should have included various storage possibilities ranging from salt caverns to high-pressure storage tanks. Nonetheless, as renewable H2 production is an emerging technology, no LCI data was found on such storage facilities. Such an analysis would require an individual LCA study beyond the scope of this thesis. However, energy for cooling NH3 has been considered.

As O2 has a positive market value, it should be considered a by-product of the fermentation, H2, and NH3 processes. Regardless, all emissions of the electrolysis were assigned to H2 and none to O2 production. The latter was thus cut off.

4.2.4 Multifunctionality and allocation

The BCHP process is multifunctional, producing three products: electricity, heat and CO2. It was thus essential to allocate emissions accordingly. Based on the information presented in section 3.2, the question remained of how to price CO2. The ISO 14044 standard favours allocation based on physical relationships, yet this is challenging for BbPs due to a lack of specific physical characteristics (Cucurachi & Steubing et al., 2021). In the case of biomass combustion, for example, the output is energy in the form of electricity and CO2. The latter has no energy, and the former has no mass. Given these reasons, basing the allocation on physical relationships was impossible, and another method for solving multifunctionality was necessary.

The economic value is the only appropriate way to allocate CO2 for CCU (Von Der Assen et al., 2013). Even though economic allocation can be a suitable method in many cases, it raises challenges when performed in an ex-ante LCA due to uncertainty of certain market developments based on price fluctuations, government interference and technological development (Ahlgren et al., 2015); (Njakou Djomo et al., 2017). Indeed, the operating cost of capturing facilities is hard to determine and depends on various factors (Kim et al., 2013). Additionally, it is hard to predict the future CO2 price due to high fluctuations measured through various market trading schemes, such as the ETS (Emissions Trading System). Some estimate a CO2 price of concentrated gas between 60-450 US$ per ton (Quadrelli et al., 2011). Other sources predict the CO2 price will soon be zero or even negative due to increased efforts to sequester or store CO2, based on rising climate change concerns (Centi & Perathoner, 2011); (Quadrelli & Centi, 2011).

Besides trading prices, which rely on broader economic developments, there is a concrete price to build and operate a carbon-capturing unit, which seemed appropriate as an economic allocation factor. Some state a CCS price between 265-350€/t CO2 would present a viable investment case (Aalbers & Bollen, 2017). Assuming the expenditures related to CO2 transport and storage to be 19£/tCO2 (Ricardo Energy & Environment, 2020), which is approximately 25€/tCO2, the price for carbon capture without storage, is, therefore, assumed to be between 240-325€/t CO2. According to the British Royal Society, the capturing price, excluding storage, is 140-270$/t CO2 (Ricardo Energy & Environment, 2020), equivalent to approximately 120-230 €/t CO2 (XE, 2022). Nevertheless, it was also noted that Global Thermostat, another low-temperature DAC company capable of capturing CO2 from the air and point sources, has future ambitions to deliver the gas for as little as 11-38€/t CO2 (Fasihi et al., 2019). As a price of 120€/t is within this current range, it was used as the default value.

4.2.5 Data sourcing

To avoid incompleteness and possibly to mislead results, the depth of analysis of this attributional LCA was as thorough as possible. The most recent secondary data was used for all processes related to MP production based on a comprehensive analysis of a pilot-scale setup (Järviö et al., 2021), which was the most recent data available. For the electrolysis, data was based on projections of a not-yet operational 6MW AEL plant (Koj et al., 2017), the most detailed LCI data available. Two recently published LCA studies gave insight into DAC production systems (Deutz & Bardow, 2021); (Terlouw et al., 2021). SMR was based on a source (Spath & Mann, 2001) which excluded CCS. A more recent study (Al-Qahtani et al., 2021) was thus used to model this process. Data on green NH3 production was also taken from a recent study (Boero et al., 2021). Additional data was retrieved from the ecoinvent databases (ecoinvent version 3.7, 2020). The approach provided sufficient background information to support certain modelling decisions related to the system boundary and cut-offs. Such modelling decisions were carefully documented in appendix 1 & 2 to provide maximum transparency.

4.3 LCIA of the pilot-scale LCA

4.3.1 Classification

During the classification step, all externalities caused by humans and the environment were quantified uniformly to match the output of the FU, as displayed in the inventory table. Negative values resemble flows from the environment into the system boundary, and positive values correspond to flows back into the environment. The classification was done automatically through the AB. The inventory tables are in appendix 2.

4.3.2 Characterisation results and evaluation

During the characterisation, CFs assigned the environmental flows from the inventory table to one or more ICs, where they were unified through category indicators. This step was done automatically through characterisation models. However, not all flows were assigned a CF, as some CFs were missing for specific ICs. This limitation was elaborated under section 2.2.4 for ICs related to human toxicity and eco-toxicity, yet this also applies to other ICs.

Table 3 shows the impacts of the characterisation results relative to the baseline scenario. As this phase aimed to determine the best-performing parameters, the results were displayed in this way to make them easily comparable. Only one parameter was changed at a time regarding the baseline scenario to give a transparent overview of the advantages and disadvantages. Any changes exceeding 5% were marked as significantly declining or improving compared to the baseline scenario. The characterisation results for the different technical parameter scenarios are in appendix 2.

Table 3. Relative characterisation results of the pilot-scale MP product system in relation to the baseline-scenario

Impact category/scenario	baseline	heat natural gas	heat geothermal	heat CHP	CO2 DAC	CO2 PCC	NH3 on site	SMR CCS	wind 3MW
climate change total	100.0%	85.0%	85.5%	85.4%	47.4%	47.0%	93.9%	114.4%	103.6%
freshwater and terrestrial acidification	100.0%	72.9%	73.6%	78.6%	85.7%	94.0%	101.3%	120.9%	113.9%
freshwater ecotoxicity	100.0%	95.3%	100.5%	99.6%	98.2%	91.8%	104.5%	60.8%	113.6%
freshwater eutrophication	100.0%	82.1%	83.4%	83.0%	54.1%	52.4%	99.4%	102.8%	135.5%
marine eutrophication	100.0%	85.9%	87.0%	95.5%	81.7%	96.3%	103.2%	106.1%	118.7%
terrestrial eutrophication	100.0%	91.2%	92.0%	103.7%	94.3%	114.7%	106.7%	109.2%	121.0%
carcinogenic effects	100.0%	98.9%	104.2%	100.2%	97.8%	88.9%	104.8%	39.0%	92.6%
ionising radiation	100.0%	78.6%	79.1%	79.5%	42.3%	42.6%	98.7%	119.1%	105.0%
non-carcinogenic effects	100.0%	107.1%	108.0%	127.9%	103.0%	138.3%	115.3%	107.3%	143.1%
ozone layer depletion	100.0%	72.5%	72.9%	74.1%	86.7%	88.7%	89.9%	98.9%	105.2%
photochemical ozone creation	100.0%	84.1%	85.3%	92.2%	89.0%	100.6%	101.9%	160.8%	118.3%
respiratory effects, inorganics	100.0%	73.5%	74.6%	82.4%	96.2%	109.5%	106.3%	110.8%	118.6%
dissipated water	100.0%	97.9%	107.6%	100.0%	85.7%	75.4%	72.5%	94.4%	105.8%
fossils	100.0%	77.6%	78.2%	78.2%	75.1%	74.7%	89.7%	197.9%	104.8%
land use	100.0%	119.0%	119.1%	147.6%	117.7%	168.5%	121.7%	116.8%	120.3%
minerals and metals	100.0%	99.7%	100.2%	100.0%	66.7%	62.4%	98.3%	85.7%	194.8%

Noticeable improvement

Minor change

Noticeable decline

4.4 Interpretation of the pilot-scale LCA

The model's validity was assessed during the interpretation by testing assumptions and choices made during the previous LCA stages. A consistency and a completeness check were conducted, ensuring that assumptions, data, and methods aligned with the goal and scope of the study. A CA was neglected at this stage as this was comprehensively conducted in the fourth phase. Besides CA, SA is also part of the interpretation stage, yet it is ingrained in all other LCA stages (Guinée et al., 2002). This integration is undoubtedly true for scenario-based ex-ante LCA studies, where the various scenarios represent a range of values per variable parameter, integrating SA into the LCI analysis. Therefore, no additional SA was performed at this stage, especially as this was done expansively in phase four.

4.4.1 Consistency and completeness check

The consistency and completeness checks were somehow limited at this stage because this comparative attributional LCA was not performed between two different product systems but merely between varying parameters. These checks were more appropriate in the study's fourth phase, where the product system was compared against a reference system. Regardless, both checks at this stage ensured that necessary data was accessible and had been comprehensively analysed.

During the LCIA, scenarios were analysed where the baseline scenario was compared to changing parameters. There was a slight data mismatch between the baseline scenario and the varying parameter scenarios since the former was built on established processes retrieved from the ecoinvent database (ecoinvent version 3.7, 2020). In contrast, the data for the functions of some of the different parameters were based on literature. The ecoinvent database is the most established LCA database; however, its data is often outdated (van der Giesen et al., 2020). The information for the varying parameters was mainly from LCIs found in scientific articles, mostly from 2021. It was more recent than most ecoinvent processes while simultaneously displaying more uncertainty due to the technologies' lower TRL.

Despite some ecoinvent processes being outdated, the data is well established, which presumes a certain level of completeness. Recently published literature also provides some assurance of data completeness, at least for the stage of the TRL. Given this assessment for the default scenario versus the scenarios of the varying parameters, completeness seemed more comprehensive than consistency overall. Generally, ex-ante studies question completeness over consistency, as novel technologies are usually assessed versus incumbent ones. The inconsistencies were therefore accepted for the sake of the research, and completeness was given higher priority.

4.5 Evaluation of the second phase

This phase aimed to evaluate MP's best performance at the pilot-scale while considering economic feasibility.

All three alternatives for heat performed better than supplying it through the chemical industry; however, impacts for non-carcinogenic effects and LU increased, most significantly through the supply of BCHP. Out of the three assessed alternatives, heat through natural gas performed best on average, yet no transport distance and equipment were considered.

Producing CO2 through DAC and PCC showed a clear advantage over using CO2 from the chemical industry. For the latter, there was a clear advantage of 12 ICs over the baseline scenario, with the results for climate change, ionising radiation, and freshwater eutrophication approximately 50% lower than those of the baseline scenario. Yet, there was a clear disadvantage for terrestrial eutrophication, non-carcinogenic effects, photochemical ozone creation, and LU. The latter increased by almost 70% compared to the baseline scenario.

The advantages of producing green NH3 on-site were debatable. Impacts of climate change, ozone layer depletion, water and fossil use were considerably less compared to the baseline scenario. Nevertheless, the results of terrestrial eutrophication, non-carcinogenic effects, respiratory effects, and LU increased, the latest most considerably.

1MW wind turbines outperformed 3MW wind turbines in all ICs but one, where the difference was marginal. Using 3MW turbines increased the impacts by roughly 20% on average across all ICs, except for mineral and metal use, where externalities were almost 100% higher compared to the baseline scenario, showing the sensitivity of the results. When producing H2 through SMR with CCS, instead of AEL electrolysis, results increased in 10 ICs and decreased in four. Based on these findings, it is not recommended to use SMR with CCS. However, it might still be feasible, depending on the outcomes of the comparative analysis of the ex-ante LCA.

It would have been more appropriate to perform allocation for the co-production of CO2 and products from the chemical industry. Such distribution would have assigned more burdens toward the baseline scenario besides fossil CO2 emissions, which have been included, as discussed in section 4.1.3. This approach would have more clearly shown the benefits of using biogenic CO2 from PCC through biomass combustion over fossil CO2 from the chemical industry. However, this inclusion would not have changed the comparative results. In the case of green on-site NH3 production, no clear advantage or disadvantage could be observed; regardless, as on-site production becomes more efficient, an advantage over supply through the chemical industry is expected.

5 Phase three: analysis of varying technical & surrounding parameters

5.1 Assessment of changing technical parameters

This section aimed to assess the technological parameters that influence the environmental performance when the MP product system is scaled up from pilot to manufacturing scale. The analysis is built on the literature review conducted in section 3 and the recommendations regarding the optimised product system in section 4.5. MP production is an emerging technology still under development, but so are most of the emerging technology parameters recommended in section 4.5. As these technologies mature over time, the increase in performance must thus be considered accordingly. This increase in performance is displayed through varying technical parameters related to an optimised MP product system, which will be evaluated in the following section.

Based on a comprehensive analysis, the following parameters were identified:

- Energy and production efficiency based on electricity and heat use
- Nutrient utilisation level related to TRL, MRL and MPL
- Material and resource efficiency based on economies of scale
- Type of baseload energy

The last parameter was not linked to the scale of production but rather to the overall performance of the MP product system.

5.1.1 Energy efficiency

5.1.1.1 Electrolyser efficiency

It was highlighted in section 3.5 that the efficiency of the electrolyser primarily dictates the overall performance of the MP product system. One former MP LCA has assumed efficiency levels of 53.0kWh/kg H2 (Sillman et al., 2020), which was not grounded in empirical MP production data. Others have measured efficiency of 60%, upon which a baseload scenario was built (Järviö et al., 2021). 60% efficiency is approximately equivalent to 63.1kWh/kg H2, based on a calculation in appendix 1. The authors of the latest MP LCA (Järviö et al., 2021) have assumed an efficiency of 79%, upon which an optimal scenario was setup (Järviö et al., 2021). Such an efficiency level is in line with the highest electrolyser efficiency levels, with PEM in the range of 62–82% and AEL in the field of 67-82%, measured as a higher heating value (Carmo et al., 2013).

In the current state-of-the-art electrolytic H2 production, namely PEM and AEL, the water oxidation and reduction reactions happen concurrently at two electrodes (Smolinka et al., 2015). As this occurs in the same space, it leads to problems like H2/O2 intersection (Guillet & Millet, 2015); (Millet, 2015); (Smolinka et al., 2015), obstructing the process under inconsistent supply of energy through intermittent renewable technologies (Rausch et al., 2014); (Wallace & Symes, 2018). By uncoupling the oxidation and reduction reactions, electrochemical thermally activated chemical (E-TAC) water electrolysis overcomes the challenge of H2/O2 intersection, avoiding high production efficiency losses which lead to voltage efficiencies of 98.7% or 39.9kWh/kg H2 (Dotan et al., 2019). Due to additional heat losses, the system's overall performance is expected to be 41.9kWh/kg H2. This new water-splitting method is still under development, and the above results are based on a proof-of-concept experiment (Dotan et al., 2019). Regardless, a 600MW E-TAC electrolysis facility is currently being built, expected to start green H2 production by late 2023 (H2PRO, 2022).

There are two ways of supplying H2 and O2 to the bioreactor: in-situ water electrolysis and external water electrolysis, which is the conventional way (Sillman et al., 2020). Due to a lower mass transfer of the gases to the liquid solution, the growth rate gets prohibited when using the latter method (Yu, 2014). Nevertheless, external water electrolysis can result in inferior energy use, yet safety needs to be considered as gases supplied to the bioreactor can spark explosions (Sillman et al., 2020). 54% was the highest efficiency level reported for in-situ electrolysis (Liu et al., 2016), while other values were much lower (Torella et al., 2015). It was unclear if this study's (Järviö et al., 2021) primary LCI reference data was based on external or in-situ electrolysis. Still, given the higher efficiency considered, compared to the previously reported 54%, the study's (Järviö et al., 2021) data was assumed to have been based on external H2 and O2 supply.

5.1.1.2 Green on-site NH3 production efficiency

According to an LCA study on green NH3 production through water electrolysis, changes in production efficiency levels were related to the amount of electricity needed, partially based on the size of the production facility (Boero et al., 2021). Electricity is required for the H2 production through water electrolysis, the N2 air capturing unit used to fuse the gases through the well-refined Haber-Bosch process, the facility construction phase, and the product refrigeration. For a 20.000t NH3/year production facility, 11MWh of electricity was needed to produce 1.000kg of liquid NH3; for a 100.000t NH3/year production facility, this input was 9-10MWh. An additional 60-75kWh of electricity was needed for cooling 1.000kg liquid NH3, yet this was independent of the size of the production facility (Boero et al., 2021).

5.1.1.3 DAC production efficiency

Based on an LCA study using one of Climework's industrial-scale DAC temperature–vacuum swing adsorption systems, per kg CO2 produced, 0.7kWh of electricity is needed today and 0.5kWh in the future (Deutz & Bardow, 2021). Additionally, 4.7MJ of heat are required today and 2.2MJ in future, assuming a heat pump's Coefficient of Performance of 2.51. Further 11.9MJ and 5.4MJ of heat below 100° C are needed, now and in the future. This heat is assumed emission-free, given the low temperature and vast availability. This data was representative of daily average measurements and optimisation possibilities (Deutz & Bardow, 2021).

5.1.2 Nutrient utilisation level

The nutrient utilisation level during the fermentation process is relevant for CO2, H2, O2 and NH3 and was measured at 85-90% and estimated to be 99% for optimal performance (Järviö et al., 2021). Even though it was not clear what information this assumption was based on, full nutrient utilisation is theoretically possible (Lee, 2015), as already discussed in section 3.5.

5.1.3 Material and resource efficiency

5.1.3.1 Electrolysis material and resource efficiency

Data on material and resource efficiencies for the AEL electrolysis plant was based on the amount of deionised water, the KOH, the electrolysis plant life expectancy and the electrolyser's stack life expectancy (Delpierre, 2019); (Delpierre et al., 2021). The amount of steel and nickel are directly, yet not exclusively, related to the plant and stacks' life expectancies. These metal inputs are also expected to decrease due to system optimisation.

Over twenty years, the current steel and nickel inputs for a 6MW AEL plant are 200t and 19t, respectively (Koj et al., 2017). According to their analysis of various sources, the authors estimate the steel consumption to be in the range of 10-30kg/kW until 2050 (Delpierre, 2019); (Delpierre et al., 2021). A literature review of numerous pilot-scale LCA studies on AEL plants found current nickel values in the range of 0.2-2kg/kW (Delpierre, 2019). For KOH, predictions for future use were seen in the range of 1-2g/kg H2, which were in line with current levels. Water use was found in the range of 10-19kg H2O/kg H2, with predictions of future use in the field of 9-10kg H2O/kg H2 (Delpierre, 2019). The plant's life expectancy was twenty-30 years, and the stack's lifetimes were between 80.000-130.000 hours (Delpierre et al., 2021).

E-TAC electrolysis has a simplified, membrane-free technological setup compared to electrolytic water splitting. This novel setup makes certain construction materials redundant, increasing the overall material efficiency (Dotan et al., 2019).

5.1.3.2 Green, on-site NH3 production material and resource efficiency

Changes to the green NH3 production system were related to the size of the production plant and the amount of NH3 and O2 leaking into the air (Boero et al., 2021). Per tonne of NH3 produced, 0.08kg and 0.07kg of NH3 and 1040 Nm3 and 1020 Nm3 of O2 are emitted, given a yearly production of 20.000t and 100.000t, respectively. NH3 is a relevant air pollutant (Boero et al., 2021) whose emissions need to be assessed carefully; O2, on the other hand, is not considered an environmental emission.

5.1.3.3 DAC's material and resource efficiency

Resource efficiencies for the DAC system related to the use of concrete, steel, the construction of the building hall, transportation, and land, for which both industrial and transformation of natural land were considered (Terlouw et al., 2021). The number of resources used relates to the size of production. Data was given for plants capable of capturing 4.000t

CO2/year and 100.000t CO2/year. Please refer to appendix 2 for more information, as the LCI data used for the DAC systems (Terlouw et al., 2021) was not transparently displayed.

5.1.4 Type of baseload energy

Nuclear power is a more sustainable energy source for generating MP than energy derived through wind and solar (Järviö et al., 2021). Additionally, using atomic power for MP production would make energy and gas storage solutions redundant; nevertheless, the Uranium used to fuel nuclear power is a finite resource. Besides, radioactive atomic waste poses issues with intragenerational justice. Such ethic concerns, however, are not quantifiable in an attributional LCA study. As this study focused on using sustainable and renewable energy sources for MP production, nuclear power was not further evaluated.

Geothermal energy, on the other hand, does not cause any pollution or toxic substances during operation (Duffield & Sass, 2003). Compared to intermittent energy sources, geothermal energy is not influenced by seasonal cycles and changing weather conditions (Surindra et al., 2019), making the energy and H2 supply for MP production constant. Geothermal energy is available through heat kept in rocks and water a couple of kilometres below the earth's crust (Dickson & Fanelli, 2003). This energy is endlessly available if harvested in a closed-loop pressurised steam system (Aneke et al., 2011); (Rudiyanto et al., 2017). In addition, the land requirements for geothermal power plants are far less than those of conventional power plants (Edrisi & Michaelides, 2013).

Bioenergy derived from biomass conversion is currently the most abundant type of renewable energy worldwide (Bagherian et al., 2021). It can be retrieved from products, by-products and residues (Bhavanam & Sastry, 2011) in the form of trees, energy crops, and side streams such as bark and sawdust from several industries (Malico et al., 2019). Compared to its fossil counterparts, there are several disadvantages of using biomass for bioenergy generation, related to low energy density and heating value, as well as high moisture content. To overcome these limitations while limiting transportation costs, biomass products can be pre-treated into charcoal or pellets. These products are mainly sourced from outside of Europe (Malico et al., 2019), making the overall cost and emission reduction for transportation questionable. The latter is the dominant pre-treatment method in Europe, yet these products only make up 9% of input materials used for bioenergy production (Malico et al., 2019).

Since 2010, the use of bioenergy in Europe has almost doubled (Afrouzi et al., 2021); (International Renewable Energy Agency, 2022), primarily through heating applications (Bagherian et al., 2021). This increase led to a production capacity of 9% and 16% of total electricity and heat demand (Banja & Jégard, 2017). Bioenergy is thus at the core of renewable energy supply, particularly in Europe (Bagherian et al., 2021). To further reduce fossil energy demand, CHP facilities were developed, which are more efficient than conventional power plants, with efficiencies of up to 90% (Wahlroos et al., 2014). Using CHP, 12% less biomass is needed to generate an equal sum of electricity and heat (Wahlroos et al., 2014). There is a vast growth potential for CHP applications which would reduce GHG emissions (Bagherian et al., 2021). Countries are following this trend, such as Great Britain, which set a cap for electricity-only production (Wahlroos et al., 2014). There are several thermochemical methods for CHP generation, of which combustion is the most advanced (Ahmad et al., 2016), with only 10% of bioenergy not produced through combustion (van Loo & Koppejan, 2007).

Besides bioenergy, hydropower is a highly suitable method for renewable electricity generation, covering 6.1% of primary energy demand worldwide, with a yearly expansion of 3% (Turkenburg et al., 2012). Regardless, the impacts on the environment caused by this method need to be well understood. LCA studies on hydropower plants often fall short of accounting for biogenic CH4 emissions caused by the degradation of organic matter in the water reservoirs (Hertwich, 2013). The global average emissions from hydropower are estimated to be 3g CH4/kWh (Hertwich, 2013). CH4 oxidises into CO2 in the atmosphere and thus has a 28-fold global warming potential compared to CO2 of biogenic origins (Muñoz & Schmidt, 2016). To reduce the impacts of climate change through hydropower plants, minimising the surface area is thus essential (Hertwich, 2013). In addition, hydropower plants have caused environmental degradation linked to loss of biodiversity, effects on fauna and flora, water scarcity and landscape intrusion, as summarised in the study by (Botelho et al., 2017). Thus, these effects must be considered carefully, especially as they are mostly not quantifiable through the LCA methodology.

5.2 Assessment of morphological field

5.2.1 The strategic niche management and multilevel perspective

Strategic niche management is an analytical tool to determine the progress of a novel technology embedded in an incubation system, which provides a safe space for the innovation's development (Caniëls & Romijn, 2008); (Schot & Geels, 2008). During this progression, the new technology must compete with the fully developed incumbent system, despite still being in the research and development phase (Geels & Schot, 2007). Suppose the emerging technology overcomes challenges connected to this progression and external pressures. In that case, it can eventually develop into a market niche and compete with the incumbent technology or system (Kamp & Vanheule, 2015). The following aspects were suggested to examine niche progress: explicitly framing and forming expectations, developing strong network formations, and enabling learning processes (Raven, 2005). In the following paragraph, these drivers are elaborated on in more detail.

The framing and forming of expectations directly influence the technology's development through parameter choices, the amount and type of stakeholders involved, and the available funding (Hoogma et al., 2002). Expectations can additionally influence such progression if they are spread amongst the stakeholders and provide visions while being legitimized through testing and research (Kamp & Vanheule, 2015). The strength of network formations dictates niche development by attracting attention, carrying expectations, and enabling learning (van der Laak et al., 2007). Successful technological advancement depends on solid networks built by diverse stakeholders with divergent abilities and roles (Raven, 2005).

Additionally, actors' ideas and expectations must align with the niche progress through frequent teamwork and interplay amongst the various actors (van der Laak et al., 2007). Learning can contribute to the orientation of expectations while also affecting them (Kamp & Vanheule, 2015). Intense learning depends on knowledge sharing, trust, the closeness between stakeholders, and reflection on many characteristics (van der Laak et al., 2007); (Kamp et al., 2004). Learning goes beyond reaching targets for the technology's development and its fundamental norms and principles; it also includes progress from infrastructure and manufacturing, users, and social groups, as well as governmental bodies and governing frameworks (Hoogma et al., 2002).

The multi-level perspective is a framework that adds to strategic niche management by including external factors influencing the innovation's upscaling process, often beyond the reach of the internal niche dynamics (Kamp & Vanheule, 2015). The framework analyses the socio-technical system in which the niche is embedded through landscape, regime, and niche dynamics (Geels, 2002). Regarding change, the former usually has the slowest dynamic, yet also an unpredictable and destabilising one, e.g. caused by geopolitical conflicts, variations in energy prices, and accessibility of resources (Hofstede, 2005); (Geels & Schot, 2007); (Romijn et al., 2010).

Below the landscape lies the regime level where the incumbent technologies are situated (Kamp & Vanheule, 2015). This level is commonly steady and impervious to introducing emerging technologies (Kamp & Vanheule, 2015), as the existing ones are locked in and "path-dependent" (Verbong & Geels, 2007). The more unstable a regime, the easier it is for emerging technologies to establish themselves. Instability is caused by internal conflict and landscape pressures directly influencing regime dynamics (Kamp & Vanheule, 2015), as well as pressure from the niche if it has developed enough motion (Geels & Schot, 2007). Such movement can be rooted in lower costs, backing from influential stakeholders, and enhanced achievement and functionality of the invention (Kamp & Vanheule, 2015). Given the volatile and uncertain nature surrounding niche progression, novel technologies are very receptive to landscape and regime dynamics (Geels & Schot, 2007). The interplay between the three levels dictates the success behind the scaling of the novel technology (Kamp & Vanheule, 2015). Conditional on favourable multi-level dynamics and the right moment in time, fundamental inventions have the chance to become part of the overriding regime (Geels & Schot, 2007).

5.2.2 The surrounding parameters of MP production

Regardless of reducing the dependency on fossil resources, bioenergy is a finite resource that faces numerous challenges. Indeed, most of the wood for bioenergy production in European comes from uncertified sources (Sikkema et al., 2017). If such standards were to be met in future, it might affect supply security (Malico et al., 2019). Moreover, as demand for bioenergy is projected to increase further in Europe (Wahlroos et al., 2014), this will strain resource supply in other countries (Malico et al., 2019). Due to low-level backing, several plans for biomass facilities have been stalled in favour of other energy supply sources, some of which are less sustainable (Wahlroos et al., 2014). Great Britain is Europe's largest importer of solid biomass (Malico et al., 2019).

The method of producing MP requires much energy, making the source of electricity a crucial part of determining its overall sustainability performance (Sillman et al., 2020); (Järviö et al., 2021). Regardless of the high energy demand, concepts of MP production are expanding due to the use of renewable energy technologies (Matassa et al., 2016). MP can realistically replace 13% of the protein in total livestock feed demand by 2050, yet this would also require around 10% of the collective solar and wind energy available if it was produced through water electrolysis, affirming the importance of large-scale implementation of renewable energy technologies (Pikaar et al., 2018). Additionally, given the pressure on various industries to reduce their CO2 footprint, an increase in demand for wind and solar power also leads to issues of supply for rare earth metals to produce wind turbines and solar panels, hampering the widespread employment of these renewable technologies (Smith Stegen, 2015). This competition poses the question of availability, especially for young companies with renewable energy sources at the core of their invention. Availability might particularly be problematic for wind energy, where capital costs are high while available funds are often low.

Besides supply stability and source of electricity, volumetric productivity and electricity costs have the most remarkable ability to reduce MP production prices (Nappa et al., 2020). As previously mentioned, renewable energy production costs are levelled with their fossil counterparts (Sillman et al., 2019). Given these recent price developments, which posed a challenge for renewable energy implementation for a long time, a further price reduction is assumed in future. Lower renewable electricity costs will also lessen the price of H2 production, yet they are not the sole reason for such price improvements. Due to the increased efficiency and lower economic costs compared to PEM and AEL, a lower TCH can be considered for E-TAC electrolysis (Dotan et al., 2019). H2Pro, the company currently developing this technology, claims that a novel setup reduces expenses related to assemblage, material use, and upkeep while making the need for harmful buffer solutions redundant (Dotan et al., 2019). This technological progression will thus lead to prices of 1$/kg H2, making it the cheapest source of green H2 globally (H2PRO, 2021). In comparison, the SMR and SMR with CCS prices are approximately 1.9$/kg H2 and 1.3$/kg H2, respectively (Al-Qahtani et al., 2021).

Regardless of these promising price reductions, to reach costs in the range of SBM's, the MP price under optimal conditions would need to drop a further 8-10 times (Nappa et al., 2020). One way to reduce H2 production costs while decreasing environmental emissions is to use excess O2 as a by-product (Kato et al., 2005), consequently lowering the MP price. O2 is, for example, needed for medical purposes, while electrolytic-O2 is the only process anticipated to compete with the purity levels produced by the well-established cryogenic air separation method (Squadrito et al., 2021). For the largest part, O2 is used by the steel industry, mining and metal refining, and the pulp and paper industry (Nicita et al., 2020). Regardless, green H2 supply should prioritise industries that are hard to decarbonise, such as the steel, primary chemical and NH3, maritime, and long-haul aviation industries (Agora, 2021). To widely implement green H2 production and thus make it available for novel applications, such as MP, further policy support is thus needed in the long run, as current CO2 prices are too low to incentivise large-scale renewable H2 production (Agora, 2021).

Such a price drop might thus seem out of reach for the time being, yet if MP were to supplement more expensive animal products for human consumption, this would drive down MP production costs due to economies of scale. Such a scenario might make MP feed alternatives economically competitive with the incumbents. However, MP feed alternatives can be made using a broader source of microbial origins because there are stricter regulations for human food (Ritala et al., 2017). MP as a food alternative is thus, to date, mainly produced from cheap wastes from the food and beverage industries (Ritala et al., 2017).

Furthermore, there are still numerous challenges that microbial proteins are facing in this respect, as biomass production methods need to meet all health and safety standards for human consumption (Nappa et al., 2020). For example, doubts about using cyanobacteria, known as Spirulina, as a nutrition alternative have been voiced due to its phylum’s capability to generate damaging neurotoxins (Spolaore et al., 2006); (Cox et al., 2016). Other safety challenges relate to handling and storing combustible gases at substantial manufacturing levels (Nappa et al., 2020).

To overcome the difficulties hampering the wide-scale implementation of MP, it is thus vital to create visions while aligning stakeholders’ expectations. Suppose the environmental and health benefits of using MP over conventional protein sources are clearly shown. This clarity would incentivise stakeholders to enable novel concepts such as MP production access to renewable energy sources. Showing such benefits is thus vital to stir early decision-making processes related to the innovation’s technological setup and policy intervention to avoid lock-ins later. Only through solid learning and network formations can niche and regime dynamics help MP transition from a niche development to a well-established technology. The degree of influence on policy agendas, key stakeholders, public acceptance, and public funding will play an essential role in this transition, along with access to renewable energy, green H2 and NH3.

5.2.3 PESTEL diagram

Based on the analysis of sections 3.0 and 5.2.2, the most influential quantitative parameters of the socio-technical MP system were identified through a PESTEL diagram. PESTEL stands for political, economic, sociological, technological, environmental, and legal.

Political: policy support, lobbying of established stakeholders, power of the established suppliers, subsidies for agricultural production, health targets, access to renewable energy supply

Economic: price development, consequences on employment, funding grants for R&D for biobased food products, economies of scale, health care costs

Sociological: stakeholder involvement, food consumption habits, acceptance of novel production methods, general disaffirmation of GMO food in the EU, education on environmental issues and solutions, ecological awareness, sustainable consumption habits, advertisement, ethical sourcing of materials/elements, independence of soil and weather conditions, resource efficiency, water saving, resource availability, food availability, local food production

Technological: technological development, technical efficiency, the flexibility of food production (no dependency on soil or weather conditions), comparatively easy to scale up the production system, grid stability

Environmental: access to renewable energy sources is necessary for low ecological footprint, land and water availability, and less soil strain. MP production can potentially have a positive impact on biodiversity and toxicity related to

Legal: food regulation, legislation to mitigate climate change and air pollution, EU food and safety laws

5.3 Causal loop diagram and cross-consistency analysis

A causal loop diagram (CLD) combined the most relevant qualitative and quantitative parameters (figure 8). This step made the broader influencing parameters more tangible and clearly showed their relationship to the technical parameters. Even though only the latter were quantifiable in the LCI, through the CLD, modelling choices became more transparent by relating the technical parameters to the broader context. Following the CLD, a CCA was performed to limit the number of plausible scenarios based on logical contradictions and empirical constraints (table 4). This narrowing down process was done by testing the legitimacy between circumstances through a pairwise assessment.

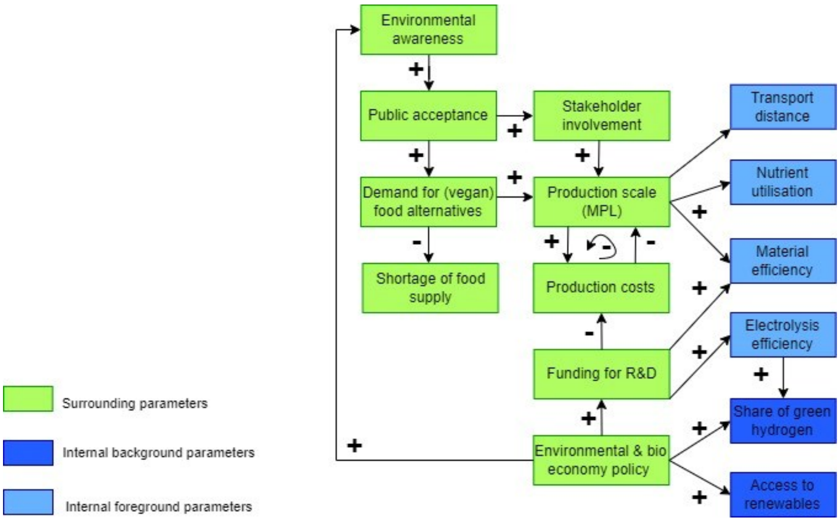


Figure 8. Causal loop diagram

5.4 Evaluation of the third phase

During the third phase, technical and surrounding parameters were defined to significantly influence the MP products system’s development. Through a CLD and a CCA, the relationship between the technical and morphological fields was made explicit. This approach set the groundwork for parameter choices for different scenarios analysed during the ex-ante LCA performed in the next phase.

Table 4. Cross-consistency analysis of technological and surrounding parameters

Fully consistent	3
Maybe consistent	2
Inconsistent	1
No correlation	0

H = high
L = low

		Demand vegan alternatives		Environmental awareness		Funding for R&D		Bioeconomy policy support		Production costs		Production scale MPL		Public acceptance		Shortage of food supply		Stakeholder involvement		Access to renewables		Electrolysis efficiency		Material efficiency		Nutrient utilisation		Use of green H2	
		H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L	H	L
Demand vegan alternatives	H																												
	L																												
Environmental awareness	H	3	1																										
	L	1	3																										
Funding for R&D	H	3	1	3	1																								
	L	1	3	1	3																								
Bioeconomy policy support	H	3	1	3	1	3	1																						
	L	1	3	1	3	1	3																						
Production costs	H	1	3	0	0	1	3	1	3																				
	L	3	1	0	0	3	1	3	1																				
Production scale MPL	H	3	1	0	0	0	0	0	0	1	3																		
	L	1	3	0	0	0	0	0	0	3	1																		
Public acceptance	H	3	1	3	1	2	1	2	1	0	0	3	1																
	L	1	3	1	3	1	2	1	2	0	0	1	3																
Shortage of food supply	H	1	3	1	3	0	0	1	3	3	1	1	1	3	1														
	L	3	1	3	1	0	0	3	1	1	3	3	3	1	3														
Stakeholder involvement	H	3	1	3	1	3	1	3	1	1	3	2	1	3	1	0	0												
	L	1	3	1	3	1	3	1	3	3	1	1	2	1	3	0	0												
Access to renewables	H	2	1	3	1	2	1	3	1	2	1	3	1	3	1	0	0	3	1										
	L	1	2	1	3	1	2	1	3	1	2	1	3	1	3	0	0	1	3										
Electrolysis efficiency	H	0	0	0	0	3	1	3	1	1	3	3	1	0	0	0	0	0	0	0	0								
	L	0	0	0	0	1	3	1	3	3	1	1	3	0	0	0	0	0	0	0	0								
Material efficiency	H	0	0	0	0	3	1	3	1	1	3	3	1	0	0	0	0	0	0	0	0	0	0						
	L	0	0	0	0	1	3	1	3	3	1	1	3	0	0	0	0	0	0	0	0	0	0						
Nutrient utilisation	H	0	0	0	0	3	1	3	1	1	3	3	1	0	0	0	0	0	0	0	0	0	0	0	0				
	L	0	0	0	0	1	3	1	3	3	1	1	3	0	0	0	0	0	0	0	0	0	0	0	0				
Use of green H2	H	0	0	0	0	3	1	3	1	3	1	3	1	3	1	0	0	2	1	3	1	3	1	3	1	0	0		
	L	0	0	0	0	1	3	1	3	1	3	1	3	1	3	0	0	1	2	1	3	1	3	1	3	0	0		

6 Phase four: scenario development and ex-ante LCA

6.1 Goal and scope definition of the ex-ante LCA

6.1.1 Goal

This study's final phase aims to build a storyline that matches the qualitative and quantitative parameters while the values for the changing technical parameters were to be defined. Based on these values, this chapter aims to determine an optimised MP product system's environmental performance when scaled from a pilot-scale to commercial production levels while using a combination of intermittent and baseload energy sources. Additionally, the purpose is to evaluate the MP product system's performance with different baseload energy sources. The central goal is to compare the different MP production scenarios against the production of SBM, in line with the main research question. Based on this analysis, the purpose is to make recommendations for further system improvements and future research aims. The final objective is to answer the last sub-research question by evaluating the extent to which the developed framework has contributed to current methods for explorative scenario approaches using the ex-ante LCA methodology for emerging technologies. The further goal definition regarding target audience, reviewing, steering committee, commissioner, and free topic choice, is in line with the pilot-scale LCA study.

6.1.2 Building of scenarios

Through the CCA, a storyline for the different scenarios started to emerge. This approach created scenarios based on the surrounding parameters, evaluated in section 5.2, that dictate the technical ones, evaluated in section 5.1. In the next step, sub-scenarios were developed, where defined values were assigned to the technical parameters. Table 5 shows these sub-scenarios related to the nutrient utilisation level, material and energy efficiencies, and type of baseload energy.

6.1.2.1 Scenario building for the morphological field

Three scenarios were created based on the analysis in sections 3.0, 5.2, and 5.3. A detailed parameter table can be found in appendix 2.

6.1.2.2 Scenario building for the technological field

Based on phase two's results and phase three's analysis, an optimal technological system was modelled considering three baseload energy sources: bioenergy using a CHP biomass combustion plant with a PCC unit, hydropower, and geothermal power. Changing technical parameters, defined in section 5.1, were given values that change over time. As in the pilot-scale scenario, wind also provided energy for other foreground processes such as fermentation and drum drying, yet for only 35% of the time due to availability. Given its large-scale implementation in Iceland, geothermal energy was not supplemented by wind-powered electricity, as this seemed redundant. Combining the different baseload energy sources and the technology's development created 12 scenarios, ensuring a broad range of results.

BCHP was chosen as the heat source for scenarios with hydropower and bioenergy as baseload solutions. Despite the outstanding performance of heat generation through geothermal power plants, it was not considered for these two scenarios, given there is no geothermal heat availability in some regions of the world. It could be debated if the heat should be supplied through BCHP or natural gas; however, given its lower impacts on climate change and fossil use, the former was chosen.

Table 5. Scenarios based on surrounding parameters

Demand for vegan alternatives	Environmental awareness	Funding for R&D	Bioeconomy policy support	Procuction costs	Production scale MPL
Vegan alternatives remain niche products	Environmental awareness is low and remains conceptual	Funding is low and insufficient	Low policy support; climate targets are not reached	Production costs are high, funding is needed as product not market competitive	MRL of 10 might not be reached; production scale remains at a low MPL below 5
Vegan alternatives penetrate the market on a large scale	Environmental awareness is supported through concrete individual choices	Funding is sufficient for technological and market development	Sufficient policy support; climate targets are likely reached	Production costs are market competitive, yet remain high; no funding needed	MPL between 5-50; product is commercially produced
Vegan alternatives become dominant over animal products	Environmental awareness is institutionalised on a broad scale	Funding is enough to support lobbying and institutionalisation	Policy support beyond climate targets, tackling food security & resource use globally	Production costs outperform animal alternatives	MPL of 50-100; fulll economies of scale are reached

Public acceptance	Shortage of food supply	Stakeholder involvement	Access to renewables	Use of green hydrogen
There is low public acceptance and animal products remain the first choice	Remains high due to inefficient resource use	Remains low with dispersed interest, low network formation & learning	Little access as renewable energy production remains centralised	Low due to high cost related to H2 production; grey & blue H2 remain dominant
MP is publicly accepted; MP is at par with animal products	Effects of more efficient resources use start emerging on global level	Network formation starts to emerge, interests start to align	Renewable energy becomes more de-centralised and accessible	Green H2 is largely implemented, yet still needs subsidies to remain competitive
MP have become part of normality and are no longer extraordinary	Production in scarce regions leads to more equal distribution	Solid network formations change institutional structures	Renewable energy is fully accessible	Green H2 is fully implemented due to low production price

🟦 **Worst case / pilot-scale scenario; 2020-2030**

🟨 **Medium case / small commercial production; 2030-2040**

🟩 **Best case / high commercial production; 2040-2050**

SMR with CCS was not further considered for MP production, as the advantages and disadvantages over water electrolysis were already displayed in section 4.3.2. AEL water electrolysis was thus chosen as the H2 source for all scenarios in this phase. The AEL facility's material inputs mainly depend on the plant and stack life expectancies. These material inputs, therefore, changed as life expectancy is anticipated to increase over time. Values for the changing parameters in 2020 were mainly based on the LCI of (Koj et al., 2017). Between 2030 and 2040, the plant´s life expectancy is 20 years, with a stack lifetime of 10 years. In 2040 the plant´s life expectancy is 25 years with a stack lifetime of 12.5 years, while in 2050, these values increase to 30 and 15 years, respectively. Appendix 2 shows a detailed list of the material inputs related to the plant's and stacks' life expectancies.

Concerning the deionised water used for electrical water splitting, the values were in the range of the future predictions of 9-10kg H2O/kg H2 (Delpierre, 2019). Based on recent literature (Delpierre et al., 2021) and calculations found in appendix 2, the steel consumption of the AEL plan equated to a range of 55.5-177t per AEL plant, respectively. Using the same source (Delpierre et al., 2021) and calculations, nickel inputs were between 1.1t and 11.8t per AEL plant, below those reported by (Koj et al., 2017). As no future predictions for nickel use were found, the values were based on the variations of the current range of 0.2-2kg/kW. The measures used to determine the steel and nickel inputs were based on an hourly production rate of 118kg H2/h and considered the efficiency of the electrolysis as well as the plant and cell stacks' lifetime. As the predicted values for KOH matched the current values, this range of 1-2g/kg H2 was used as the technology progressed. Electrolysis efficiency levels ranged from 63.1kWh/kg H2 to 41.9kWh/kg H2. Despite E-TAC's higher material efficiency, this reduction was not considered due to a lack of data.

Green on-site NH3 production was chosen for all scenarios, given its better performance than supplying it externally from fossil sources. Three production sizes were considered for these facilities: 20.000t NH3/year, 60.000t NH3/year, and 100.000t NH3/year. Changes in electricity inputs and NH3 emissions were considered accordingly. The nutrient utilisation level was modelled between 85-99% efficiency, progressively increasing over the 30 years. For the gas inputs, these values were 1.8-2.0kg CO2, 2.8-3.2kg H2, and 2.4-2.7kg NH3, per kg MP produced. For reasons discussed under section 4.2.3, O2 was left out of the analysis.

CO2 was supplied from point sources using a PCC unit attached to a BCHP plant for the scenarios using bioenergy and hydropower as baseload energy sources. As no data was available for pellets, wood chips were used instead as input into the CHP process. The process encompassed the facility, the biomass feedstock, the emissions to air, and the discarding of the ashes (Treyer, 2014). Additionally, materials for operation included: lubricating oil, organic chemicals, sodium chloride, chlorine and decarbonized water (Treyer, 2014). DAC was used in the place of point sources in the case of geothermal power. For the DAC system, a 4.000t CO2/year capturing unit was modelled until 2040, which was assumed to increase its capacity to 100.000t CO2/year afterwards. Electricity, material efficiencies, and heat inputs were changed in line with this increase in production capacity.

6.1.3 Scope

The general scope is the same as for the attributional LCA study of the pilot-scale MP system; the analysis is from cradle to gate using a "0/0" modelling approach for biogenic carbon. As in phase two, the scope of this assessment is a detailed LCA study following the ISO 14040 and 14044. Still, compared to the previously conducted LCA, there are differences regarding the temporal, technological and geographical scope discussed in the following sections.

6.1.3.1 Temporal scope

The temporal scope resembles a progression from 2020 until 2050 in 10-year intervals. This progression aims to show the potential development of the production system over time. This progression could also be seen as a bad, a middle, and a worst-case scenario, as highlighted under section 2.2.4.

6.1.3.2 Technological scope

In 2020, the year the LCI data (Järviö et al., 2021) was documented, the MP product system resembles a pilot-scale facility with TRL 6-7. In 2030 the TRL was assumed to reach a value of 9, while a MRL of 10 might also be achieved. In 2040 total manufacturing capacity is accomplished with an MPL of up to 50. In 2050 a MPL above 50 resembles MP as new, incumbent technology. E-TAC water electrolysis momentarily has a moderate TRL, presumably reaching total manufacturing capacity by 2050. In the MP scenarios for 2050, it was thus considered instead of AEL. This technological progression is in line with an appropriate time frame discussed in section 3.5.

6.1.3.3 Geographical scope

Three different locations were considered in this study. Geothermal energy was assumed to be retrieved from Iceland, as 65% of primary energy in Iceland is produced in this way (Government of Iceland, 2016). The scenario using hydropower as an energy baseload was set in Finland, given its high share in the primary energy production mix. Even though variations in this energy source can be expected due to weather conditions, in 2016, an electricity share through hydropower of 18.4% was reported (Statistics Finland, 2017). For the biomass combustion plant, England was chosen as the geographical location for reasons highlighted under section 4.1.3.3.

6.1.4 Function, FU, and reference flow

The function and FU were the same as in section 4.1.4. The reference flows are the following:

- The first reference flow is the production of 1kg MP, with 65% protein and 5% moisture, before packaging, using wind power and biomass combustion as energy sources. From now on, this reference flow will be referred to as MP Bio for simplicity.
- The second reference flow is the production of 1kg MP, with 65% protein and 5% moisture content, before packaging, using wind power and hydropower as energy sources. From now on, this reference flow will be referred to as MP Hydro for simplicity.
- The third reference flow is the production of 1kg MP, with 65% protein and 5% moisture content, before packaging, using geothermal energy. From now on, this reference flow will be referred to as MP Geo for simplicity.
- The fourth reference flow is the production of 1.3kg SBM, before packaging, produced in Brazil with 45-55% protein content. From now on, this reference flow will be referred to as SBM BR for simplicity.
- The fifth reference flow is the production of 1.3kg SBM, before packaging, produced in the rest of the world (except Brazil) with 45-55% protein content. From now on, this reference flow will be referred to as SBM RoW for simplicity.

The protein content of SBM lies between 44-56% of the dry mass (DM); (Banaszkiewicz, 2011). The crude fat content lies between 0.55-3.3% of the DM (Banaszkiewicz, 2011). In comparison, the MP analysed in this study has a protein content of 65% and a fat content of 6% (Järviö et al., 2021). It was, therefore, necessary to increase the amount of SBM to match the comparison to MP. 1.3 seemed like an appropriate scaling factor based on comparing the two alternatives' protein and fat content. It was assumed that the moisture content of SBM matched that of MP. Regardless, a comparison based on the amount of protein can be misleading as different protein sources vary in their nutrition content (Järviö et al., 2021), posing a specific limitation to this study.

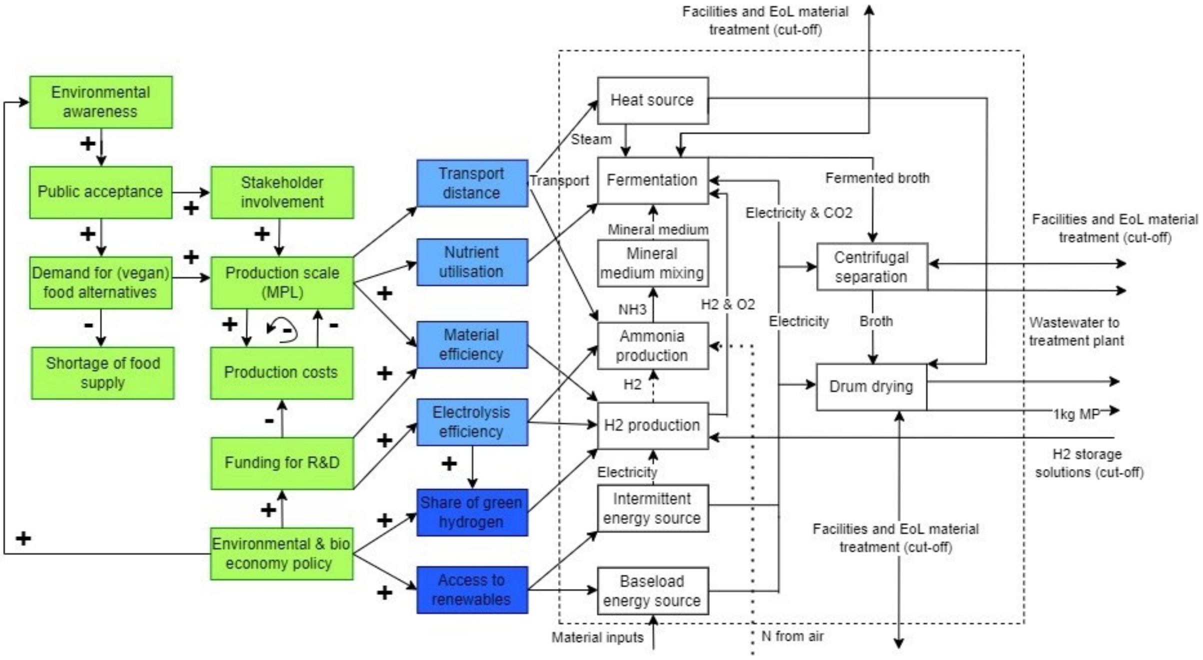
Fishmeal showed better results for climate change, ozone layer depletion, energy demand, freshwater use, and LU than MP; only for acidification, MP had slightly better results (Järviö et al., 2021). Regardless, for reasons of biodiversity loss and natural resource depletion, fishmeal is generally considered unsustainable, despite its advantages over MP in most ICs. It was therefore not considered further in this study.

6.2 LCI of the ex-ante LCA

There were no changes between the economic and environmental system boundary, and allocation choices, between phase two and phase four. Biogenic CH4 emissions were considered in the analysis for the hydro reservoir, which has often been neglected in LCA studies, as discussed in section 5.1.4. Regarding the cut-offs, some were added compared to section 4.2.3. As no LCI data was available regarding material inputs for E-TAC water electrolysis, these materials were based on predictions for the facility in 2050. Additionally, no transport or storage was included for CO2 from the PCC unit to the hydropower plant.

6.2.1 Flow chart

The CLD developed in section 5.3 was integrated into the MP product system flowchart, displayed in figure 9 below.



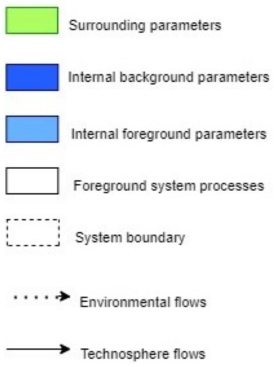


Figure 9. CLD and flowchart showing the relationship between surrounding parameters and the MP product system

6.2.2 Data sourcing

Besides modelling systems related to baseload energy, CO2 capturing, green NH3 and H2 production, the MP production data was taken from the only available empirical source (Järviö et al., 2021). Material and energy efficiencies related to the water-splitting process were retrieved from two LCA studies on AEL electrolysis (Koj et al., 2017); (Delpierre et al., 2021). A recently published paper on E-TAC electrolysis gave insight into the expected efficiency (Dotan et al., 2019). A techno-economic MP analysis was used for production price indications and other qualitative parameters, thus the morphological MP field (Nappa et al., 2020). The environmental and economic costs of H2 production were retrieved from a recent publication that compares various grey, blue and green H2 production methods (Al-Qahtani et al., 2021). The same sources were used in phase four as in phase two regarding future predictions of material and energy efficiencies of DAC and green NH3 production systems. SBM reference flows were modelled using the LCA database. Data on alternative methanotrophic microbial protein sources were taken from recent literature.

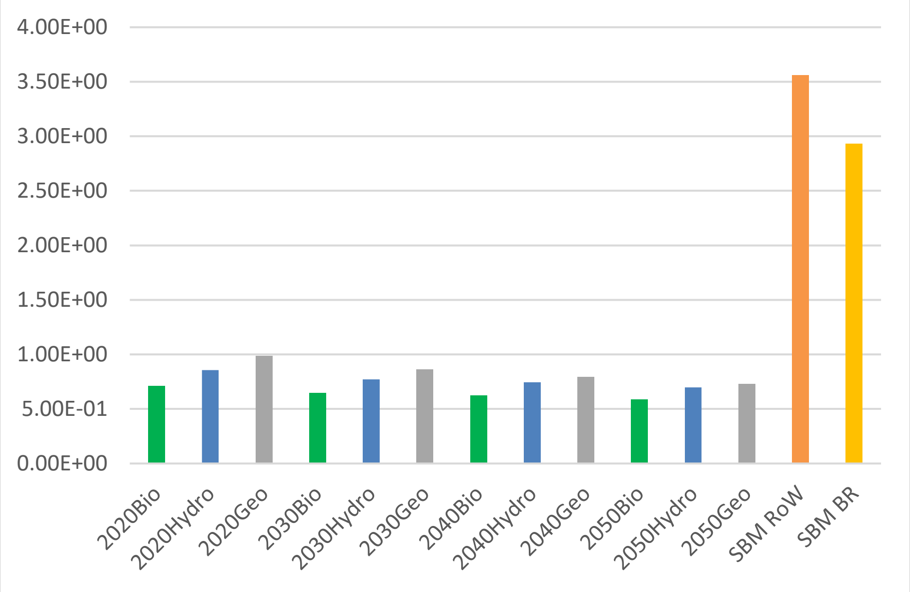
6.3 LCIA of the ex-ante LCA

For more information on ICs and classification, please refer to section 4.3. Limitations to the impact assessment stage for ex-ante LCA studies were discussed in sections 2.1.4 and 2.2.4. The characterisation results presented in this section were thus assessed carefully with these limitations in mind.

6.3.1 Characterisation results & evaluation

The characterisation results for the different scenarios per FU are shown in figure 10, with each graph displaying one IC. A table with the numeric results is provided in the appendix 2 for more detailed information.

For non-carcinogenic effects, impacts of SBM production are displayed as negative numbers. These negative numbers are an indication of a lack of missing CFs. The results in this IC were thus non-comparable to MP production and were consequently left out of the comparative assessment. Therefore, a comparison between the incumbent and MP reference flows was drawn between 15 ICs.

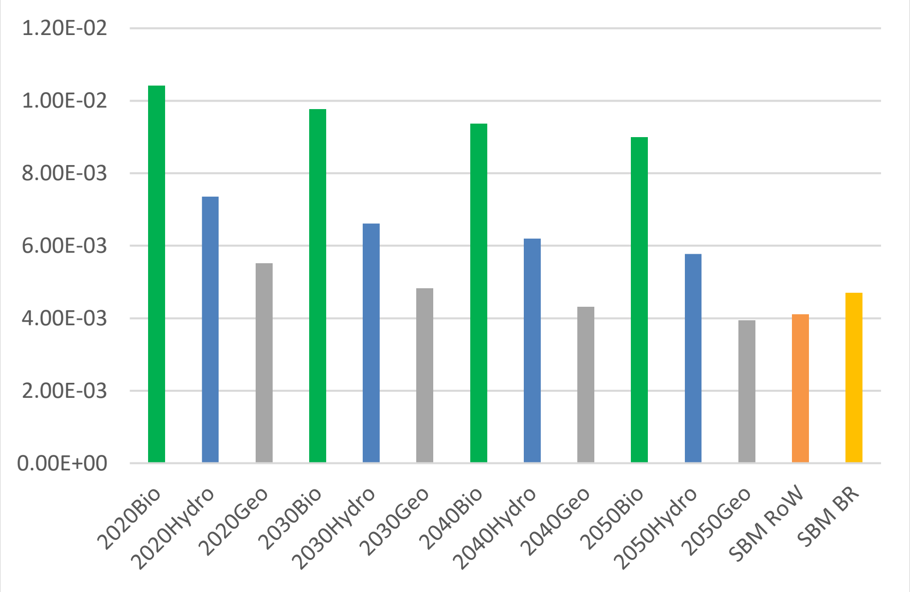


ILCD 2.0 2018 midpoint, climate change, kg CO2-eq.

Throughout all 12 scenarios, the impacts of climate change were below 1kg CO2-eq per kg MP produced. 2020 Geo showed the worst performance of the three MP scenarios, yet those results were at least 200% below those of the two SBM reference flows. All three MP product systems improved their performance over 30 years by about 0.25kg CO2-eq.

Bio had the lowest impacts in total; in the “best-case” in 2050, these impacts were nearing 0.5 CO2-eq per kg MP produced.

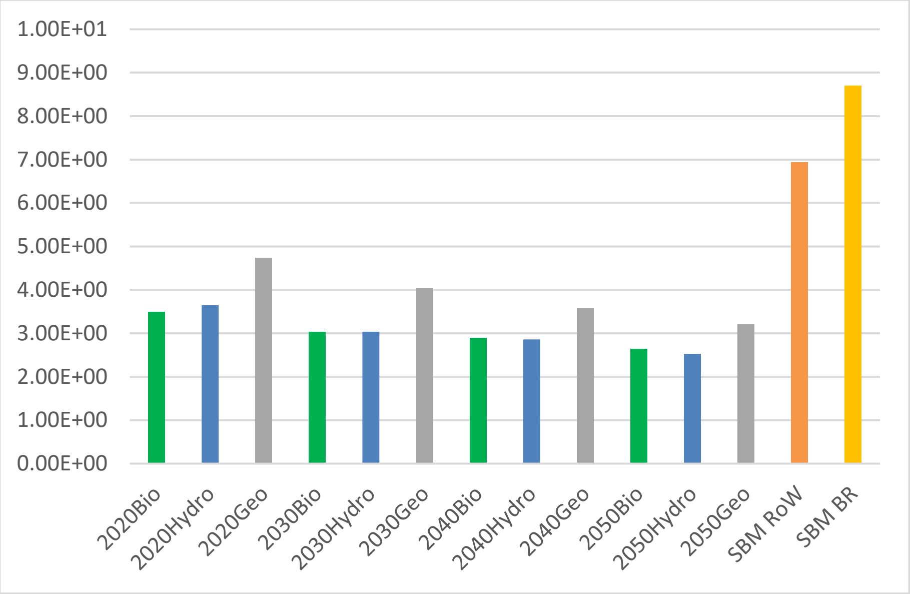
Even though Geo performed worst out of the three MP production systems, these results aligned with hydro in 2050. This alignment resulted from Geo’s more substantial decrease in impacts over time.



ILCD 2.0 2018 midpoint, ecosystem quality, freshwater and terrestrial acidification, mol H+ eq.

Impacts for the 2020 scenarios showed an increase of approximately 50-180% compared to the impacts of the two incumbent systems. Neither Bio nor Hydro could compete with SBM, despite a steady decrease over time. Geo started competing with the incumbent system in 2040. In 2050 Geo outperforms SBM RoW; however, this advantage was only 5% approximately. The impacts of the best-case scenarios for Bio and Hydro in 2050 still exceeded SBM by about 30-70%.

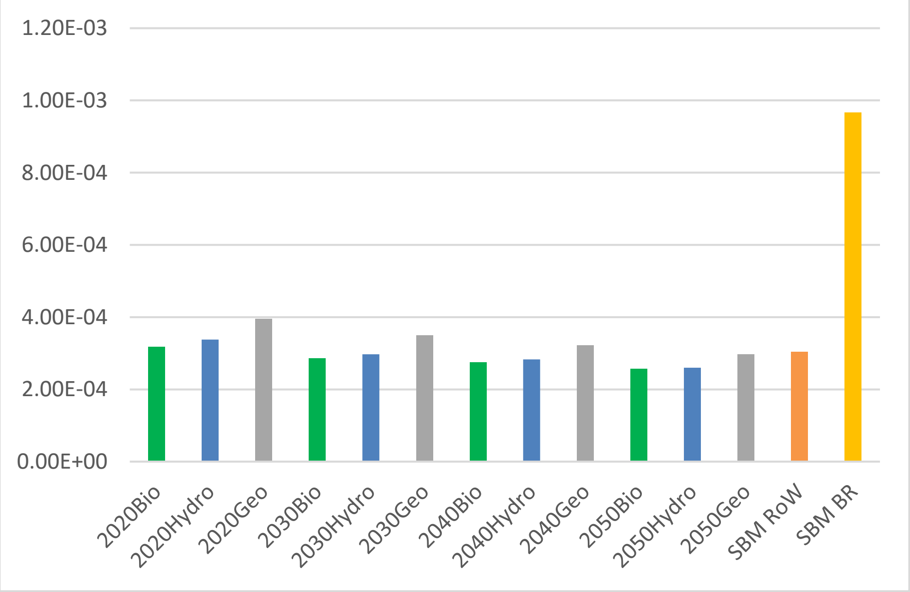
Over time, the decrease in emissions was roughly 30% for Bio and Geo and 25% for Hydro.



ILCD 2.0 2018 midpoint, ecosystem quality, freshwater ecotoxicity, CTU

The impacts for all MP scenarios were lower than those of the incumbent systems. 2020 Geo, the worst-performing MP scenario, showed a reduction of over 40% compared to the incumbents. In 2050, this reduction was over 100%.

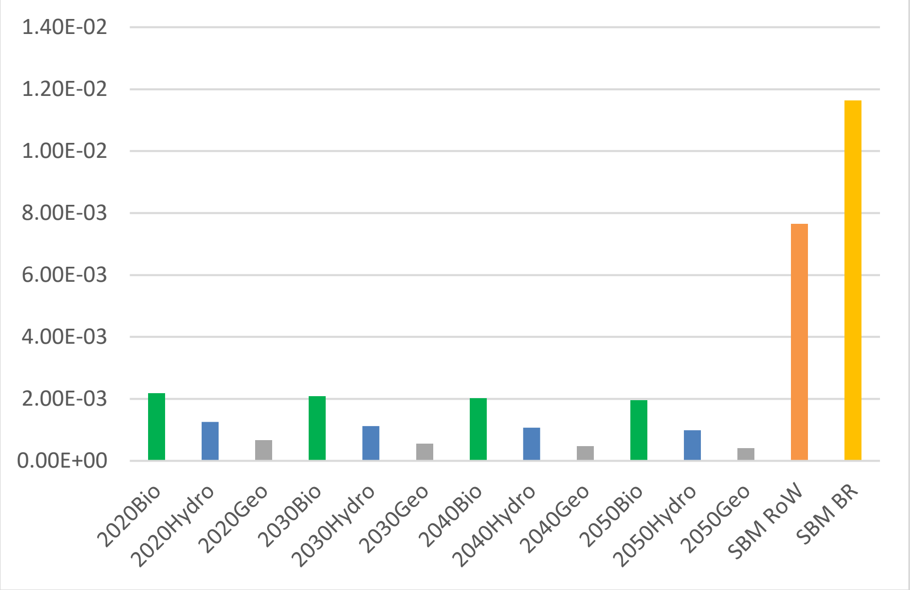
Over the 30 years, the impacts of the three MP product systems decreased by 25-50%.



ILCD 2.0 2018 midpoint, ecosystem quality, freshwater eutrophication, kg P-eq.

The impacts for all MP product systems outperformed SBM BR. Compared to SBM RoW, the impacts of the MP production scenarios were not competitive until 2030, when Bio started to display advantages. Throughout the 30 years, Bio performed best out of the three MP product systems, while Geo performed worst.

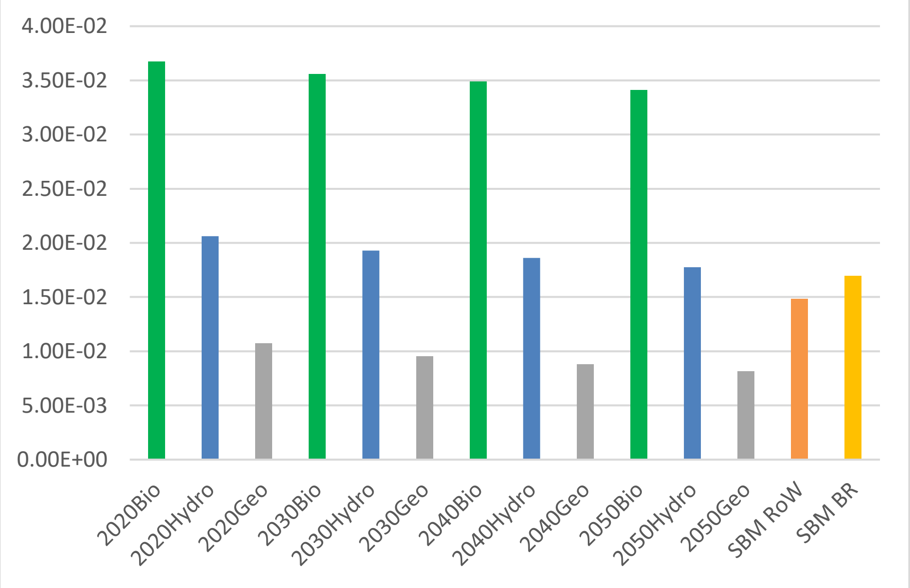
The impacts of MP production in 2020 were approximately 25% and 50% above SBM RoW for Hydro and Geo, respectively. In 2050 Hydro displayed an advantage of about 10%, while Geo still lagged by 5%.



ILCD 2.0 2018 midpoint, ecosystem quality, marine eutrophication, kg N-eq.

There was a clear advantage of MP production over SBM production. In 2020 Bio, the worst-performing MP scenario outperformed SBM RoW by 350%. In 2050 this relation was almost twice as much, as impacts were practically halved over the 30 years.

The other two MP product systems displayed a similar reduction over time. Hydro and Geo generally outperformed Bio by approximately 50% and 400%, respectively.

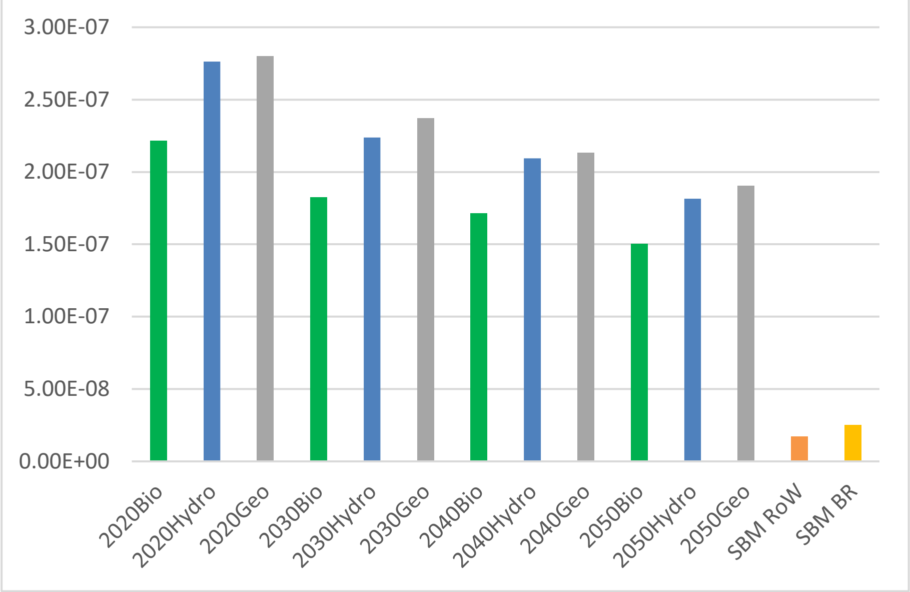


ILCD 2.0 2018 midpoint, ecosystem quality, terrestrial eutrophication, mol N-eq.

Geo was the only MP production system that outperformed SBM. This advantage was about 50% in 2020 and 100% in 2050.

In 2050, the impacts of Hydro were 10% above those of SBM RoW and level with SBM BR. In 2020, the difference to the former was approximately 30% and 15% to the latter.

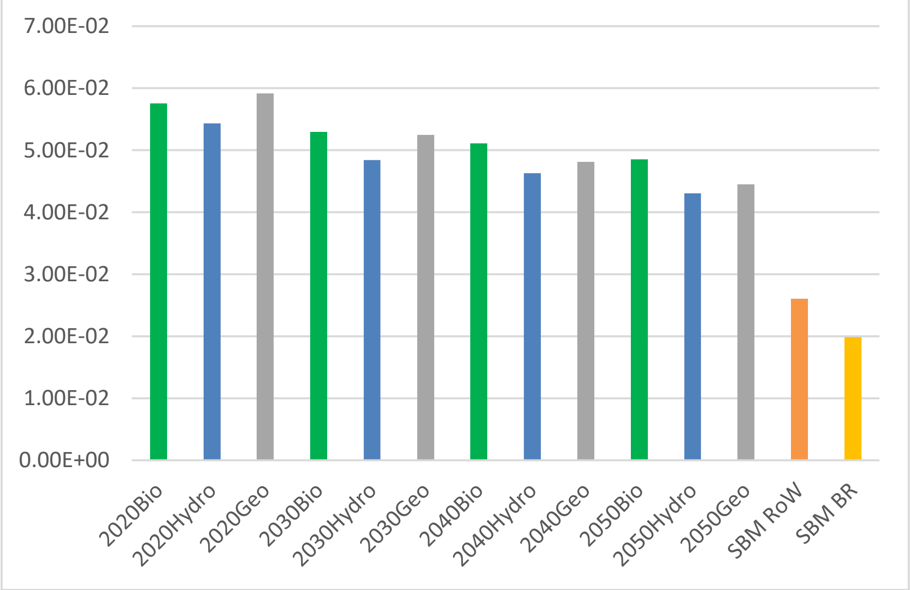
MP Bio performed worse than SBM RoW by almost 150% in 2020 and 70% in 2050.



ILCD 2.0 2018 midpoint, human health, carcinogenic effects, CTUh

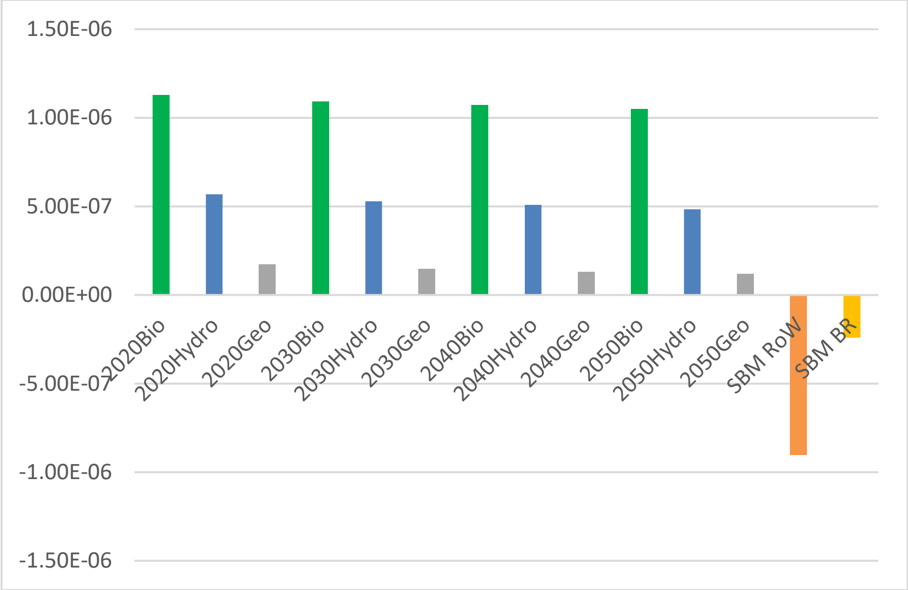
The impacts of the MP product systems were higher than SBM by several orders of magnitude.

The impacts for all three MP product systems decreased by 20-40% over time. Nonetheless, this reduction was insignificant compared to the impacts of both SBM BR and SBM RoW.



ILCD 2.0 2018 midpoint, human health, ionising radiation, kg U235-eq.

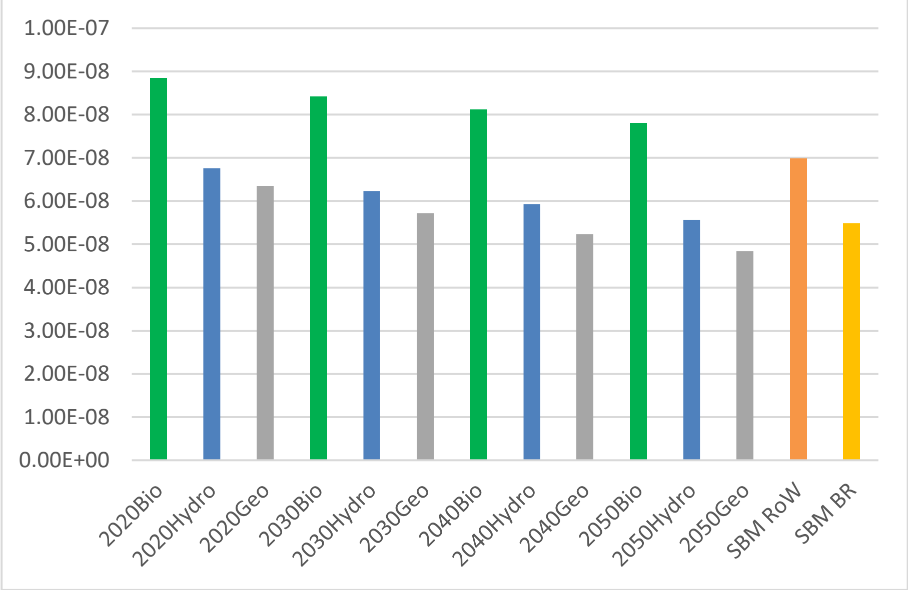
The impacts of all three MP systems could not compete with those of the incumbent systems. In the worst-case scenario, the difference between SBM RoW and 2020 Geo was almost 150%. In the best case, 2050 Hydro, this difference was around 60%.



ILCD 2.0 2018 midpoint, human health, non-carcinogenic effects, CTUh

The impacts of Geo were the lowest out of the three MP production systems. In 2020, these impacts were almost six times lower than Bio's and nearly three times lower than Hydro's.

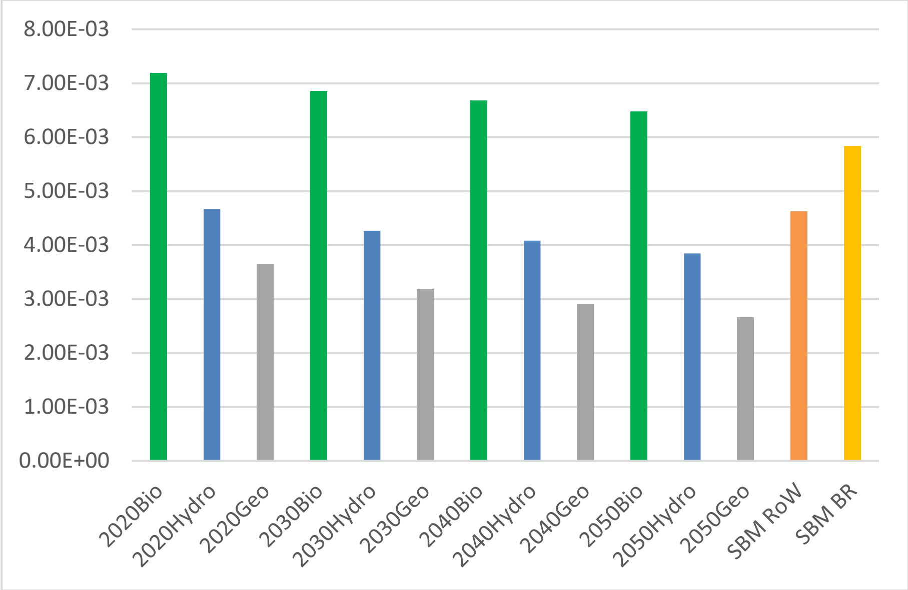
Over time, there was an insignificant reduction for all three MP product systems.



ILCD 2.0 2018 midpoint, human health, ozone layer depletion, kg CFC-11-eq.

MP Bio could not compete with SBM in any case.

The other two MP product systems outperformed SBM RoW in 2020. Additionally, both performed better than SBM BR, Geo from 2040 and Hydro from 2050.

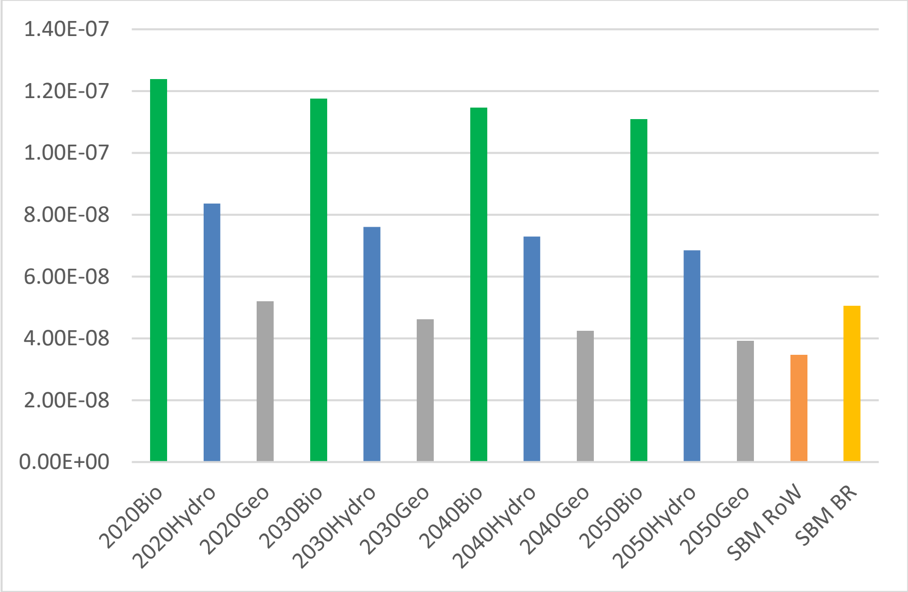


ILCD 2.0 2018 midpoint, human health, photochemical ozone creation, kg NMVOC-eq.

Geo performed best out of the three MP product systems and outperformed SBM even at the pilot-scale. Impacts for Geo declined by roughly 25% over time.

Hydro also outperformed both comparative reference flows at the pilot-scale. In 2050, with a fully developed MPL, the impacts of Hydro were between 20-50% lower than those of SBM.

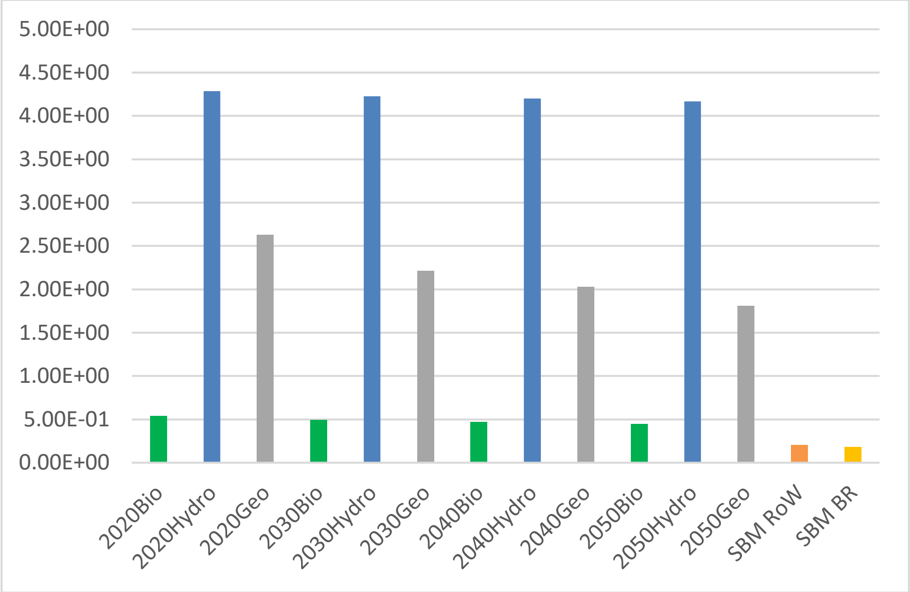
Bio was uncompetitive with the incumbent systems, even at fully developed MPL.



ILCD 2.0 2018 midpoint, human health, respiratory effects, inorganics, disease incidences

None of the MP product systems were competitive with SBM RoW, even though Geo came close in 2050. Only Geo outperformed SBM BR, yet only from 2030.

The impacts of the worst-performing scenario, Bio 2020, were over five times those of SBM RoW. For Hydro, the disadvantage was between 35-100%, compared to the incumbent systems.

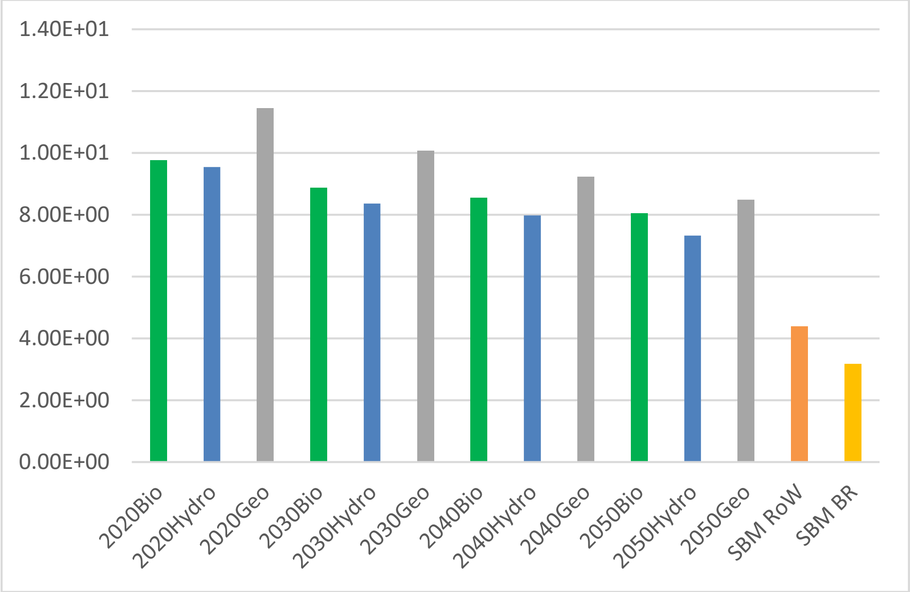


ILCD 2.0 2018 midpoint, resources, dissipated water, m3 water

None of the three MP production systems could compete with SBM production. For Hydro and Geo, the results were several orders of magnitude larger than those of the incumbent product systems.

Outcomes of Hydro barely reduced over time; those of Geo declined by about 25% over 30 years. In 2050 impacts of Geo were less than half those of Hydro.

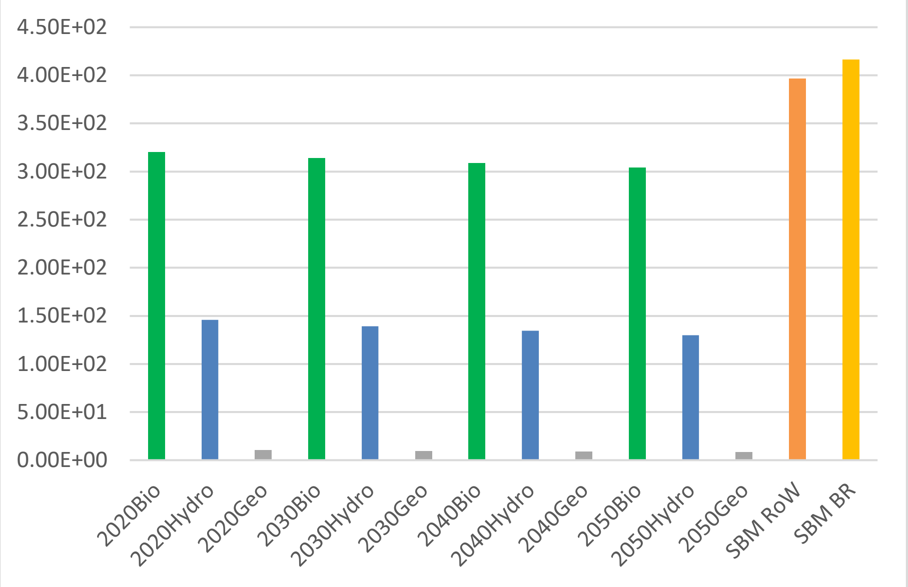
Bio’s results were in the same magnitude as the incumbent product systems. Yet even at fully developed MPL, the impacts were still over 100% larger than those resulting from the production of SBM RoW.



ILCD 2.0 2018 midpoint, resources, fossils, megajoule

None of the MP product systems could compete with SBM, even at high MPL. In 2050, impacts were between 80% and 100% higher than SBM RoW, with Bio showing the least and Geo showing the most.

The outcomes of Bio and Hydro were very similar over time. In 2020 results of Geo were roughly 20% higher than those of the other two MP reference flows. In 2050 these differences decreased.



ILCD 2.0 2018 midpoint, resources, land use, points

Out of all assessed ICs, LU showed the most noticeable difference between the three MP product systems. The results of all three MP reference flows were competitive with SBM even at the pilot-scale. The outcomes of Hydro were over three times less than those of the incumbents. The effects of Geo were many orders of magnitude below those of the incumbents. Bio's performance was most closely aligned with that of the incumbents; the difference between the various scenarios was between 10-20%

All three MP product systems showed marginal change over time.

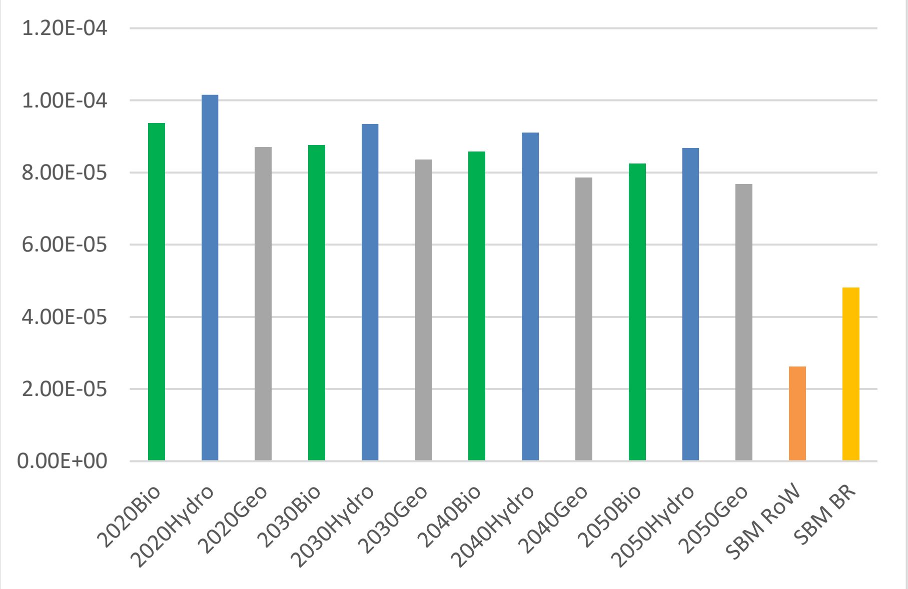


Figure 10. Characterisation results of the ex-ante LCA

ILCD 2.0 2018 midpoint, resources, minerals and metals, kg Sb-eq.

The results were similar to fossil resources; none of the MP product systems could compete with the incumbents. At high MPL, impacts for the MP reference flows were between 70-300% higher compared to SBM.

Between 2020-2050 impacts of MP reference flows declined by 15-25%.

Summary of the characterisation results for Geo

Geo’s overall performance at TRL 6-8 outperformed the two incumbent systems in six of 15 assessed ICs: climate change, ecosystem quality, marine eutrophication, terrestrial eutrophication, photochemical ozone creation, and LU. For freshwater eutrophication and ozone layer depletion, Geo 2020 performed better than one of the two compared product systems but worse than the other. There was a total disadvantage compared to SBM for seven ICs in 2020: freshwater and terrestrial acidification, carcinogenic effects, ionising radiation, respiratory effects, dissipated water, fossil resources, and minerals and metals.

At MPL 50-100, an advantage over SBM also occurred for freshwater and terrestrial acidification and ozone layer depletion, thus in eight ICs. For the latter, this advantage started to appear in 2040. The advantage of the incumbent systems remained in five ICs at high MPL.

Summary of the characterisation results for Hydro

Compared to SBM, there were advantages in five ICs in 2020: climate change, freshwater ecotoxicity, marine eutrophication, photochemical ozone creation, and LU. For freshwater eutrophication and ozone layer depletion, overall benefits emerged in 2040 and 2050, respectively. There was a general disadvantage, even at high MPL, compared to SBM for eight ICs: freshwater and terrestrial acidification, terrestrial eutrophication, carcinogenic effects, ionising radiation, respiratory effects, dissipated water, fossil resources, and minerals and metals.

Summary of the characterisation results for Bio

At TRL 6-8 and MPL 50-100, there was an absolute advantage to SBM in four assessed ICs: climate change, freshwater ecotoxicity, marine eutrophication, and LU; for freshwater eutrophication, this was also true from 2030. There was a total disadvantage compared to SBM for 10 ICs: freshwater and terrestrial acidification, terrestrial eutrophication, carcinogenic effects, ionising radiation, ozone layer depletion, photochemical ozone creation, respiratory effects, dissipated water, fossil resources, and minerals and metals.

Summary of the overall characterisation results

Rooted in the above analysis of the characterisation results, Geo performed best overall, Hydro performed second best, and Bio performed last out of the three assessed MP product systems. Based on the number of favourable ICs, only Geo was advantageous over the incumbents, yet only at high MPL; neither Hydro nor Bio could compete with SBM, even in the best-case scenario.

6.3.2 Normalisation results and evaluation

6.3.2.1 Context of normalisation

As comparative characterisation results were closely aligned, especially for Hydro, closer evaluation was necessary. Normalisation is used to help decision-making during the LCIA stage by presenting the magnitude of the characterisation results in relation to a reference value through a NF (Crenna et al., 2019). NFs are usually made-up of area-specific or global inventories of environmental and economic flows combined with absent elementary streams, e.g. using estimations for emissions that influence specific ICs (Cucurachi et al., 2014). NFs are combined with an impact assessment method and the associated models (Sala et al., 2017).

As mentioned in section 2.4.4, there are limitations regarding the USEtox model related to toxicity ICs. Currently, the model only portrays 3000 substances (Hou et al., 2020). The authors (Hou et al., 2020) compare the Toxic Substance Control Act by the U.S. Environmental Protection Agency (EPA), which includes over 85,000 substances. Furthermore, the USEtox model misses 19% and 67% of CFs for ecotoxicity and human toxicity, respectively, on top of the already limited depiction (Hou et al., 2020). In addition to these limitations, there are estimations of over 100.000 new organic substances that enter the global market every year, for which impacts are primarily unknown. Even though this number could not be independently verified, these numbers indicate how little is known of the effects on human health and the environment caused by new chemical flows.

These unknown CFs generally lead to high relevance for toxicity-related ICs compared to the normalised results of other ICs. Some normalisation results are consequently more reliable for some ICs than others. The reliability of the NFs is summarised in table 6 below. In the table, NFs are judged by the completeness of the inventory data and the inventory robustness. The latter is dictated by the quality of the data and the robustness of the impact assessment method based on model quality (Sala et al., 2017) and recommendations from the ILCD (Hauschild et al., 2011). This assessment thus gives insight into the possible source of uncertainty in calculating the NFs. The recommendation level, also displayed in table 6 for the EF impact assessment, summarises inventory completeness and robustness. The recommendation levels range from I, meaning the model is suitable and advocated, to III, meaning the model is still approved and yet to be applied carefully.

6.3.2.2 Prioritisation of normalised results

Assessing the overall advantages and disadvantages of the MP and SBM reference flows was difficult based on the characterisation results. Thus, priority had to be given to some ICs. Prioritisation was based on the significance of the characterisation results, their recommendation level, and the planetary boundaries. In the following paragraphs, the results were evaluated based on such prioritisation.

The normalised results for 2020 and 2050 of 14 ICs were displayed in graphs (figure 11 & figure 12). Impacts of non-carcinogenic effects were left out of the normalisation results, as they showed negative figures for reasons discussed in section 6.3.1. Missing CFs for environmental flows of chemicals generally lead to low world reference values compared to other ICs. Therefore, toxicity-related ICs have low recommendation levels, between II and III. Normalisation results for carcinogenic effects and freshwater ecotoxicity were thus the highest. SBM displayed high relevance while not matching the lower results of the MP reference flows for freshwater ecotoxicity. The opposite was true for carcinogenic effects. Thus, the comparative advantages and disadvantages between the incumbent and the emerging technology were somewhat balanced for these two assessed toxicity-related ICs. Given this balance, the high inaccuracy and uncertainty, toxicity-related ICs were not further evaluated or considered for the overall comparison. Furthermore, MP’s impacts for carcinogenic effects were several orders of magnitude above those of all other ICs. Thus, they were not displayed in the graph, as it would have been hard to read the other results.

The results for freshwater eutrophication in the case of SBM BR also showed exceptionally high relevance, which could be a sign of missing CFs. Nonetheless, this could not be verified. Consequently, these impacts were still considered. Based on the highest values of the normalisation results, besides toxicity-related ICs, the main priority was given to climate change, eutrophication-related ICs, water use, LU, and mineral and metal use. The latter and freshwater eutrophication had the highest contribution for all five reference flows. The first four ICs have a medium to high recommendation level, while the latter have a low one.

Despite their high recommendation levels, ionising radiation and ozone layer depletion results showed little relevance for both the emerging and incumbent technologies. The potential advantages and disadvantages of the two product systems were thus barely considered. The significance of the normalisation results for all other ICs, namely freshwater and terrestrial acidification, photochemical ozone creation, respiratory effects, and fossil use, was in the mid-range; at the same time, their recommendation levels vary. These four ICs were thus given equal importance.

Therefore, priority was thus given to ICs with high relevance and high recommendation levels. This prioritisation of ICs is also, for the focal part, in line with the planetary boundaries. Based on figure 6, the limits for climate change, biochemical flows, i.e. eutrophication, and land-system change, have already been exceeded. Freshwater use is within the safety boundary, yet it was still prioritised given the relatively high normalisation results for Hydro and Geo. Based on the planetary boundaries, the impacts of freshwater and terrestrial acidification and ozone layer depletion are within the safety zone. At the same time, those of atmospheric aerosol loading, i.e. particulate matter, have not yet been quantified to assess their planetary limit. Assigning little relevance to ozone layer depletion while placing medium importance on the other two ICs thus seemed reasonable.

6.3.2.3 Summary of the normalisation results for 2020

Geo 2020

For climate change and LU, Geo outperformed SBM. Geo also displayed a slight advantage to SBM for the three eutrophication ICs. However, SBM showed better results for mineral, metal, and water use than Geo. For the ICs displaying mid-range importance, SBM performed better overall, displaying an advantage in three out of four ICs.

Hydro 2020

As for Geo, Hydro outperformed SBM for climate change and LU. For water use and mineral and metal use, SBM surpassed Hydro. In the ICs related to eutrophication, comparative results levelled each other out. For the ICs displaying mid-range importance, SBM performed better overall, as is the case for Geo.

Bio 2020

A clear advantage over ICs with high significance was observed for climate change and LU. For eutrophication ICs the comparative difference is marginal. For freshwater, mineral and metal use, and all four ICs displaying mid-range importance, SBM beat Bio 2020.

Overall evaluation of normalisation results for 2020

Based on the above-elaborated prioritisation, Geo was somewhat level with SBM even at TRL 6-8. However, the high relevance of minerals and metals makes this claim debatable. Despite an optimal system setup, neither of the other two MP reference flows was competitive with SBM in 2020. These findings thus change the characterisation results for Geo 2020.

6.3.2.4 Summary of the normalisation results for 2050

Geo 2050

As the MPL increases, the advantage over SBM in the three eutrophication ICs becomes more apparent, while the disadvantage remained for water, mineral, and metal use compared to 2020. For the ICs displaying mid-range significance, SBM had a slight overall advantage.

Hydro 2050

At high MPL, Hydro displayed a clear total advantage in the three eutrophication ICs. As for Geo, the substantial disadvantage for water, mineral, and metal use remained. The worse overall performance compared to SBM for ICs displaying mid-range importance was also still apparent.

Bio 2050

As Bio outperformed SBM for eutrophication overall, there was a slight advantage compared to SBM for ICs with high significance, even though the benefit for LU was marginal compared to the other two MP reference flows. Besides ionising radiation and ozone layer depletion, the normalised results for water use showed the most negligible significance for Bio; yet the result was still approximately double that of SBM. SBM's noticeable advantage remained for minerals and metals and the four ICs displaying mid-range importance.

Overall evaluation of normalisation results for 2050

For the ICs of the highest significance, all three MP reference flows outperformed SBM overall. Regardless, the normalised results for minerals and metals remained exceptionally high for MP production compared to SBM, as did the water use for Geo and Hydro; however, the opposite was true for LU, somehow levelling out the results in these two ICs. Despite MP’s advantage in ICs of high importance, only Geo outperformed SBM when considering ICs of medium relevance. For Hydro and Geo, no clear overall advantage or disadvantage could be observed at high MPL.

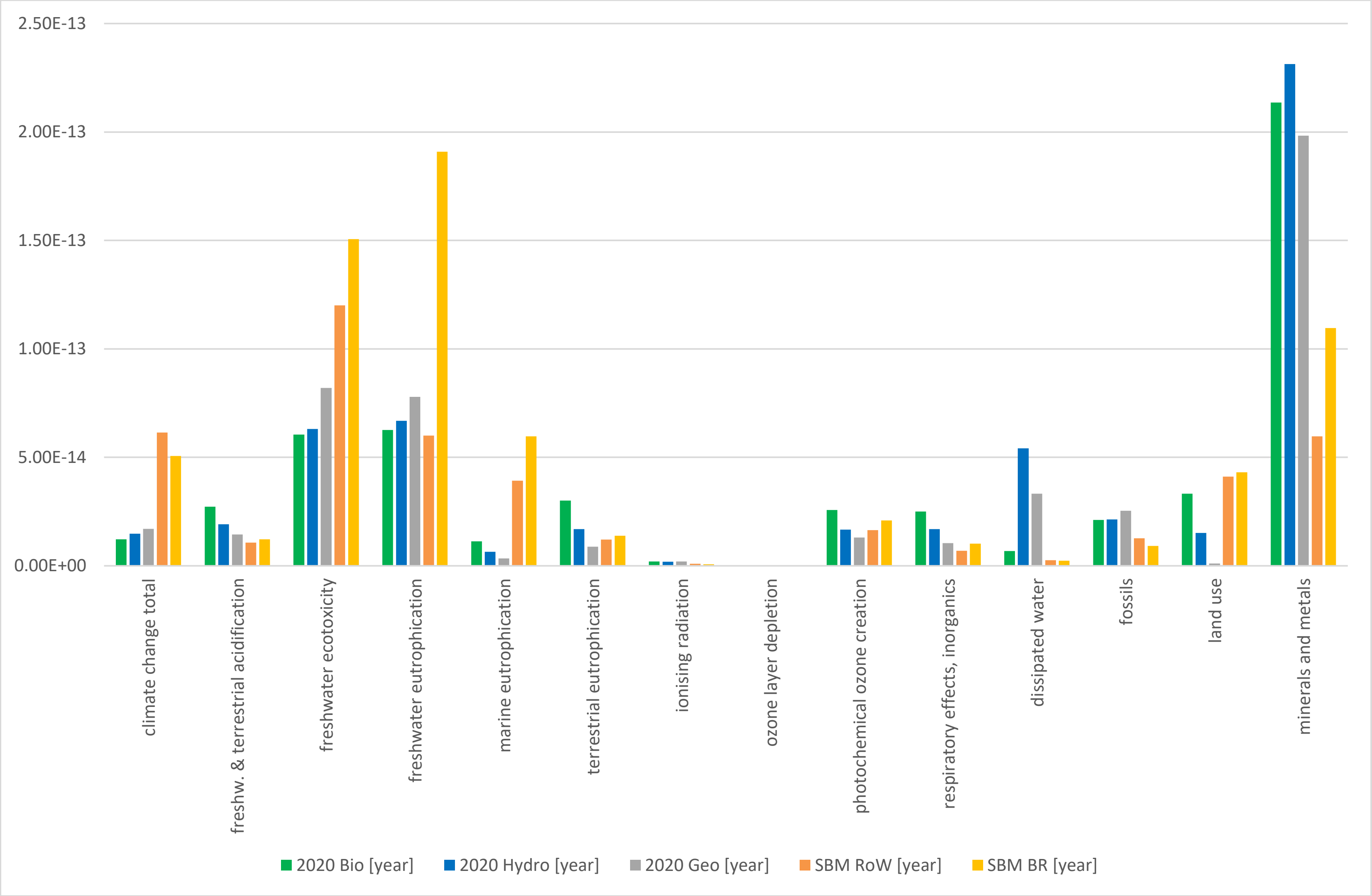


Figure 11. Normalisation results for MP production in 2020 and current SBM production

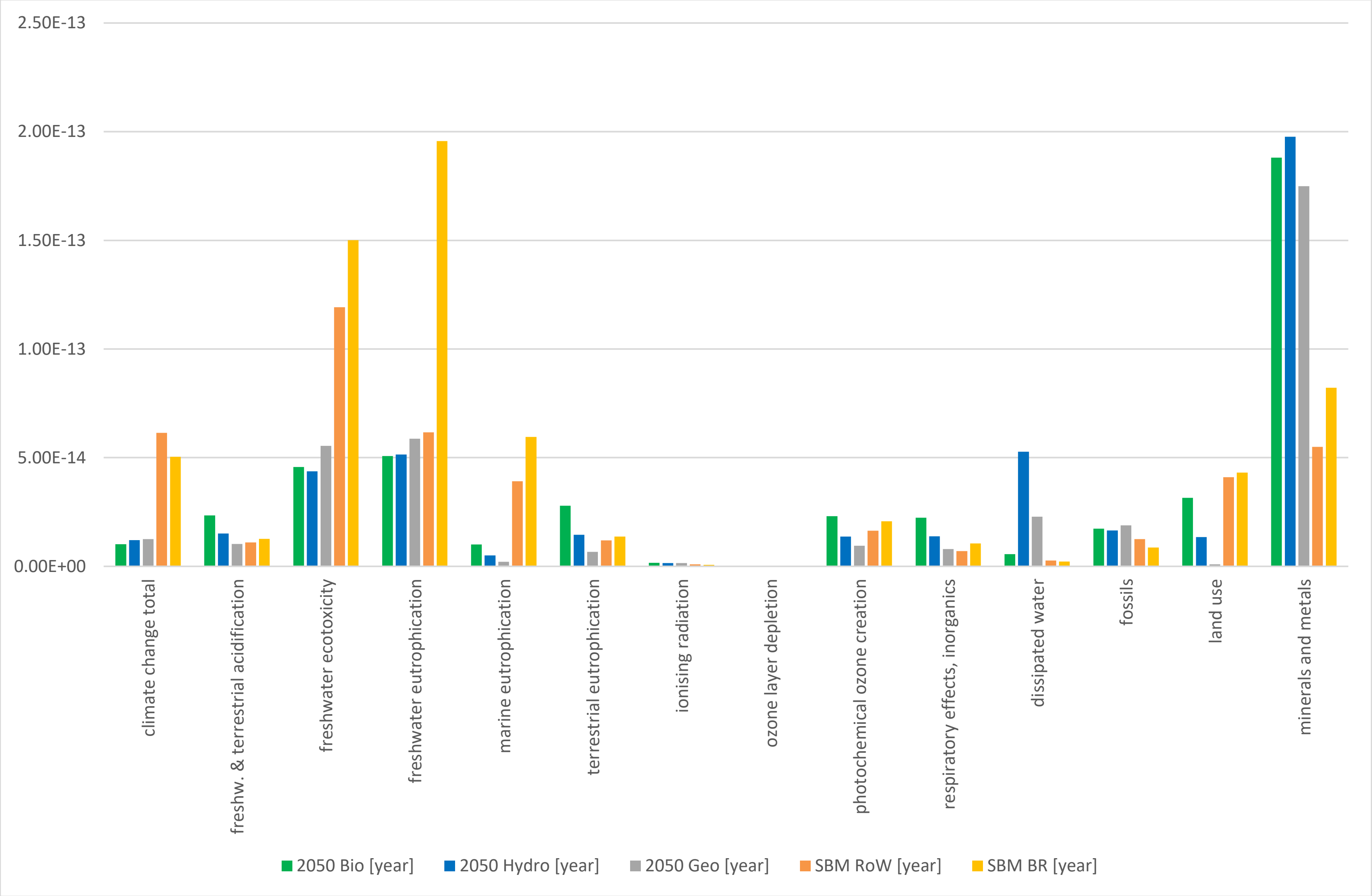


Figure 12. Normalisation results for MP production in 2050 and current SBM production

Table 6. Normalisation factors and recommendation levels (Sala et al., 2017)

Impact category	Model	Unit	global NF for EF	Inventory coverage completeness	Inventory robustness	Recommendation level of EF impact assessment
Climate change	IPCC (2013)	kg CO2-eq	5.8E+13	II	I	I
Ozone depletion	WMO (2014)	kg CFC-11-eq	3.3E+08	III	II	I
Human toxicity, cancer	USEtox Saouter et al. (2018)	CTUh	3.4E+05	III	III	II/III
Human toxicity, non-cancer	USEtox Saouter et al. (2018)	CTUh	5.9E+06	III	III	II/III
Particulate matter	Fantke et al., 2016	disease incidences	4.95E+06 (a)	I/II	I/II	I
Ionising radiation	Frischknecht et al., 2000	kBq U-235-eq.	2.9E+13	II	III	II
Photochemical ozone formation	Van Zelm et al., 2008 as applied in ReCiPe (2008)	kg NMVOC eq.	2.8E+11	III	I/II	II
Acidification	Posch et al., 2008	mol H^+-eq	3.8E+11	II	I/II	II
Eutrophication, terrestrial	Posch et al., 2008	mol N-eq	1.2E+12	II	I/II	II
Eutrophication, freshwater	Struijs et al., 2009	kg P-eq	5.1E+09	II	III	II
Eutrophication, marine	Struijs et al., 2009	kg N-eq	2.0E+11	II	II/III	II
Land use	Bos et al., 2016 (based on)	pt	9.64E+15 (b)	II	II	III
Ecotoxicity freshwater	USEtox Saouter et al. (2018)	CTUe	5.8E+13	III	III	II/III
Water use	AWARE 100 (based on Frischknecht & Jolliet, 2016)	m^3 water eq of deprived water	7.91E+13 (b)	I	II	III
Resource use, fossils	ADP fossils (van Oers et al., 2002)	MJ	4.5E+14	I	II	III
Resource use, minerals and metals	ADP ultimate reserve (van Oers et al., 2002)	kg Sb-eq	4.4E+08	I	II	III

6.4 Interpretation of the ex-ante LCA

6.4.1 Consistency and completeness check

Table 7. Consistency check

Check	MP product systems	SBM product systems	Compare APS to OFPS	Action
Data source	Literature/ ecoinvent 3.7	ecoinvent 3.7	Not consistent	No action
Data accuracy	Some uncertainty	Good	Not consistent	No action
Data age	1990 - 2021	2011 - 2021	Consistent	No action
Technology coverage	Experimental data	Existing, large-scale technology	Not consistent	No action
Time-related coverage	Novel technology	Conventional technology	Not consistent	No action
Geographical coverage	Europe member country where possible, else global	Brazil and the rest of the world	Not consistent	No action
Multifunctionality	Yes	No	Not consistent	SA
Cut-off	Yes	No	Not consistent	No action

Table 7 was created to state inconsistencies between MP and SBM production. There were some general inconsistencies between the LCI of the MP product system and the SBM model. The former was based on pilot-scale data, while the latter was based on a well-established industrial-scale production system. To limit the impacts these inconsistencies might have on the accuracy of the results, recent data sources were used for MP production, keeping grey literature to a minimum. In fact, for all data used for modelling purposes, only published scientific literature was used. Despite efforts, companies in the field of MP production were unwilling to share their data for this publication.

Multifunctionality was inconsistent between the MP and SBM models, as it only applied to the former. The only multifunctional process in the study was heat and power co-generation, which also produces CO2. Allocation was based on the CO2 price, which is very volatile and highly impacted the results. Therefore, the SA performed in phase two was carefully considered in the analysis of phase four to contain the fluctuations in the results, thus limiting the variations between the two models.

Despite these inconsistencies, the two LCIs were considered as complete as possible. However, there were some cut-offs for the emerging technology, so the MP model was not as comprehensive as the incumbent system. Regardless, these cut-offs were considered a minor lack of completeness. For the sake of generating useable results, completeness was therefore given priority over consistency in this study.

6.4.2 Contribution analysis

6.4.2.1 Contribution analysis for the MP system

Based on the characterisation results, it was essential to reflect on possible solutions for specific ICs to evaluate if the MP production system could be improved further. In this section, a CA was performed for the three MP product systems in 2050. This year was chosen as the reference, displaying the most optimised scenarios. Based upon the analysis of the optimised design, other hotspots were identified, showing the most substantial room for improvement.

In the first step of the CA, the product contributions were traced back to the product's origin, neglecting other hotspots along the supply chain. In the tables below, the most prominent contributors labelled "rest" sum up all contributions below 5%. The second-largest contributor was electricity which was not evaluated further as inputs came from various unit processes, each contributing below 5%. A more thorough analysis was conducted in the CA’s second step based on the AB’s Sankey function. These Sankey diagrams can be found in the appendix 2.

Wastewater treatment had a positive contribution for some ICs, which is displayed with a negative connotation. These negative values indicate circumvented emissions by lessening pollutants and nutrient concentrations through the treatment facility, which would otherwise end up in the environment (Szulc et al., 2021).

ILCD 2.0 2018 midpoint, climate change

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	5.09E-01	4.47E-01	3.89E-01
electricity, high voltage	3.09E-01	4.40E-01	3.40E-01
nylon 6-6, glass-filled	5.31E-02	5.57E-02	
heat, district or industrial, other than natural gas	6.66E-02	5.73E-02	
pig iron			1.19E-01
clinker			9.93E-02
hard coal			5.26E-02
sodium phosphate	6.22E-02		

Bio: H2 production made up 30.4% caused by windmill production, with 15.0% related to steel production and 6.3% to glass-fibre reinforced plastic. The nutrient solution contributed 39.1%; of these impacts, potassium carbonate and sodium phosphate contributed 15.3% and 18.2%, respectively. 6.2% were related to the production of sodium phosphate itself, while a further 9,0% were indirectly linked through the production of phosphoric acid. 17.2% came from wood chip production, of which transport made up 7.2%. Wood chips were the main contributor to CO2 and electricity production for all foreground processes, except for H2 production. The former contributed 14.9% and the latter 12.1%. 5.2% of impacts caused by CO2 were related to the carbon capture process.

Hydro: Like Bio, H2 production made up 30.7% caused by windmill production; more precisely, 15.7% related to steel and 6.6% to glass-fibre reinforced plastic. Electricity production for all other foreground processes made up 24.5%, mainly caused by non-alpine hydro reservoirs. The nutrient solution accounted for 30.9%; 11.9% and 14.2% were related to potassium carbonate and sodium phosphate. 7.0% were indirectly linked to the latter through the production of phosphoric acid. CO2 accounted for 10.6%, with wood chips contributing 8.1%.

Geo: 61.3% of impacts came from deep-well drilling for geothermal power related to electricity production. Of these impacts, 42.3% were related to H2 production and 15.3% to all other foreground processes. When tracing these impacts further upstream, 35.2%, 10.9%, and 9.8%, 2.4% related to steel production, the Icelandic electricity mix, Portland cement, and potassium carbonate, respectively. The nutrient solution contributed 31.2%; potassium carbonate and sodium phosphate comprised 11.7% and 13.7%, respectively.

ILCD 2.0 2018 midpoint, ecosystem quality, freshwater and terrestrial acidification

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	2.46E-01	2.98E-01	4.43E-01
electricity, high voltage	5.59E-01	4.57E-01	2.29E-01
nickel, 99.5%			
heat, district or industrial, other than natural gas	6.26E-02	7.04E-02	6.85E-02
copper	5.48E-02	8.14E-02	
sulfuric acid	7.82E-02	9.32E-02	1.44E-01
transport, freight, sea, bulk carrier for dry goods			6.05E-02
sinter, iron			5.47E-02

Bio: 53.7% of impacts came from heat and power co-generation; 31.8% were associated with CO2, and approximately 20% with electricity production. 26.3% related to the nutrient solution; of these impacts, 6% came from potassium carbonate and 16.5% from sodium phosphate, with 13.8% related to phosphoric acid. 15% were related to H2 production, which can be traced back to windmill production.

Hydro: Of the total impacts, 31.7% were linked to the nutrient solution, 24.8% to the production of CO2, and 24.1% to the production of H2. The latter was related to windmill production. The impacts of CO2 come from the heat and power co-generation process. For the nutrient solution, 7.2% relate to potassium carbonate and 19.7% to sodium phosphate, of which the most considerable part is linked to phosphoric acid.

Geo: The nutrient solution made up 48.5%; 11.1% was caused by potassium carbonate and 30.3% by sodium phosphate, with phosphoric acid contributing 25.4%. H2 made up 33% of electricity generation; many of these impacts were linked to steel needed for deep-well drilling. All other foreground processes cause 10.7% of inputs through electricity generation.

ILCD 2.0 2018 midpoint, ecosystem quality, freshwater ecotoxicity

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00

Bio: 26% were related to CO2 and 19.3% to electricity generation. These impacts were the main contributors to heat and power co-generation, contributing 44%; 12% related to the actual process, 22% to wood ash treatment through landfarming, and 9.8% to wood chips, mainly concerning transport. 31.9% related to H2

Rest	2.56E-01	4.45E-01	3.11E-01
wood ash mixture, pure	2.18E-01	1.37E-01	
ferrochromium, high-carbon, 68% Cr	1.15E-01	1.60E-01	
electricity, high voltage	1.23E-01	8.00E-02	
slag, unalloyed electric arc furnace steel	1.04E-01	1.42E-01	1.03E-01
brake wear emissions, lorry	8.18E-02		
H3PO4 purification residue	1.02E-01	1.06E-01	8.01E-02
drilling waste			5.06E-01
residue from shredder fraction from manual dismantling			
basic oxygen furnace waste			
wastewater, average		-7.08E-02	

production, related primarily to steel, with a contribution of 24.1%. 24.4% were related to the nutrient solution, with 10.2% from landfilling phosphoric acid treatment residue.

Hydro: H2 production contributed 44.5%, mainly related to steel for windmills with 35%. CO2 contributed 24.8%; of these impacts, 13.7% related to landfarming wood ash, and 7.0% to the heat and power co-production. 26.3% came from the nutrient solution, with 17.6% associated with sodium phosphate, of which 10.6% came from landfilling phosphoric acid treatment residue. Wastewater treatment had a positive contribution of 7.1%.

Geo: 82% related to deep-well drilling for geothermal power used for electricity production. 48.7% of this contribution was connected to landfarming drilling waste. 21.2% were related to the nutrient solution; 13.3% were tied to sodium phosphate, of which 8% were linked to landfilling phosphoric acid treatment residue.

ILCD 2.0 2018 midpoint, ecosystem quality, freshwater eutrophication

product	**2050Bio**	**2050Hydro**	**2050Geo**
Total	1.00E+00	1.00E+00	1.00E+00
Rest	4.64E-02	3.61E-02	9.88E-03
sulfidic tailings from copper mine operation	2.48E-01	2.80E-01	1.34E-01
spoil from hard coal mining	2.42E-01	2.50E-01	3.48E-01
spoil from lignite mining	2.33E-01	2.27E-01	2.37E-01
H3PO4 purification residue	1.53E-01	1.37E-01	1.24E-01
Concentrated broth	7.76E-02	6.98E-02	6.31E-02
basic oxygen furnace waste			8.35E-02

Bio: 44.5% related to the nutrient solution; 25.7% to sodium phosphate and 12.8% to potassium carbonate. Of the former, 15.3% were connected to landfilling phosphoric acid treatment residue. 39.5% related to H2 production caused by windmill production; of these impacts, 13.0% were linked to steel and 13.3% to copper.

Hydro: 45.6% was connected to H2 production linked to windmills; copper and steel were the main contributors to these impacts with 16.1% and 15.8%. The nutrient solution was the second largest contributor with 40.9%; 11.5% stemmed from potassium carbonate and 23.2% from sodium phosphate. 14% of the latter stream was caused by landfilling phosphoric acid purification residue.

Geo: Electricity production through geothermal power made up 52.3%, almost exclusively related to deep-well drilling, of which steel production accounted for 38.0%. Further upstream, 39.5% of impacts associated with geothermal power were caused by H2 production. The nutrient solution caused 36.8%, with 21.0% linked to sodium phosphate; this stream was mainly related to phosphoric acid production, with 12.4% stemming from landfilling phosphoric acid treatment residue.

ILCD 2.0 2018 midpoint, ecosystem quality, marine eutrophication

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	1.97E-01	7.24E-01	1.87E+00
electricity, high voltage	8.03E-01	7.46E-01	4.48E-01
wastewater, average		-4.70E-01	-1.32E+00

Bio: 51.8% came from CO2 production and 32.1% from electricity production for all foreground processes except H2 generation. Downstream these two flows contributed 72.0% towards the heat and power co-generation process and 12.7% towards wood chips. The positive impact of wastewater treatment was 34.5%. The nutrient solution and H2 production contributed 16.6% and 15.0%.

Hydro: 62.0% came from the heat and power co-generation process, almost exclusively related to CO2 production. 27.9% related to the production of H2 stemming from windmill production. The nutrient solution pertained 24.1%, 10.4% from sodium phosphate and 8.6% from potassium carbonate. The positive impact of wastewater treatment was 49.1%.

Geo: Wastewater treatment had a positive impact of 142.5%. Deep-well drilling for geothermal power, related to electricity generation, held 120%, with 78.5% related to steel production. 84.9% of impacts caused by electricity generation were connected to H2 generation and 29.1% to all other foreground processes. The nutrient solution added 70.1%, generated through sodium phosphate and potassium carbonate with 30.0% and 24.7%. Phosphoric acid production contributed 22.7% to the former.

ILCD 2.0 2018 midpoint, ecosystem quality, terrestrial eutrophication

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	2.61E-01	3.45E-01	4.22E-01
electricity, high voltage	7.39E-01	6.55E-01	2.20E-01
blasting			1.05E-01
transport, freight, sea, bulk carrier for dry goods			8.41E-02
clinker			5.76E-02
transport, freight train			5.77E-02
diesel burned in building machine			5.39E-02

Bio: 80.3% related to wood combustion, with 69.0% linked to the heat and power co-generation process; upstream, these impacts were caused by CO2 and electricity use for all foreground processes except H2 production, with 45.2% and 27.4% contributions. 10.9% and 7.9% were related to the nutrient solution and the carbon capture infrastructure.

Hydro: heat and power co-generation caused 58.0% of impacts, almost exclusively related to CO2 production. H2 generation contributed 14.3% linked to windmill production. The nutrient solution was 15.5%, with 7.1% related to sodium phosphate production.

Geo: 55.6% of impacts were linked to deep-well drilling for geothermal power, with 35.5% associated with steel production. Upstream, 39.8% and 13.7% of these impacts were caused by electricity generation for H2 production and the other foreground processes. The nutrient solution was 37.6%, with sodium phosphate and potassium carbonate making up 17.1% and 11.4%. 10.6% of the former was related to phosphoric acid production and 5.6% to soda ash production.

ILCD 2.0 2018 midpoint, human health, carcinogenic effects

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	1.40E-01	1.18E-01	7.06E-02
ferrochromium, high-carbon, 68% Cr	2.78E-01	3.12E-01	9.98E-02

Bio: 52.9% of impacts stemmed from H2 production, almost exclusively related to steel production for windmills. 30.6% related to the nutrient solution, with 24.0% arising from sodium phosphate production; this share was almost solely associated with landfilling phosphoric acid purification residue.

Hydro: Like Bio, 59.6% were related to H2 production, stemming from steel production for windmills. 26.8% came from the nutrient

slag, unalloyed electric arc furnace steel	2.58E-01	2.82E-01	2.54E-01
H3PO4 purification residue	2.09E-01	1.75E-01	1.65E-01
wood ash mixture, pure	6.15E-02		
basic oxygen furnace waste	5.31E-02	5.79E-02	1.31E-01
sludge from steel rolling		5.60E-02	5.62E-02
drilling waste			2.24E-01

solution, with 17.0% associated with landfilling phosphoric acid purification residue.

Geo: 46.9% and 17.3% of impacts related to H2 production and electricity use for all foreground processes except for H2 generation. These two streams were the main contributors to geothermal power production, which caused 73.3%; 19.0% of this amount came from landfilling drilling waste and 49.3% from steel production. The nutrient solution comprised 24.5% of impacts, with 16.0% connected to landfilling phosphoric acid purification residue.

ILCD 2.0 2018 midpoint, human health, ionising radiation

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	4.09E-02	4.08E-02	3.98E-02
tailing from uranium milling	5.07E-01	5.29E-01	5.58E-01
low-level radioactive waste	2.32E-01	1.99E-01	1.56E-01
electricity, high voltage	1.16E-01	1.21E-01	1.30E-01
spent nuclear fuel	1.04E-01	1.09E-01	1.16E-01

Bio: H2 use caused 39.0%, with 22.0% connected to Nitrogen production. The rest of the impacts caused by H2 were linked to energy production through wind turbines, with no further specification. 37.3% had derived from the nutrient solution, consisting of 5.7% magnesium sulphate, 12.4% sodium phosphate, and 16.8% potassium carbonate. 10.4% arose from the production of CO2.

Hydro: 43.4% came from H2 production. 36.6% derived from the nutrient solution, made up of 5.5% magnesium sulphate, 12.0% sodium phosphate, and 16.3% potassium carbonate. 9.3% were connected to the production of CO2.

Geo: 45.7% were caused by H2 use, with 19.8% related to nitrogen production. The rest of the impacts from H2 could be traced back to geothermal electricity production, which made up 40.0%; Electricity needed for all other foreground processes made up 9.5%. Of the effects on geothermal power production, 20.1% were linked to steel production for deep-well drilling and 9.2% to deep-well simulation. 34.4% stemmed from the nutrient solution, consisting of 5.1% magnesium sulphate, 11.1% sodium phosphate, and 15.1% potassium carbonate.

ILCD 2.0 2018 midpoint, human health, non-carcinogenic effects

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	1.57E-01	3.07E-01	5.30E-01
wood ash mixture, pure	6.40E-01	5.48E-01	
electricity, high voltage	2.03E-01	1.77E-01	
steel, low-alloyed		8.03E-02	3.01E-01
copper			
drilling waste			3.01E-01
wastewater, average		-1.12E-01	-3.54E-01
zinc monosulfate			1.25E-01
H3PO4 purification residue			9.68E-02

Bio: 20.0% related to the heat and power co-generation process, and 64.0% to landfilling the wood ash mixture; these impacts were mainly linked to CO2 use and electricity use for all foreground processes except H2, with 48.0% and 29.5% contributions. 10.9% were related to the production of H2 and 9.3% to the nutrient solution.

Hydro: CO2 contributed 62.4%, and heat for LP steam 11.2%; these flows were the main contributors towards heat generation and landfilling of wood ash mixture, with 17.0% and 54.8% contributions. 20.7% related to the H2 production and 13.7% to the nutrient solution. The treatment of wastewater had a positive contribution of 11.8%.

Geo: 87.6% related to deep-well drilling for geothermal power, 26.0% linked to landfarming drilling waste and 47.5% to steel production. 56.3% of the burdens associated with deep-well drilling were caused by H2 production and 20.2% by the electricity demand for all other foreground processes. The nutrient solution contributed 43.5%; 25.1% through sodium phosphate, while potassium carbonate and zink monosulfate had equal contributions of 6.3%. Wastewater treatment has a positive impact of 37.5%, while CO2 made up 8,0%.

ILCD 2.0 2018 midpoint, human health, ozone layer depletion

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	2.19E-01	2.62E-01	1.85E-01

Bio: 53.2% linked to heat and power co-generation, 30.0% through wood chips, and 22.1% through the Rankine cycle. The main contributors to these impacts were CO2 production, with 31.7%, and electricity production, except for H2 generation, with 20.1%. H2 use was responsible for 19.4%, 12.7% related to windmills. The

petroleum	5.03E-01	5.00E-01	4.80E-01
refrigerant R134a	2.05E-01	1.47E-01	
transport, pipeline, long-distance, natural gas	7.18E-02	9.12E-02	1.29E-01
coke			8.12E-02
natural gas, high pressure			6.28E-02
trichloromethane			
sodium hydroxide, without water, in 50% solution state			6.22E-02

nutrient solution contributed 20.3%, with sodium phosphate contributing 9.4% and potassium carbonate contributing 7.8%.

Hydro: 38,0% linked to the heat and power co-generation process; of these impacts, 15.8% related to the organic Rankine cycle, and 21.5% to wood chips. CO2, with a 34.5% contribution, was the leading cause of these burdens. H2 generation made up 27.9% of impacts, of which 20.4% came from wind turbine production. 24.4% stemmed from the nutrient solution, with sodium phosphate contributing 11.1% and potassium carbonate contributing 9.3%.

Geo: 43.6% related to electricity for H2 production and 12.1% to the electricity needed for all other foreground processes. Almost all of these impacts tracked back to deep-well drilling, contributing 51.0%, with 32.1% caused by steel production and 5.8% by portland cement. 31.6% of total effects originated from the nutrient solution, with sodium phosphate contributing 14.1% and potassium carbonate contributing 11.8%.

ILCD 2.0 2018 midpoint, human health, photochemical ozone creation

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	3.74E-01	5.20E-01	4.72E-01
electricity, high voltage	5.71E-01	4.80E-01	1.75E-01
power sawing, without catalytic converter	5.47E-02		
coke			1.62E-01
sinter, iron			7.56E-02

Bio: 51.0% were caused by the heat and power co-generation process, and 15.7% by the production of wood chips. Of this total contribution, 39.0% were caused by CO2, 7.1% by heat, and 30.7% by electricity for all foreground processes, except for H2 generation. 13.7% came from H2 creation, which was related to windmill production. 14.0% stemmed from the nutrient solution, of which sodium phosphate contributed 6.5%.

Hydro: 46.1% came from CO2 and 8.3% from heat for LP steam. These impacts were almost exclusively related to the heat and power co-generation process and the production of wood chips, with contributions of 39,0% and 12.0%. 23.1% arose from H2, mainly caused by the production of windmills. The nutrient

transport, freight, sea, bulk carrier for dry goods			6.13E-02
blasting			5.42E-02

solution comprised 18.2%, with sodium phosphate and potassium carbonate contributing 8.3% and 6.0%.

Geo: 44.8% were connected to electricity generation for H2 production and 16.2% to the electricity needed for all other foreground processes. Almost all these impacts came from deep-well drilling. 48.1% of the effects of deep-well drilling came from steel production and 5.2% from portland cement. 28.5% stemmed from the nutrient solution with sodium phosphate and potassium carbonate making up 12.7% and 9.2%. The contribution of CO2 was 6.3%.

ILCD 2.0 2018 midpoint, human health, respiratory effects, inorganics

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	2.97E-01	3.99E-01	4.62E-01
electricity, high voltage	5.69E-01	4.31E-01	
heat, district or industrial, other than natural gas	6.72E-02	7.62E-02	7.58E-02
electricity, high voltage, for internal use in coal mining	6.70E-02	9.40E-02	2.30E-01
diesel burned in building machine			
sinter, iron			1.52E-01
sulfuric acid			7.96E-02

Bio: 36.8% came from CO2 and 23.5% from electricity for all foreground processes, except H2 production, while 6.6% related to heat for LP steam. These impacts came from the heat and power co-generation process and wood chips production, with 55.0% and 6.5% contributions. H2 production contributed 15.4%, stemming from windmills. The nutrient solution encompassed 18.1%, with sodium phosphate contributing 9.7%.

Hydro: 41.3% came from CO2 and 7.5% from heat for LP steam, mainly stemming from the heat and power co-generation process. 25.1% come from H2 production, originating from windmills. 22.6% were linked to the nutrient solution, of which sodium phosphate contributed 11.8%, and potassium carbonate contributed 5.4%.

Geo: 39.7% were related to electricity generation for H2 production and 14.3% for electricity generation needed for all other foreground processes. These impacts mainly came from deep-well drilling, with 49.7% originating from steel production. The nutrient solution caused 37.1%, with sodium phosphate and potassium carbonate contributing 19.5% and 8.8%. The contribution of CO2 was 5.9%.

ILCD 2.0 2018 midpoint, resources, dissipated water

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	4.11E-01	7.88E-02	1.87E-01
electricity, high voltage	2.87E-01	9.21E-01	8.13E-01
wastewater, average	-2.19E-01		
sulfuric acid	1.79E-01		
phosphoric acid, fertiliser grade, without water, in 70% solution state	1.72E-01		
phosphoric acid, industrial grade, without water, in 85% solution state	1.71E-01		

Bio: 67.8% was related to the nutrient solution. Sodium phosphate and potassium carbonate contributed 59.2% and 5.2%. Of the former, 34.9% came from phosphor fertiliser. H2 generation made up 27.4%, mainly related to wind-mill production. 13.2% came from CO2 caused by the heat and power co-generation process, while wastewater treatment had a positive contribution of 23.1%.

Hydro: electricity generation for all foreground processes, except for H2 generation, contributed 89.8%, caused by non-alpine hydro reservoirs. The nutrient solution contributed 7.2%, mainly through sodium phosphate.

Geo: electricity generation was the driving factor of water use, with 53.0% related to H2 production and 18.9% to all other foreground processes. These impacts were caused by non-alpine hydro reservoirs needed for deep-well drilling. The nutrient solution made up 19.9%, mostly stemming from sodium phosphate.

ILCD 2.0 2018 midpoint, resources, fossils

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	1.66E-01	1.59E-01	1.72E-01
petroleum	2.63E-01	2.12E-01	1.43E-01
hard coal	2.68E-01	2.97E-01	4.12E-01
natural gas, high pressure	1.36E-01	1.40E-01	1.42E-01

Bio: the nutrient solution was the most substantial contributor with 36.6%, the impacts stemming from sodium phosphate and potassium carbonate with 14.0% and 16.3% contributions. H2 generation made up 33.7% originating from wind-mill production, whereas electricity generation for all other foreground processes made up 11.8%. CO2 accounted for 14.2% of total impacts.

Hydro: 41.7% stemmed from H2 generation, linked to wind-mill production. Electricity for all other foreground processes made up

nylon 6-6, glass-filled	5.77E-02	7.52E-02	
hard coal, run-of-mine	5.97E-02	6.45E-02	7.83E-02
uranium ore, as U	5.07E-02	5.27E-02	5.33E-02

6.7. The nutrient solution encompassed 36.0%, caused by sodium phosphate and potassium carbonate with 13.5% and 15.8%. CO2 accounted for 12.5% of total impacts.

Geo: of the total fossil resource use, 55.0% were related to deep-well drilling for geothermal power, most originating from steel production. H2 generation, with 42.3%, was the main contributor to these impacts. Electricity generation for all other foreground processes made up 13.6%. The nutrient solution led to 32.4%, with sodium phosphate and potassium carbonate making up 12.0% and 14.0%. CO2 accounted for 5.4% of total impacts.

ILCD 2.0 2018 midpoint, resources, land use

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	3.24E-02	5.90E-02	3.02E-01
wood chips, wet, measured as dry mass	9.68E-01	9.41E-01	1.04E-01
phosphoric acid, fertiliser grade, without water, in 70% solution state			3.63E-01
geothermal power plant, 5.5MWel			8.08E-02
sawlog and veneer log, softwood, measured as solid wood under bark			8.92E-02
road			6.14E-02

Bio: 54% related to CO2 production, 31.8% to electricity generation and 9.7% to heat generation; almost all the land occupation stemmed from the production of wood chips, which comprised 92.5%.

Hydro: 79.9% of total impacts were related to CO2 production and 14.4% to heat generation. These impacts were almost exclusively traced back to the production of wood chips, responsible for 89.9% of all LU.

Geo: the nutrient solution encompassed 60.0% of overall land requirements; 49.2% originated from sodium phosphate and 7.5% from potassium carbonate.

ILCD 2.0 2018 midpoint, resources, minerals and metals

product	2050Bio	2050Hydro	2050Geo
Total	1.00E+00	1.00E+00	1.00E+00
Rest	1.39E-01	1.54E-01	9.69E-02

Bio: The nutrient solution caused 76.8% of the total depletion of minerals and metals, with sodium phosphate, cobalt, and potassium carbonate making up 59.7%, 8.1%, and 6.6%, respectively. H2 production was responsible for 14.3% of the total use.

zinc concentrate	6.04E-01	5.94E-01	6.83E-01
lead concentrate	1.45E-01	1.33E-01	1.38E-01
lime	1.12E-01	1.19E-01	8.27E-02
chromite ore concentrate			

Hydro: The nutrient solution comprised 74.6% of overall mineral and metal use; 57.7% of this flow originated from sodium phosphate, 7.8% from cobalt, and 6.4% from potassium carbonate. H2 production made up 18.5%.

Geo: The nutrient solution utilized 68.3% of minerals and metals; 52.9% originated from sodium phosphate, 7.2% from cobalt and 5.8% from potassium carbonate. H2 production made up 15.6% and CO2 8.4%.

6.4.2.2 Contribution analysis for SBM and discussion on future development

Climate change: for SBM BR, 48% of impacts on climate change were traced back to the clear-cutting of primary forests; a further 16% were linked to LUC. For soybean production from the rest of the world, almost 60% were directly related to soybean production, which could not be further specified. Nearly 25% were caused through LUC, mainly associated with the clear-cutting of secondary forests.

Neither the direct impacts on climate change through soybean production nor the indirect impacts on LUC are expected to decline over the coming decades. Even if clear-cutting from primary and secondary forests were decreased, this would still lead to forgone carbon sequestration, meaning that on the land where soybeans are planted, the amount of carbon that could be sequestered by a tree instead needs to be considered. "Natural regeneration" should be seen as the baseline upon which LUC is evaluated, even if the land is currently not in use (Koponen & Soimakallio, 2015). Such an approach is mainly neglected in agro-LCA studies (Koponen & Soimakallio, 2015).

Therefore, it is concluded that impacts on climate change caused by SBM production will not be significantly reduced until 2050. Thus, the advantages of MP over SBM are not expected to alternate for climate change.

Fresh water and terrestrial acidification: impacts were very dispersed over numerous processes. The most relevant single contributor was clear-cutting of forests, with 20% implications from primary forests in Brazil and roughly 13% from secondary forests from the rest of the world. The next most considerable contribution was soybean production, reaching 6-8% for the two production locations. For soybean production for the rest of the world, transport made up about 7%.

As impacts are very dispersed, it was harder to predict to what extent future changes to the background system will impact the results of soybean production regarding acidification. Regardless, no substantial improvements can be expected for the most prominent contributors, namely clear-cutting of forests and soybean production, as methods for the latter are well established. Given that pH levels increase significantly after clear-cutting (Nykvist & Rosén, 1985), a similar approach as "natural regeneration" should be taken for impacts on acidification. Therefore, effects caused by clear-cutting are not expected to decrease over time if allocated appropriately.

Thus, no significant changes can be anticipated for SBM production until 2050 for acidification. Results of the comparative analysis in 2050 are therefore unlikely to change.

Freshwater ecotoxicity: as the impacts were more extensive than those of the MP production systems by a factor of three, it was assumed that a reduction to make SBM production compatible seemed unrealistic. A CA was therefore neglected.

Freshwater eutrophication: here, most impacts were related to SBM production. Unfortunately, no further information could be retrieved through the Sankey function about the origin of these impacts. Due to these constraints, a more detailed CA was not possible.

Marine eutrophication: as impacts of SBM for RoW and BR were several orders of magnitude higher than all other reference flows, a CA would not have affected the results of the comparative analysis.

Terrestrial eutrophication: most impacts were directly linked to SBM production and 7.7% to transport. 40.0% of the former came from clear-cutting primary forest to arable land; the rest of the impacts could not be identified. As the effects of clear-cutting regarding terrestrial eutrophication are permanent due to issues related to nutrient run-off, an improvement in this IC can not be expected in this respect. As far as this analysis was conducted, no changes to the comparative study regarding the future SBM product systems are likely.

Carcinogenic effects: as impacts of SBM were 4-7 times lower than those of the MP product systems, a further analysis was unnecessary as this would not change the results of the comparative analysis.

Ionising radiation: the impacts of the incumbents were almost half compared to all other reference flows, even at fully developed MPL. A further CA for SBM, therefore, was redundant.

Ozone layer depletion: 18.9% of impacts were related to diesel production for transport. The only other single contributor above 5.0% was pesticide production with 19.1%. No evidence could be retrieved to state if the impacts of pesticides on ozone layer depletion were likely to increase or decrease over time. In summary, it is assumed that even slightly decreased effects would probably not change the results of the CA significantly.

Photochemical ozone creation: similarly to terrestrial eutrophication, around 50.0% were directly related to clear-cutting primary forest to arable land. As the effects of clear-cutting are permanent, an improvement in the impacts can not be expected in this respect. Based on this assessment, a considerable future reduction in results can not be expected.

Respiratory effects inorganics: 27.0% stemmed from clear-cutting primary forests for arable land. 6.5% was derived from phosphorus fertiliser. As a major increase in recycled phosphate can be expected in the coming decades, with the EU's incentive that expects all phosphate from municipal sewage sludge to be recycled until 2030, these impacts are expected to decrease over time.

8.4% are related to diesel. Further, 11.4% pertained to transport, assumed to originate from non-renewable transportation fuel. A decrease in emissions can be expected, yet this would hardly affect the comparative analysis, as the results of SBM were already better than all other MP reference flows, even Geo 2050, which was almost at par with SBM BR.

Dissipated water: the main contributors were SBM production and sulphuric acid production. No indication was found suggesting that water use for these processes would change over time. Therefore, improvements in water use for SBM production by 2050 seemed negligible.

Resources, fossil: 40% of impacts were related to petroleum, likely stemming from transportation and machinery, even though this could not be further validated. Therefore, these impacts will likely further reduce over time as renewable energy sources will increasingly power these sectors. Still, this will not impact the comparative analysis as the advantage over the MP product systems is already unreachable even at a fully developed MPL of 50-100.

Land use: as almost all impacts on LU are directly related to soybean plantation, results will likely not change significantly over time. They will, therefore, not skew the outcome of the comparative analysis.

Minerals and Metals: 41% of impacts can be traced back to zinc mine operations used for lime production, which is a prominent input for soybean production. Besides the effects caused by lime production, 27% of overall impacts are related to phosphate fertiliser, 21% of which were linked to primary zinc production and 7% to primary lead production.

There are ongoing efforts to recycle zinc and lead through melting point recycling. However, no evidence could be found to what extent this will influence future market developments. As more phosphate recycling can be expected, impacts on minerals and metals will likely decrease over time for SBM production, further increasing the advantage over the assessed MP product systems.

Based on the above CA for SBM, slight improvements can be expected for some ICs. Regardless, significant changes to the results of the comparative analysis based on a future SBM product system cannot be anticipated overall.

6.4.3 Sensitivity analysis

Besides the CA, a SA was performed to test hypotheses regarding the multifunctional process and the system boundary. The presumably most sensitive parameters are only relevant for Bio and Hydro, as Geo is not influenced by intermittent energy supply or CO2 price fluctuations. The latter was thus left out of the SA for the most part. The below results of the tested parameters are relative to Bio and Hydro in 2050.

The scenario analysis performed in phases two and four included SA to a certain degree; nevertheless, testing the sensitivity regarding assumptions made for the only multifunctional process had been neglected so far. Thus, research was performed to validate the impact the CO2 price would have on the MP product system's overall environmental performance and the model's robustness. As the BCHP process is producing electricity, heat, and CO2, the capturing price of the latter was assumed to have a substantial impact on the total results, especially given the high energy demand of this product system, as previous studies have highlighted (Sillman et al., 2020); (Järviö et al., 2021). The default CO2 capturing price for all previous scenarios was 120€/t. During the SA (table 8), a CO2 price of 25€ and 280€ was tested per ton caught, in line with the highest and lowest estimations found in the literature, as discussed in section 4.2.4. All calculations regarding the economic allocation factors were documented in appendix 1.

As displayed in table 8, the CO2 capturing price largely impacted the overall performance of the MP product system in three out of four cases. Due to a lower price, there was an impact increase for all ICs for Bio. On average, impacts increased by approximately 30%. A decreased CO2 capturing price for Hydro lowered emissions by roughly 7% on average across all ICs. On the other hand, a higher capturing price reduced impacts for Bio by roughly 15% on average, while this change barely affected Hydro's performance.

Besides the CO2 capturing price, the system boundary displayed a shortcoming which needed further evaluation: it turned out that H2 and NH3 production through water electrolysis is only cost-competitive under continuous energy supply due to high capital expenditure costs. This information was gathered through informal conversations with experts at a late stage of this study, while these claims were verified through literature findings (Fasihi & Breyer, 2020). Regardless of the timing of these findings, it was possible to integrate the information into the assessment through a SA.

The results showed the sensitivity towards increased baseload electricity use instead of solely using wind energy for H2 and NH3 production. Increasing the share over intermittent supply led to a sharp increase in results for Bio by an average of approximately 65% across all ICs. LU increased by almost 150%, while results of numerous other ICs rose by over 100%. Only carcinogenic effects and mineral and metal use decreased, yet minimally compared to other ICs. For Hydro, a shift towards more baseload electricity production decreased all emissions, apart from climate change and water use, which increased by approximately 50% and 250%, respectively. When exchanging hydropower through non-alpine reservoirs with electricity through river run-off, the same reductions for all ICs were observed, except for the increase in climate change and water use. In these two ICs, results decreased by roughly 35% and 92%, respectively, relative to Hydro 2050. This stark contrast can be explained as biogenic CH4 emissions are only assigned in the case of hydro reservoirs (see section 5.1.4). At the same time, the water from the river mainly stays within the environmental system boundary.

In addition to the tested parameters in table 8, the SA evaluated the potential impact reduction of using O2 left over from the MP production process. Assuming 99% nutrient utilisation and a large-scale on-site green NH3 production facility, approximately 1.6kg O2 could be produced per kg MP (see appendix 1). This presumes all O2 left over from the fermentation and electrolysis can be utilised. The average market price of compressed O2 at 200 bar is 2.8€/kg, based on public tenders prices from 2014-2018 of 1.4-4.2€/kg (Nicita et al., 2020). Thus 1.6*2.8 = 4.48€ could be gained per kg MP from utilising the leftover O2. At a current MP production price of 5.3-9.1€/kg MP or above (Nappa et al., 2020), the average cost is assumed to be at least 7.2€/kg. Based on economic allocation, this would offset 30-40% of MP's emissions, regardless of the type of baseload energy, while the technology is still developing. If the MP price is as low as 2.1€/kg, which is only possible under optimal conditions (Nappa et al., 2020), thus at a high MPL as discussed in section 3.5, this will decrease emissions by 60-65%. However, as previously mentioned, such a low production cost would only be possible if green H2 prices were reduced substantially.

The SA's results showed the sensitivity towards all tested parameters. In the case of Bio, there is a trade-off between economic and ecological benefits. As MP's production costs decreased due to cheap CO2 capturing prices and continuous H2 generation, thus lowered capital expenditure costs, results increased considerably. With a higher capturing price, results across all ICs reduced, yet much less. For Hydro, economic and environmental benefits were aligned, yet only if non-alpine hydro reservoirs were replaced by run-off-river electricity production. Otherwise, results for climate change and water use increased, for the latter disproportionally. While a lower CO2 capturing price also reduced Hydro's emissions, it barely affected the results. Including O2 as a by-product would make MP more cost-competitive to SBM while lowering externalities noticeably. Under optimised system parameters, production costs might match those of SBM in such a case.

Table 8. Relative characterisation results of SA based on different CO2 prices, relative to Bio and Hydro 2050

Impact category/scenario	Bio; CO2 price: 120€/t	Bio; CO2 price: 25€/t	Bio; CO2 price: 280€/t	Bio; H2 through CHP & wind	Hydro; CO2 price: 120€/t	Hydro; CO2 price: 25€/t	Hydro; CO2 price: 280€/t	Hydro; H2: non-alpine reservoir & wind	Hydro: run-off-river	Hydro; H2: run-off- river & wind
climate change total	100.0%	118.3%	91.6%	127.2%	100.0%	97.9%	100.3%	144.1%	79.4%	65.3%
freshwater and terrestrial acidification	100.0%	141.1%	81.2%	192.7%	100.0%	91.3%	101.4%	91.5%	99.8%	90.5%
freshwater ecotoxicity	100.0%	123.4%	89.3%	128.4%	100.0%	96.7%	100.5%	65.7%	100.1%	65.9%
freshwater eutrophication	100.0%	111.6%	94.7%	109.4%	100.0%	98.4%	100.3%	79.8%	99.6%	78.2%
marine eutrophication	100.0%	156.0%	74.4%	227.6%	100.0%	84.9%	102.5%	88.5%	99.7%	87.5%
terrestrial eutrophication	100.0%	152.0%	76.2%	221.0%	100.0%	86.4%	102.2%	94.8%	99.8%	94.2%
carcinogenic effects	100.0%	107.7%	96.5%	75.9%	100.0%	99.1%	100.1%	58.0%	100.1%	58.6%
ionising radiation	100.0%	123.3%	89.3%	142.7%	100.0%	96.4%	100.6%	91.3%	98.4%	85.1%
non-carcinogenic effects	100.0%	157.6%	73.7%	234.5%	100.0%	83.0%	102.8%	91.8%	99.9%	91.4%
ozone layer depletion	100.0%	135.2%	83.9%	177.7%	100.0%	93.3%	101.1%	93.3%	99.6%	91.9%
photochemical ozone creation	100.0%	146.6%	78.7%	205.5%	100.0%	89.3%	101.7%	91.4%	99.7%	90.2%
respiratory effects, inorganics	100.0%	144.9%	79.5%	200.5%	100.0%	90.1%	101.6%	90.4%	99.7%	89.2%
dissipated water	100.0%	121.6%	90.1%	138.7%	100.0%	99.7%	100.1%	355.1%	9.4%	7.7%
fossils	100.0%	118.3%	91.6%	126.2%	100.0%	97.4%	100.4%	81.3%	99.5%	79.2%
land use	100.0%	159.6%	72.7%	241.3%	100.0%	81.0%	103.1%	96.2%	100.0%	96.3%
minerals and metals	100.0%	102.1%	99.0%	93.6%	100.0%	99.7%	100.0%	85.9%	99.9%	85.6%
average change across all ICs	100%	132.5%	85.1%	165.2%	100%	92.8%	101.2%	106.2%	92.8%	78.5%

Noticeable improvement
Minor change
Noticeable decline

7 Discussion

7.1 Limitations of the study

This section reflects upon the study's shortcomings and evaluates what impact these limitations might have on the overall results.

LCA is limited to assessing particular ICs, while some environmental externalities are not quantified. Issues related to biodiversity were thus not considered. As biomass plantation and hydro reservoirs affect biodiversity issues, as discussed in previous sections, this thesis is limited in this way. Besides limitations of the LCA method, the out-of-date background data sources and incomplete CFs posed further restrictions to the validity of the overall analysis. These missing CFs limit the assessment of toxicity-related ICs, which applies to the emerging and incumbent technology. One could thus argue that this incompleteness might have a limited effect on the comparative analysis, as these shortcomings apply to both systems. In addition to known limitations, there remain "unknown unknowns" regarding novel impacts yet to be discovered. No evidence was found to indicate which unknown CFs or ICs might become relevant to the two product systems in the future.

Besides the methodology's limitations, more accuracy of the results could have been achieved by including specific cut-offs, which would take more time and access to first-hand data. One could, for example, expand the analysis to include the recycling of water used for the fermentation process. However, this would have limited influence on the current results, as the direct water input was not identified as a hotspot for any MP product system. Indeed, when comparing the impacts of the product systems' water use, which was between 500-4500kg, to the direct water input into the fermentation process, which is 15.5kg/kg MP produced, it is apparent that the latter has a minimal contribution. Besides, the overall water demand for MP increased when water was recycled on-site due to additional electricity demand for the recycling (Järviö et al., 2021). In the same publication (Järviö et al., 2021), the water effluent treatment was modelled according to nutrient utilisation. This approach was neglected in this study due to time constraints. Evaluating how this inclusion would have changed the comparative results was not possible.

Specific choices were made regarding the inclusion of some parameters over others. This prioritisation was necessary to confine the number of possible scenarios. Priority was given to parameters expected to impact the overall performance significantly. AEL was thus used, neglecting differences that may occur compared to PEM. As elaborated in section 4.1.3.1, using PEM would slightly worsen the total results of the MP reference flow. For Hydro and Geo 2050, the results of the comparative analysis would likely not be affected by this change. For Bio 2050, this shift in results might lead to SBM showing an overall advantage. Furthermore, a BCHP unit was used with a 6.67MW capacity. This moderate unit produces heat as the main product and electricity as a by-product while using wood chips. More accuracy of the results regarding Bio could have been achieved, for example, if other production methods varying in size or material input had been considered.

Including further cut-offs such as facilities, process equipment, and their end-of-life treatment would burden the MP reference flows more. As the overall results for Hydro and Bio in 2050 align closely with those of SBM, such inclusion might make the incumbent more competitive. Nonetheless, further analysis would be needed to determine the impacts on the results caused by these changes. On the other hand, E-TAC is expected to reduce material efficiencies compared to the AEL water splitting method. However, none of the electrolyser's materials were identified as hotspots during the CA. Thus, material efficiency gains related to E-TAC would unlikely impact the comparative results. Apart from cut-offs performed for the emerging technology, the future energy mix was not included for either of the two product systems. Had this been included, it might have changed the comparative results given their close alignment. However, to confirm this statement, further research is required.

7.2 Validity and assumptions

Besides limitations caused by lacking data, it is essential to reflect upon assumptions made in the study to validate specific modelling choices. This validation is discussed in the following section.

As the number of scenarios had to be confined, the second phase only evaluated bioenergy as a baseload solution. This confinement posed a challenge when assessing the viability of some technological parameters like DAC. Indeed, there seems to be a trade-off between water and LU connected to the source of CO2 production. The main contribution towards LU for both Bio and Hydro was related to CO2 from PCC. However, using DAC instead of PCC for Hydro would increase electricity, increasing water use even more. In the case of Bio, using DAC instead of PCC would assign more emissions to the CHP process and would thus drive up the results for all ICs.

Meanwhile, using PCC instead of DAC would increase LU for Geo while decreasing water use. Given the high impacts compared to SBM for the latter, this seems a favourable solution. However, given Iceland's shortage of available CO2, the feasibility of such a solution appears questionable. Using DAC instead of PCC is thus not recommended for Hydro and Bio, while for Geo, the opposite is true.

During the second phase and the SA conducted in the fourth phase, it became apparent how sensitive the results were based on the tested parameters. Given the close alignment between the emerging technology and the incumbent one, any change in these parameters would thus affect the comparative results. The difference in wind turbines' size, the source of CO2, the CO2 capturing price, and the share of baseload energy considerably affected the overall performance, yet only for Bio and Hydro. Changes to the heat source, the price of MP production, and the reuse of O2 influenced all MP reference flows. If 3MW wind turbines were used instead of 1MW ones, SBM would likely outperform Bio and Hydro in 2050. However, it is questionable if miniature windmills generally perform better than larger ones ecologically speaking. Besides the size, the environmental performance might also be influenced by the windmills' type and model. The amount of insight MP companies have on the ecological impacts of windmills is questionable. This lack of control over some tested parameters poses a challenge to evolving companies in the field, which calls for careful evaluation of each individual project.

If H2 and NH3 were to be produced through both baseload and wind energy, as they should from an economic point of view, changes based on the type of windmills would be less substantial. Regardless, such an energy mix would lead to a clear disadvantage for Bio 2050 compared to SBM. For Hydro, the decreased emissions would not be enough to show a clear total advantage over SBM in 2050, even if river-run-off is used instead of non-alpine reservoirs. Nevertheless, using river-run-off instead would reduce the water use to 0.32m3/kg MP produced, bringing it down to the same magnitude as that of SBM, which is approximately 0.2m3 per 1.3kg produced.

A lower CO2 price would decrease production costs; however, this would also increase the environmental impacts in the case of Bio. Given a capturing price of 25€/t CO2, or lower, would lead to SBM's overall advantage over Bio in 2050. This advantage can be explained as the inclusive impacts of eutrophication would exceed those of the incumbents. Additionally, the advantage over SBM in LU would vanish in this case. On the other hand, an increased capturing price would reduce Bio's impacts yet drive MP's costs. Despite considering this impact reduction, no overall advantage of Bio over SBM appears in 2050. This is true, as results related to acidification, water use, respiratory effects, photochemical ozone creation, fossil use, and minerals and metal use remained above those of the incumbents. For Hydro, the CO2 price does not affect the comparative results.If O2 was utilised as a by-product, this could potentially reduce Bio's impacts in 2050 to the point where it outperforms SBM, even at a low CO2 capturing price. Including O2 recycling, in the case of Hydro, would lead to better overall results than SBM in 2050, regardless of the capturing price. Depending on MP's production costs, and the CO2 capturing price in Bio's case, an overall advantage might occur for all three MP reference flows over SBM, even at a lower MPL.

Furthermore, the electrolysis efficiency primarily dictated the environmental improvements over 30 years. In 2050, a value of 41.9kWh/kg H2 was assumed based on projections of lab-scale experiments. Such a high voltage efficiency is out of reach for current electrolyte electrolysis operating facilities. The comparative results thus depend on the successful implementation of E-TAC electrolysis or other revolutionary emerging technologies in the field. If neglecting such high efficiency, SBM would likely show an overall advantage over Bio and Hydro in 2050. For Geo, a total advantage would become more debatable.

7.3 Links to literature

This section compares the results to other MP LCAs to evaluate the plausibility of the results.

No two LCAs are the same, yet some display similarities, while differences can spark further discussion. However, different modelling choices, ICs, characterisation models, or category indicators might limit the comparative assessment between the studies. Regardless, some contrasts were drawn with two previous LCAs (Cumberlege et al., 2016); (Järviö et al., 2021). The results of a third previously discussed MP LCA study (Sillman et al., 2020) were not further evaluated, as the publication was based on assumptions and calculations rather than empirical data.

Firstly, the results were compared to the best-case scenarios of the only previous MP study based on empirical data (Järviö et al., 2021) to validate them. Different ICs were used in the publication (Järviö et al., 2021), yet some had the same category indicator as the ICs of the ILCD. Matching category indicators were displayed in table 9 for easy comparison.

Table 9. Comparison of category indicators with previous MP LCA (Based on Järviö et al., 2021)

	Climate change (kg CO2-eq)	Freshwater eutrophication (kg P-eq)	Marine eutrophication (kg N-eq)	Water scarcity (m3)
FHE	1.05E+00	2.63E-04	1.26E-05	9.70E-01
FHE - wind (FI)	1.20E+00	7.93E-04	5.13E-05	7.94E-01
Hydro 2050	6.92E-01	2.59E-04	9.27E-04	4.16E+00

FHE in the study (Järviö et al., 2021) was short for using Finish hydro energy, while MP production using wind energy was also assessed in Finland. The FU in the publication (Järviö et al., 2021) was the same as in this study. The results for Hydro were in the same order of magnitude as FHE and FHE wind (FI) for climate change and freshwater eutrophication. For marine eutrophication and water scarcity, on the other hand, the impacts of Hydro 2050 far exceeded those of the reference study (Järviö et al., 2021). For the former IC, results were between 17-68 times higher. There are two plausible reasons for this increase. Firstly, the upstream emissions for CO2 were accounted for in this study, which showed over 60% contribution to marine eutrophication. Secondly, windmill production for Hydro 2050 contributed almost 30%, further increasing the results compared to FHE.

For water scarcity, the impact of Hydro 2050 was between 4-5 times higher than the compared values. For Hydro 2050, almost 90% were related to non-alpine hydro reservoirs for electricity production. This electricity was exclusively needed for all MP foreground processes except H2 and NH3 production. As H2 was exclusively produced through wind energy, fewer burdens were thus put on water use compared to FHE. The impacts for Hydro 2050 in this IC were thus expected to decrease rather than increase compared to FHE. A conceivable reason for this difference might be the type of hydropower used in the reference study which was not specified. As shown in the SA, the water use could be reduced by 90% if river-run-off was used instead of non-alpine reservoirs, which would justify these inconsistencies. Additionally, there is a difference in water lost due to evaporation between various sources. 0.0292m3/kWh was assigned as environmental flow in this study (Treyer, 2019); however, lower values of 0.0167m3/kWh were reported by older ecoinvent versions (Wernet et al., 2016), further justifying the differences.

Secondly, another literature comparison was made to methanotrophic bacteria. As presented in section 1.1, the environmental impacts of FeedKind, were 2.23kg CO2-eq, 10kg water use, and 0m2 for LU, based on a best-case scenario per kg biomass produced (Cumberlege et al., 2016). Compared to the three MP reference flows, the methanotroph's impacts on climate change are approximately double at the pilot-scale and around three-four times at high MPL. FeedKind can thus not compete with MP in this IC. The ILCD for LU is given in points; therefore, no comparison could be drawn. For water use, the MP impacts for the best-case scenario are approximately 20 times as high as those of FeedKind. The latter's results for land and water use seem promising, though other ICs were neglected; at the same time, no insight into the modelling choices of the study (Cumberlege et al., 2016) were found, making the comparison limited.

Besides comparing MP's results to other studies, there are high fluctuations regarding climate change impacts and water use depending on the soybean plantation methods. These fluctuations are discussed in section 1.1 regarding 1.0kg of SBM produced. When scaling these values to match the FU of this study, the effects on climate change can be as low as 0.69kg CO2-eq and for water use as high as 605.8kg per 1.3kg SBM produced. Comparing these values to this study's various MP reference flows, their advantage over SBM for the former IC becomes less apparent. At TRL 6-8, the incumbent would thus display a slight benefit for climate change. At high MPL, only Bio would outperform SBM, while the other two reference flows would be equal. For water use, Bio's results were below 605.8kg, even in 2020, while for Hydro and Geo, this would only be true if river-run-off was used as an electricity source.

This analysis shows how sensitive the comparative results are depending on the soybean plantation method. However, in some ways, these advantages and disadvantages even out across the two ICs. In addition, the SBM reference values are the most extreme in the literature. They are thus not representative of the average method of SBM production.

7.4 Methodological reflections

The original framework (Delpierre et al., 2021) for performing explorative scenario analysis in conjunction with the ex-ante methodology for emerging technologies was further developed in this study. The following section reflects the methodological approach of the framework's four phases, presented in section 2.2.

Phase one: Gaining first-hand data on the foreground MP system, especially regarding the production steps from fermentation to drum drying, would have been valuable. The results would have thus been more accurate by gathering specialised equipment and facilities data. Three companies were unsuccessfully contacted, namely Air Protein in the USA, Solar Foods in Finland, and Deep Branch in the UK. Close to the end of this study, two other companies were found that produce MP, Arkeon and Econutri, both situated in Austria. Due to the study's advanced progression at the time, these companies were not contacted. Besides gaining first-hand data on the existing MP production system, it would have been valuable to retrieve more information on changing parameters that will likely influence the system's performance during the scale-up stage.

Phase two: Further insight from technology developers would also have been beneficial for this phase, such as assessing other forms of MP that fix N2 directly from the air (Hu et al., 2020). Drawing a comparative analysis between green on-site NH3 production and N2-fixing HOB may have been insightful. Unfortunately, no LCI data was available in the literature for such an analysis.

Phase three: Gathering first-hand information might have also given valuable insight into the moments when specific material and energy efficiencies could be expected. As nutrient utilisation and electrolysis efficiency levels are decisive for the overall environmental performance, gaining knowledge on when particular values are expected would have made the model more robust. In the current model, the progression of the nutrient utilisation level and the moment E-TAC water electrolysis becomes readily available for large-scale applications were based on assumptions. These assumptions were based on the general progression of emerging technologies and hence not very accurate towards these specific parameters. The E-TAC method was thus believed to only be available at high MPL in 2050, while the timescale to reach 99% nutrient utilisation was assumed to be the same. Between the pilot and large manufacturing scales, results differed by up to 35% in some ICs. These high fluctuations thus show the importance of precisely determining when these efficiencies can be expected.

Furthermore, conducting interviews with other firms that are not directly related to MP production would have been beneficial to gain a clearer picture of other material inputs and changing parameters. For example, the model for green NH3 production facilities excluded material inputs and facilities due to limited data availability; thus, any changing parameters related to these inputs were also neglected. Besides identifying further quantitative parameters, direct contact with experts might have provided information on what challenges companies face in implementing their products on a large scale. Due to limited data, the surrounding parameters were thus not as comprehensively assessed as the technical ones.

Phase four: A transitional approach was taken for three different baseload energy sources for the scenario analysis. Consequently, 12 scenarios gave a broad perspective of the total performance of MP production. Yet this approach did not allow for assessing other parameters over time. Assessing MP's progressive performance when using CO2 from fossil origins, for example, would have informed further discussion. As many GHG emitting industries are looking to reduce their emissions by offsetting them through CCU applications, showing the technology progression while including such parameters at the ex-ante stage would have been beneficial. However, modelling and assessing 12 scenarios already challenged the scope of this analysis, including any more possibilities would have thus not been feasible.

The developed framework included a CA of the incumbent system. The reason for this inclusion was to assess SBM's future development qualitatively. The CA identified the incumbent product system parameters likely to change over time. Further, a scenario analysis could have been performed for the incumbent system based on changes in the identified parameters, while also including the future energy mix for both product systems. This approach would reinforce the validity of the comparative results.

8 Conclusion and recommendations

8.1 Conclusion

This study assessed the environmental performance of MP compared to SBM as the most used alternative to fish meal. As previous MP LCAs displayed certain limitations, it was necessary to overcome these by testing various assumptions. The first sub-research question was. "*How to best allocate biogenic carbon from point sources for MP production?*". It was clearly shown that economic allocation is the only plausible way to allocate CO2 emissions from PCC or other point sources. In addition, it became apparent how sensitive the comparative results for MP production and SBM are to the CO2 capturing price.

The second sub-research question, "*From an economic and environmental perspective, what is MP production's most efficient technological setup?*" addressed economic feasibility, another shortcoming of previous studies. During the study's second phase, it was shown that an ecologically optimised MP production system should not supply steam, CO2, or NH3 from the chemical industry. Instead, steam should be provided through a BCHP unit, geothermal heat, or even natural gas, while NH3 should be produced on-site using green H2. CO2 should be of biogenic origin, while its production needs to be allocated appropriately. In addition, recovering O2 as a by-product is essential to making MP production economically and environmentally feasible. Electricity for all foreground processes, including H2 and NH3 production, should be supplied through baseload and wind power, except when the former is abundantly available.

Furthermore, SMR with CCS was also assessed in the study's second phase as an economically feasible alternative to green H2 production. Evaluating the results in the context of phase four's normalisation results, using blue over green H2 would increase the effects of ionising radiation and photochemical ozone creation, the latter by over 50%. At the same time, the impacts of climate change would only decrease to a minor degree. This overall decline in performance would probably even out MP's advantage over SBM in the best-case scenario, Geo 2050. SMR with CCS is thus not recommended for MP production, despite its price advantage over current green H2 generation.

The third sub-research question was: *"What are not previously identified hotspots for improving this technology?"*. Some hotspots of the different MP product systems were identified during the CA. Across all three reference flows, the nutrient solution was amongst the most prominent contributors for almost all ICs. These impacts can mainly be traced back to potassium carbonate and sodium phosphate. The latter was often linked to phosphoric acid production, more precisely to landfilling phosphoric acid purification residue. H2 production was another main contributor to all three MP reference flows. When H2 and NH3 were generated through wind turbines, as in the case of Bio and Hydro, these burdens were related to windmills; for some ICs, they were further traced to steel production. For Geo, H2 and NH3 generation caused large externalities traced back to deep-well drilling, primarily from steel and, in some ICs, landfarming drilling waste.

In the case of Hydro and Bio, the single most substantial contributor across all ICs was CO2 production leading to high impacts regarding eutrophication, non-carcinogenic effects, ozone layer depletion, photochemical ozone formation, respiratory effects, and LU. For Bio, landfilling wood ash mixture critically contributed to carcinogenic and non-carcinogenic effects and freshwater ecotoxicity to a minor extent. In the case of Hydro, landfarming wood ash mixture also contributed to the last two ICs, with these impacts exclusively related to heat generation. The heat and power co-generation process was another notable contributor to some ICs, with wood chip production contributing to some degree.

The last sub-research question focused on improving current methods of assessing the EF of emerging technologies. This study contributed to this development in three ways. Firstly, previous methods have neglected a stage where specific parameters can be tested to find optimal ecological performance at the pilot-scale. Such inclusion is essential for emerging technologies with a low TRL, as it can help stir early decision-making processes. Secondly, including a progressive development was valuable to assess not just if but also when an innovation would match the environmental performance of the incumbent. Lastly, performing a CA for the incumbent system gave valuable insight into the validity of the comparative results. This insight is beneficial in the case where comparative results closely align.

The main research question could be answered based on the results of the four phases, the conclusions of the sub-research questions, and the assessment of the study's assumptions and limitations. At the pilot-scale, non of the MP reference flows could compete with SBM. At high MPL, which can also be seen as the best-case scenario, only Geo outperformed the incumbent system. However, high impacts on mineral, metal and water use remained. Including O2 as a by-product would show this advantage more clearly, assuming a low MP production price. Such inclusion also makes Hydro competitive under optimal system conditions. Yet, this advantage would only occur at high MPL, given the results' dependency on the low product price and high production efficiency.

Nevertheless, regarding hydropower, only the scenario using electricity generation through river run-off was competitive with SBM concerning water use. In the case of Hydro and Geo, the origin and fate of water used for hydropower facilities must thus be evaluated carefully. In general, MP should only be produced in regions where water scarcity is not an issue. When considering continuous H2 and NH3 supply for economic feasibility, SBM outperformed Bio in 2050, even when considering full O2 recycling, a low MP production price, and low CO2 capturing costs. Additionally, general issues of biomass supply security make its use for MP production questionable. Furthermore, renewable energy supply security issues for other sectors need careful evaluation, as MP production could cause adverse effects in those areas.

It has been suggested by previous studies that MP would decouple protein production from LU while decreasing eutrophication and water use compared to field crops. This claim was only verified for Hydro and Geo for LU. For eutrophication, this was only true under optimised system conditions. In the case of water use, however, the opposite was true, unless electricity through river run-off was used. In this case, comparative results would even out in this IC. Additionally, it has also been recommended by previous studies to use fossil CO2 point sources. As this would decrease Geo's advantage at high MPL over SBM, such an approach is not recommended for MP as a feed alternative unless O2 was used as a by-product, given the close alignment of the comparative results.

The above conclusion is based on an optimal environmental performance, where NH3, CO2, and heat are produced on-site, while wind-mills with minor ecological impacts are used. Additionally, the electrolysis and nutrient utilisation efficiencies are the main dictators of the comparative results, which were assumed to be 41.9kWh/kg H2 produced and 99%, respectively. These values are based on lab-scale data and assumptions and are thus very uncertain. Results are therefore susceptible to these parameter changes. This sensitivity must be considered carefully when setting up MP feed alternative production facilities. Additionally, the results of this study should be viewed as recommendations rather than fixed statements, given the limitations of the LCA methodology, especially for emerging technologies.

Besides variable technical parameters, making the comparison uncertain, the study's results depend on broader socio-economic parameters for this novel protein to reach a high MPL, which is necessary to achieve an ecological and economic competitive advantage over the incumbent. Access to renewable H2 might pose a challenge to MP production companies. It is thus uncertain how much green H2 will be available for novel applications such as MP. On the other hand, more demand should create more supply, increasing economies of scale and reducing costs. Additional attention to such novel applications might thus incentivise more research and funding towards green H2 production.

Policy support is thus needed beyond CO2 price targets to drive developments in sustainable H2 generation. Besides cheap, renewable H2 supply, the novel protein will only become a viable feed alternative if high economies of scale are reached. Thus, it is necessary to promote it as a food alternative for human consumption where higher prices can be demanded. As MP would increasingly be used as a meat replacement, the production price and environmental burdens would likely decrease, making it more competitive with SBM in both respects. Successfully endorsing MP and making it part of ordinary dietary choices implies institutionalising environmental awareness and food supply issues while promoting novel protein alternatives. Solid network formation will be necessary to support lobbying and investment choices. Such developments will also foster intense learning, ultimately changing institutional structures and enabling MP to progress from a niche innovation to a well-established market.

8.2 Recommendations

More research would further evaluate the validity of the comparative results and the economic feasibility. The recommendations for future research are summarised in this section based on the study's overall assessment.

Firstly, despite its overall ecological advantage over meat products, MP intended for human consumption has previously been assessed based on certain limitations, as discussed in sections 3.1-3.6. Additionally, the EF of MP as a meat alternative has been evaluated in its powdered form. Further research is thus needed, comparing the novel protein as a final product vs various animal proteins while also considering other previous limitations. Such evaluation should critically assess CH4 emissions caused by the degradation of biogenic carbon inside artificial aquacultures in the comparative analysis between MP and fish. As this inclusion is essential in the case of hydro reservoirs, as elaborated in section 5.1.4, it seems equally necessary for aquaculture farms.

As previously discussed, green H2 will be needed by industries otherwise difficult to decarbonise, possibly posing limitations on large-scale implementation of MP production for feed alternatives. Using green H2 for MP manufacturing might thus lead to less sustainable H2 use in other sectors. Furthermore, using electricity from baseload sources might pose similar challenges. Instead of an attributional LCA, a consequential LCA would give valuable insight into such potential problem shifting. It might also be beneficial to evaluate other novel protein alternatives while comparing them to MP's EF. Methanotrophic bacteria such as KnipBio Meal might be a viable solution as a feed alternative if they are generated through by-products of bioethanol creation such as condensed distillers soluble. In this way, arable land is used to produce food and energy simultaneously, potentially making the question of energy vs crop production redundant.

It would be valuable to investigate when E-TAC water electrolysis would become commercially available, especially for novel concepts such as MP, to reduce the overall energy demand through H2 production. E-TAC and other green H2 production methods with similar efficiency levels are vital to MP's environmental and economic success. Given its significant financial advantage over other H2 production methods, research on this green H2 production method is essential. Further research into O2 recycling from water electrolysis and the fermentation process will also dictate the future viability of using MP as a feed alternative. Particular focus should also be placed on the different levels of water use, given the high impacts and variability in this IC. Additional evaluation should thus focus on the inconsistencies to other studies in this respect. Yet, grid stability should always be at the centre of such analysis.

Future research should evaluate alternative parameters and compare them to the identified hotspots. Such evaluation might strengthen Geo's advantage over SBM while potentially making it competitive at lower MPLs. For Hydro, the assessment might change the comparative results in their favour; however, this remains to be seen, especially when previously neglected limitations are also considered.

Firstly, recycled phosphate fertiliser from sewage sludge treatment should be assessed instead of the virgin resource. As aforementioned in section 4.1.2, using phosphate and sulphur directly from wastewater streams showed little environmental benefit. As an alternative, ash recovery from fluidized bed mono incineration, the most established phosphor recycling method from sewage sludge, could be considered. Yet, compared to primary production from phosphate rock, this method showed little environmental benefit besides reducing the raw material use (Smol et al., 2020), which must not be neglected given the mineral's finite supply.

Other stationary solutions could be included in a comparative analysis, most of which seem less energy intense in their approach compared to phosphorous recovery through mono incineration. Such methods are, for instance, the RSR treatment technique (RSR-Verfahren; Green Sentinel, 2022), thermal pressure hydrolysis (TerraNova Energy, 2022) or hydrothermal gasification (TreaTech, 2022). Little is known about the environmental impacts of these novel phosphate recycling methods. Such an analysis would thus potentially close a gap in the literature while giving further insight into the feasibility of MP production. As phosphor fertiliser was a hotspot in various ICs for both SBM and the MP reference flows, future scenario analysis should be conducted for both product systems.

To further reduce Bio's impacts, other inputs such as wood pellets could be considered. Using biowaste, a good supplier of renewable energy, instead of solid biomass, which is facing supply security, particularly in Europe (Bagherian et al., 2021), could be a viable solution to decrease Bio's overall impacts. However, it must be critically evaluated if such an application would cause competition with direct material recovery such as composting. From a circular economy approach, material recovery should be favoured over incineration. However, if biomass gasification, currently at TRL 5-6 (Al-Qahtani et al., 2021), is used instead of combustion, this could potentially combine energy and material recovery. Such a process generates CO2, heat, and electricity, which could be used for MP production while also generating solid carbon, which could increase soil fertility and permanently store carbon in the ground while keeping phosphate in the biosphere. Regardless, the overall performance of this novel method must be carefully considered, as biomass gasification is very energy intensive due to its endothermic reaction (Ahmad et al., 2016). To further reduce Bio's impacts, a reduction of baseload energy towards a mix of intermittent energy sources could also be considered.

Another hot spot identified during MP's CA was steel used for windmills and deep-well drilling for geothermal power facilities. As the steel sector is gradually expected to be decarbonised through green H2, this progression should be evaluated. Decarbonising the steel industry through renewable H2 might decrease overall CO2 emissions and fossil use while potentially shifting problems towards other ICs. Furthermore, a scenario analysis could also include the use of recycled steel. Lastly, basing the FU on nutritional rather than protein content might generate more accurate comparative results, as proteins vary in nutritional value. It remains to be seen if including these recommendations would change the comparative results.

9 Bibliography

Aalbers, R., & Bollen, J. (2017). *Biomass Energy with Carbon Capture and Storage can reduce costs of EU's Energy Roadmap with 15-75%: Background document to CPB Policy Brief 2017/02*. CPB Netherlands Bureau for Economic Policy Analysis. https://www.cpb.nl/sites/default/files/omnidownload/CPB-achtergronddocument-Biomass-Energy-with-Carbon-Capture-and-Storage-can-reduce-costs_0.pdf

Afrouzi, H. N., Pamodha Wimalaratna, Y., Ahmed, J., Mehranzamir, K., Chuin Liew, S., Wooi, C.-L., & Mobin Siddiquea, B. (2021). A Comprehensive Review on Available/Existing Renewable Energy Systems in Malaysia and Comparison of Their Capability of Electricity Generation in Malaysia. In L.-S. Wang, W. Cao, & S.-B. Hu (Eds.), *Entropy and Exergy in Renewable Energy*. IntechOpen. https://doi.org/10.5772/intechopen.96586

Agora. (2021). *Making renewable hydrogen cost-competitive: Policy instruments for supporting green H2*. 96. https://www.agora-energiewende.de/en/publications/making-renewable-hydrogen-cost-competitive/

Ahlgren, S., Björklund, A., Ekman, A., Karlsson, H., Berlin, J., Börjesson, P., Ekvall, T., Finnveden, G., Janssen, M., & Strid, I. (2015). Review of methodological choices in LCA of biorefinery systems - key issues and recommendations. *Biofuels, Bioproducts and Biorefining*, *9*(5), 606–619. https://doi.org/10.1002/bbb.1563

Ahmad, A. A., Zawawi, N. A., Kasim, F. H., Inayat, A., & Khasri, A. (2016). Assessing the gasification performance of biomass: A review on biomass gasification process conditions, optimization and economic evaluation. *Renewable and Sustainable Energy Reviews*, *53*, 1333–1347. https://doi.org/10.1016/j.rser.2015.09.030

Al-Qahtani, A., Parkinson, B., Hellgardt, K., Shah, N., & Guillen-Gosalbez, G. (2021). Uncovering the true cost of hydrogen production routes using life cycle monetisation. *Applied Energy*, *281*, 115958. https://doi.org/10.1016/j.apenergy.2020.115958

Amin, A. M., Croiset, E., & Epling, W. (2011). Review of methane catalytic cracking for hydrogen production. *International Journal of Hydrogen Energy*, *36*(4), 2904–2935. https://doi.org/10.1016/j.ijhydene.2010.11.035

Anderson, L., Blanco, H., Creech, C., Hergert, G., Nielsen, R., Maharjan, B., & Cupit, C. (2018). *Using High-Carbon Char as a Soil Amendment to Improve Soil Properties*. Accessed: 25.06.2022. https://cropwatch.unl.edu/2018/using-high-c-char-soil-amendment-improve-soil-properties#:~:text=Carbon-enriched Soil Amendments&text=Soil carbon is lost from,soil productivity (Figure 1).

Aneke, M., Agnew, B., & Underwood, C. (2011). Performance analysis of the Chena binary geothermal power plant. *Applied Thermal Engineering*, *31*(10), 1825–1832. https://doi.org/10.1016/j.applthermaleng.2011.02.028

Arehart, J. H., Hart, J., Pomponi, F., & D'Amico, B. (2021). Carbon sequestration and storage in the built environment. *Sustainable Production and Consumption*, *27*, 1047–1063. https://doi.org/10.1016/j.spc.2021.02.028

Arvidsson, R., Kushnir, D., Sandén, B. A., & Molander, S. (2014). Prospective Life Cycle Assessment of Graphene Production by Ultrasonication and Chemical Reduction. *Environmental Science & Technology*, *48*(8), 4529–4536. https://doi.org/10.1021/es405338k

Arvidsson, R., Tillman, A., Sandén, B. A., Janssen, M., Nordelöf, A., Kushnir, D., & Molander, S. (2017). Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. *Journal of Industrial Ecology*, *22*(6), 1286–1294. https://doi.org/10.1111/jiec.12690

Arvidsson, R., Tillman, A., Sandén, B. A., Janssen, M., Nordelöf, A., Kushnir, D., & Molander, S. (2018). Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. *Journal of Industrial Ecology*, *22*(6), 1286–1294. https://doi.org/10.1111/jiec.12690

Bagherian, M. A., Mehranzamir, K., Rezania, S., Abdul-Malek, Z., Pour, A. B., & Alizadeh, S. M. (2021). Analyzing Utilization of Biomass in Combined Heat and Power and Combined Cooling, Heating, and Power Systems. *Processes*, *1002*(9), 1–29. https://doi.org/10.3390/pr9061002

Banaszkiewicz, T. (2011). Nutritional Value of Soybean Meal. In H. El-Shemy (Ed.), *Soybean and Nutrition*. IntechOpen. https://doi.org/10.5772/23306

Banja, M., & Jégard, M. (2017). *Renewable technologies in the EU electricity sector: trends and projections: Analysis in the framework of the EU 2030 climate and energy strategy (JRC Science for Policy Report)*. (Report No. EUR 28897 EN). Publications Office of the European Union. https://doi.org/10.2760/43833

Bergerson, J. A., Brandt, A., Cresko, J., Carbajales-Dale, M., MacLean, H. L., Matthews, H. S., McCoy, S., McManus, M., Miller, S. A., Morrow, W. R., Posen, I. D., Seager, T., Skone, T., & Sleep, S. (2020). Life cycle assessment of emerging technologies: Evaluation techniques at different stages of market and technical maturity. *Journal of Industrial Ecology*, *24*(1), 11–25. https://doi.org/10.1111/jiec.12954

Bhavanam, A., & Sastry, R. C. (2011). Biomass Gasification Processes in Downd raft Fixed Bed Reactors: A Review. *International Journal of Chemical Engineering and Applications*, *2*(6), 425–433. https://doi.org/10.7763/IJCEA.2011.V2.146

Bodirsky, B. L., Rolinski, S., Biewald, A., Weindl, I., Popp, A., & Lotze-Campen, H. (2015). Global Food Demand Scenarios for the 21st Century. *PLOS ONE*, *10*(11), e0139201. https://doi.org/10.1371/journal.pone.0139201

Boero, A. J., Kardux, K., Kovaleva, M., Salas, D. A., Mooijer, J., Mashruk, S., Townsend, M., Rouwenhorst, K., Valera-Medina, A., & Ramirez, A. D. (2021). Environmental Life Cycle Assessment of Ammonia-Based Electricity. *Energies*, *14*(20), 6721. https://doi.org/10.3390/en14206721

Börjeson, L., Höjer, M., Dreborg, K.-H., Ekvall, T., & Finnveden, G. (2006). Scenario types and techniques: Towards a user's guide. *Futures*, *38*(7), 723–739. https://doi.org/10.1016/j.futures.2005.12.002

Botelho, A., Ferreira, P., Lima, F., Pinto, L. M. C., & Sousa, S. (2017). Assessment of the environmental impacts associated with hydropower. *Renewable and Sustainable Energy Reviews*, *70*, 896–904. https://doi.org/10.1016/J.RSER.2016.11.271

Campbell, B. M., Beare, D. J., Bennett, E. M., Hall-Spencer, J. M., Ingram, J. S. I., Jaramillo, F., Ortiz, R., Ramankutty, N., Sayer, J. A., & Shindell, D. (2017). Agriculture production as a major driver of the Earth system exceeding planetary boundaries. *Ecology and Society*, *22*(4), 8. https://doi.org/10.5751/ES-09595-220408

Caniëls, M. C. J., & Romijn, H. A. (2008). Actor networks in Strategic Niche Management: Insights from social network theory. *Futures*, *40*(7), 613–629. https://doi.org/10.1016/j.futures.2007.12.005

Carmo, M., Fritz, D. L., Mergel, J., & Stolten, D. (2013). A comprehensive review on PEM water electrolysis. *International Journal of Hydrogen Energy*, *38*(12), 4901–4934. https://doi.org/10.1016/J.IJHYDENE.2013.01.151

Centi, G., & Perathoner, S. (2011). CO2-based energy vectors for the storage of solar energy. *Greenhouse Gases: Science and Technology*, *1*(2011–03), 21–35. https://doi.org/10.1002/ghg3.3

Chaudhary, A., & Kastner, T. (2016). Land use biodiversity impacts embodied in international food trade. *Global Environmental Change*, *38*, 195–204. https://doi.org/10.1016/j.gloenvcha.2016.03.013

Cohen, A., Malone, S., Morris, Z., Weissburg, M., & Bras, B. (2018). Combined Fish and Lettuce Cultivation: An Aquaponics Life Cycle Assessment. *Procedia CIRP*, *69*, 551–556. https://doi.org/10.1016/j.procir.2017.11.029

Cooper, D. R., & Gutowski, T. G. (2020). Prospective Environmental Analyses of Emerging Technology: A Critique, a Proposed Methodology, and a Case Study on Incremental Sheet Forming. *Journal of Industrial Ecology*, *24*(1), 38–51. https://doi.org/10.1111/jiec.12748

Cox, P. A., Davis, D. A., Mash, D. C., Metcalf, J. S., & Banack, S. A. (2016). Dietary exposure to an environmental toxin triggers neurofibrillary tangles and amyloid deposits in the brain. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1823), 20152397. https://doi.org/10.1098/rspb.2015.2397

Crab, R., Avnimelech, Y., Defoirdt, T., Bossier, P., & Verstraete, W. (2007). Nitrogen removal techniques in aquaculture for a sustainable production. *Aquaculture*, *270*(1–4), 1–14. https://doi.org/10.1016/j.aquaculture.2007.05.006

Crenna, E., Secchi, M., Benini, L., & Sala, S. (2019). Global environmental impacts: data sources and methodological choices for calculating normalization factors for LCA. *The International Journal of Life Cycle Assessment*, *24*(10), 1851–1877. https://doi.org/10.1007/s11367-019-01604-y

Cucurachi, S., Blanco, C. F., Steubing, B., & Heijungs, R. (2021). Implementation of uncertainty analysis and moment-independent global sensitivity analysis for full-scale life cycle assessment models. *Journal of Industrial Ecology*, *26*(2), 374–391. https://doi.org/10.1111/jiec.13194

Cucurachi, S., Sala, S., Laurent, A., & Heijungs, R. (2014). Building and Characterizing Regional and Global Emission Inventories of Toxic Pollutants. *Environmental Science & Technology*, *48*(10), 5674–5682. https://doi.org/10.1021/es405798x

Cucurachi, S., Steubing, B., Siebler, F., Navarre, N., Caldeira, C., & Sala, S. (2021). *Draft prospective LCA methodology for Bio-based Products – Interim Report*. JRC European Commission.

Cucurachi, S., Van Der Giesen, C., & Guinée, J. (2018). Ex-ante LCA of Emerging Technologies. *Procedia CIRP*, *69*, 463–468. https://doi.org/10.1016/j.procir.2017.11.005

Cumberlege, T., Blenkinsopp, T., & Clark, J. (2016). *Assessment of environmental footprint of FeedKind protein*. (Report No. n.a.) Carbon Trust.

de Haes, H. A. U., & Heijungs, R. (2009). Analysis of physical interactions between the economy and the environment. In J. J. Boersema & L. Reijnders (Eds.), *Principles of Environmental Sciences* (pp. 207–237). Springer. https://doi.org/10.1007/978-1-4020-9158-2_12

del Pilar Jiménez-Donaire, M., Tarquis, A., & Vicente Giráldez, J. (2020). Evaluation of a combined drought indicator and its potential for agricultural drought prediction in southern Spain. *Natural Hazards and Earth System Sciences*, *20*, 21–33. https://doi.org/10.5194/nhess-20-21-2020

Delpierre, M. (2019). An ex-ante LCA study on wind-based hydrogen production in the Netherlands. *TU Delft Technology, Policy and Management*, *August*. https://repository.tudelft.nl/islandora/object/uuid%3Afc89a24e-9ef8-49cf-b5fb-3f3fa1e3051b

Delpierre, M., Quist, J., Mertens, J., Prieur-Vernat, A., & Cucurachi, S. (2021). Assessing the environmental impacts of wind-based hydrogen production in the Netherlands using ex-ante LCA and scenarios analysis. *Journal of Cleaner Production*, *299*, 126866. https://doi.org/10.1016/j.jclepro.2021.126866

Deng, Y., Li, J., Qiu, M., Yang, F., Zhang, J., & Yuan, C. (2017). Deriving characterization factors on freshwater ecotoxicity of graphene oxide nanomaterial for life cycle impact assessment. *The International Journal of Life Cycle Assessment*, *22*(2), 222–236. https://doi.org/10.1007/s11367-016-1151-4

Deutz, S., & Bardow, A. (2021). Life-cycle assessment of an industrial direct air capture process based on temperature–vacuum swing adsorption. *Nature Energy*, *6*(2), 203–213. https://doi.org/10.1038/s41560-020-00771-9

Dickson, M. H., & Fanelli, M. (2003). *Geothermal Energy: Utilization and Technology*. Routledge.

Dotan, H., Landman, A., Sheehan, S. W., Malviya, K. D., Shter, G. E., Grave, D. A., Arzi, Z., Yehudai, N., Halabi, M., Gal, N., Hadari, N., Cohen, C., Rothschild, A., & Grader, G. S. (2019). Decoupled hydrogen and oxygen evolution by a two-step electrochemical–chemical cycle for efficient overall water splitting. *Nature Energy*, *4*(9), 786–795. https://doi.org/10.1038/s41560-019-0462-7

Duffield, W. A., & Sass, J. H. (2003). *Geothermal energy - Clean power from the Earth's heat* (Issue 1249). (Report No. C1249). U.S. Department of the Interior, U.S. Geological Survey. https://pubs.usgs.gov/circ/2004/c1249/c1249.pdf

EASAC. (2018). *Negative emission technologies: What role in meeting Paris Agreement targets?* (Report No. EASAC policy report 35). European Academies Science Advisory Council. https://easac.eu/fileadmin/PDF_s/reports_statements/Negative_Carbon/EASAC_Report_on_Negative_Emission_Technologies.pdf

ecoinvent version 3.7. (2020). *Ecoinvent database*. Last Accessed: 20.07.2022. https://ecoinvent.org/the-ecoinvent-database/

Edrisi, B. H., & Michaelides, E. E. (2013). Effect of the working fluid on the optimum work of binary-flashing geothermal power plants. *Energy*, *50*, 389–394. https://doi.org/10.1016/j.energy.2012.10.025

Eide, M. ., Homleid, J. ., & Mattsson, B. (2003). Life cycle assessment (LCA) of cleaning-in-place processes in dairies. *LWT - Food Science and Technology*, *36*(3), 303–314. https://doi.org/10.1016/S0023-6438(02)00211-6

Ezeh, A. C., Bongaarts, J., & Mberu, B. (2012). Global population trends and policy options. *The Lancet*, *380*(9837), 142–148. https://doi.org/10.1016/S0140-6736(12)60696-5

Fantke, P., Aurisano, N., Bare, J., Backhaus, T., Bulle, C., Chapman, P. M., De Zwart, D., Dwyer, R., Ernstoff, A., Golsteijn, L., Holmquist, H., Jolliet, O., McKone, T. E., Owsianiak, M., Peijnenburg, W., Posthuma, L., Roos, S., Saouter, E., Schowanek, D., … Hauschild, M. (2018). Toward harmonizing ecotoxicity characterization in life cycle impact assessment. *Environmental Toxicology and Chemistry*, *37*(12), 2955–2971. https://doi.org/10.1002/etc.4261

Fantke, P., Aylward, L., Bare, J., Chiu, W. A., Dodson, R., Dwyer, R., Ernstoff, A., Howard, B., Jantunen, M., Jolliet, O., Judson, R., Kirchhübel, N., Li, D., Miller, A., Paoli, G., Price, P., Rhomberg, L., Shen, B., Shin, H.-M., … McKone, T. E. (2018). Advancements in Life Cycle Human Exposure and Toxicity Characterization. *Environmental Health Perspectives*, *126*(12), 125001. https://doi.org/10.1289/EHP3871

Fasihi, M., Efimova, O., & Breyer, C. (2019). Techno-economic assessment of CO2 direct air capture plants. *Journal of Cleaner Production*, *224*, 957–980. https://doi.org/10.1016/j.jclepro.2019.03.086

Frischknecht, R., Fantke, P., Tschümperlin, L., Niero, M., Antón, A., Bare, J., Boulay, A.-M., Cherubini, F., Hauschild, M. Z., Henderson, A., Levasseur, A., McKone, T. E., Michelsen, O., i Canals, L. M., Pfister, S., Ridoutt, B., Rosenbaum, R. K., Verones, F., Vigon, B., & Jolliet, O. (2016). Global guidance on environmental life cycle impact assessment indicators: progress and case study. *The International Journal of Life Cycle Assessment*, *21*(3), 429–442. https://doi.org/10.1007/s11367-015-1025-1

Frischknecht, R., & Jolliet, O. (2016). *Global Guidance for Life Cycle Impact Assessment Indicators Volume 1. UNEP/SETAC Life Cycle Initiative*. United Nations Environment Programme.

Gabriel, A., Victor, N., & C, P. J. (2014). Review Article Cactus pear biomass , a potential lignocellulose raw material for Single Cell Protein production (SCP): A Review. *Int.J.Curr.Microbiol.App.Sci*, *3*(7), 171–197. http://www.ijcmas.com

Gavankar, S., Suh, S., & Keller, A. A. (2014). The Role of Scale and Technology Maturity in Life Cycle Assessment of Emerging Technologies: A Case Study on Carbon Nanotubes. *Journal of Industrial Ecology*, *19*(1), 51–60. https://doi.org/10.1111/jiec.12175

Geels, F. W. (2002). Technological transitions as evolutionary reconfiguration processes: a multi-level perspective and a case-study. *Research Policy*, *31*(8–9), 1257–1274. https://doi.org/10.1016/S0048-7333(02)00062-8

Geels, F. W., & Schot, J. (2007). Typology of sociotechnical transition pathways. *Research Policy*, *36*(3), 399–417. https://doi.org/10.1016/j.respol.2007.01.003

Goldberg, I. (1985). *Single Cell Protein* (Vol. 1). Springer. https://doi.org/10.1007/978-3-642-46540-6

Government of Iceland. (2016). *Ministry of Industries and Innovation*. Accessed: 22.01.2022. https://www.government.is/topics/business-and-industry/energy/

Guillet, N., & Millet, P. (2015). Alkaline Water Electrolysis. In A. Godula-Jopek (Ed.), *Hydrogen Production: Electrolysis* (pp. 117–166). Wiley-VCH. https://doi.org/https://doi.org/10.1002/9783527676507.ch4

Guinée, J. (2001). Handbook on life cycle assessment — operational guide to the ISO standards. *The International Journal of Life Cycle Assessment*, *6*(5), 255. https://doi.org/10.1007/BF02978784

Guinée, J., Cucurachi, S., Henriksson, P. J. G., & Heijungs, R. (2018). *Digesting the alphabet soup of LCA*. *23*(7), 1507–1511. https://doi.org/10.1007/s11367-018-1478-0

Guinée, J., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., van Oers, L., Wegener Sleeswijk, A., Suh, S., Udo de Haes, H. A., de Bruijn, H., van Duin, R., & Huijbregts, M. A. J. (2002). *Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards*. Kluwer Academic Publishers. https://doi.org/https://doi.org/10.1007/BF02978897

H2PRO. (2022). *H2Pro announces and celebrates laying the cornerstone of its first production facility, capable of producing 600MW/year of Green Hydrogen systems*. Accessed: 22.04.2022. https://medium.com/@press_H2PRO/h2pro-announces-and-celebrates-laying-the-cornerstone-of-its-first-production-facility-capable-of-8cd0460a5a5e

Hammond, G. P., & Seth, S. M. (2013). Carbon and environmental footprinting of global biofuel production. *Applied Energy*, *112*, 547–559. https://doi.org/10.1016/j.apenergy.2013.01.009

Han, Y., Wang, J., Zhao, Z., Chen, J., Lu, H., & Liu, G. (2017). Fishmeal Application Induces Antibiotic Resistance Gene Propagation in Mariculture Sediment. *Environmental Science & Technology*, *51*(18), 10850–10860. https://doi.org/10.1021/acs.est.7b02875

Hasan, M. R. (2001). Nutrition and Feeding for Sustainable Aquaculture Development in the Third Millennium. In R. P. Subasinghe, P. Bueno, M. J. Phillips, C. Hough, S. E. McGladdery, & J. R. Arthur (Eds.), *Aquaculture in the Third Millennium. Technical Proceedings of the Conference on Aquaculture in the Third Millennium* (pp. 193–219). NACA, Bangkok and FAO, Rome. https://www.researchgate.net/profile/Rohana-Subasinghe/publication/266249350_Nutrition_and_Feeding_for_Sustainable_Aquaculture_Development_in_the_Third_Millennium_Nutrition_and_feeding_for_sustainable_aquaculture_development_in_the_third_millennium_In/lin

Hauschild, M., Goedkoop, M., Guinee, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Pennington, D., Pant, R., Sala, S., Brandao, M., & Wolf, M. (2011). *Recommendations for Life Cycle Impact Assessment in the European context - based on existing environmental impact assessment models and factors (International Reference Life Cycle Data System - ILCD handbook)*. EUR 24571 EN. Publications Office of the European Union. https://doi.org/10.2788/33030

Hertwich, E. G. (2013). Addressing biogenic greenhouse gas emissions from hydropower in LCA. *Environmental Science and Technology*, *47*(17), 9604–9611. https://doi.org/10.1021/es401820p

Hirooka, M. (2006). *Innovation Dynamism and Economic Growth*. Edward Elgar Publishing. https://doi.org/10.4337/9781845428860

Hofstede, G. (2005). *Cultures and organizations: software of the mind: intercultural cooperation and its importance for survival.* Mc-Graw Hill. http://homecont.ro/pitagora/Hofstede-4-dimensiuni.pdf

Hoogma, R., Kemp, R., Schot, J., & Truffer, B. (2002). *Experimenting for Sustainable Transport: The Approach of Strategic Niche Management* (1st ed.). Routledge. https://doi.org/10.4324/9780203994061

Hou, P., Jolliet, O., Zhu, J., & Xu, M. (2020). Estimate ecotoxicity characterization factors for chemicals in life cycle assessment using machine learning models. *Environment International*, *135*, 105393. https://doi.org/10.1016/j.envint.2019.105393

Hu, X., Kerckhof, F. M., Ghesquière, J., Bernaerts, K., Boeckx, P., Clauwaert, P., & Boon, N. (2020). Microbial Protein out of Thin Air: Fixation of Nitrogen Gas by an Autotrophic Hydrogen-Oxidizing Bacterial Enrichment. *Environmental Science and Technology*, *54*(6), 3609–3617. https://doi.org/10.1021/acs.est.9b06755

Hulst, M. K., Huijbregts, M. A. J., Loon, N., Theelen, M., Kootstra, L., Bergesen, J. D., & Hauck, M. (2020). A systematic approach to assess the environmental impact of emerging technologies: A case study for the GHG footprint of CIGS solar photovoltaic laminate. *Journal of Industrial Ecology*, *24*(6), 1234–1249. https://doi.org/10.1111/jiec.13027

IEA. (2021). *Ammonia Technology Roadmap: Towards more Sustainable Nitrogen Fertilizer Production*. (Report No. CC BY-NC 3.0 IGO) International Energy Agency. https://www.iea.org/reports/ammonia-technology-roadmap

International Renewable Energy Agency. (2022). *Bioenergy Data: Installed Capacity Trends*. Accessed: 05.04.2022. https://www.irena.org/bioenergy

Järviö, N., Maljanen, N.-L., Kobayashi, Y., Ryynänen, T., & Tuomisto, H. L. (2021). An attributional life cycle assessment of microbial protein production: a case study on using hydrogen-oxidizing bacteria. *Science of The Total Environment*, *776*, 145764. https://doi.org/10.1016/j.scitotenv.2021.145764

Jones, S. W., Karpol, A., Friedman, S., Maru, B. T., & Tracy, B. P. (2020). Recent advances in single cell protein use as a feed ingredient in aquaculture. *Current Opinion in Biotechnology*, *61*, 189–197. https://doi.org/10.1016/j.copbio.2019.12.026

Kamp, L. M., Smits, R. E. H. M., & Andriesse, C. D. (2004). Notions on learning applied to wind turbine development in the Netherlands and Denmark. *Energy Policy*, *32*(14), 1625–1637. https://doi.org/10.1016/S0301-4215(03)00134-4

Kamp, L. M., & Vanheule, L. F. I. (2015). Review of the small wind turbine sector in Kenya: Status and bottlenecks for growth. *Renewable and Sustainable Energy Reviews*, *49*, 470–480. https://doi.org/10.1016/j.rser.2015.04.082

Kang, D., Palmer, C., Mannini, D., Rahimi, N., Gordon, M. J., Metiu, H., & McFarland, E. W. (2020). Catalytic Methane Pyrolysis in Molten Alkali Chloride Salts Containing Iron. *ACS Catalysis*, *10*(13), 7032–7042. https://doi.org/10.1021/acscatal.0c01262

Kato, T., Kubota, M., Kobayashi, N., & Suzuoki, Y. (2005). Effective utilization of by-product oxygen from electrolysis hydrogen production. *Energy*, *30*(14), 2580–2595. https://doi.org/10.1016/j.energy.2004.07.004

Kemper, J. (2015). Biomass and carbon dioxide capture and storage: A review. *International Journal of Greenhouse Gas Control*, *40*, 401–430. https://doi.org/10.1016/J.IJGGC.2015.06.012

Kim, J. Y., Han, K., Ahn, C. K., Lee, M. S., Rhee, C. H., & Chun, H. D. (2013). Operating Cost for CO2 Capture Process Using Aqueous Ammonia. *Energy Procedia*, *37*, 677–682.

https://doi.org/10.1016/j.egypro.2013.05.156

Koj, J., Wulf, C., Schreiber, A., & Zapp, P. (2017). Site-Dependent Environmental Impacts of Industrial Hydrogen Production by Alkaline Water Electrolysis. *Energies*, *10*(7), 860. https://doi.org/10.3390/en10070860

Koponen, K., & Soimakallio, S. (2015). Foregone carbon sequestration due to land occupation—the case of agro-bioenergy in Finland. *The International Journal of Life Cycle Assessment*, *20*(11), 1544–1556. https://doi.org/10.1007/s11367-015-0956-x

Kramer, G. J., & Haigh, M. (2009). No quick switch to low-carbon energy. *Nature*, *462*(7273), 568–569. https://doi.org/10.1038/462568a

Kupferschmidt, K. (2015). Feature: Why insects could be the ideal animal feed. *Science*. https://doi.org/10.1126/science.aad4709

Lassaletta, L., Billen, G., Grizzetti, B., Anglade, J., & Garnier, J. (2014). 50 year trends in nitrogen use efficiency of world cropping systems: the relationship between yield and nitrogen input to cropland. *Environmental Research Letters*, *9*(10), 105011. https://doi.org/10.1088/1748-9326/9/10/105011

Lee, B. H. (2015). *Fundamentals of Food Biotechnology, 2nd Edition*. John Wiley & Sons. https://www.wiley.com/en-us/Fundamentals+of+Food+Biotechnology%2C+2nd+Edition-p-9781118384954

Little, D. C., Newton, R. W., & Beveridge, M. C. M. (2016). Aquaculture: a rapidly growing and significant source of sustainable food? Status, transitions and potential. *Proceedings of the Nutrition Society*, *75*(3), 274–286. https://doi.org/10.1017/S0029665116000665

Liu, C., Colón, B. C., Ziesack, M., Silver, P. A., & Nocera, D. G. (2016). Water splitting–biosynthetic system with CO 2 reduction efficiencies exceeding photosynthesis. *Science*, *352*(6290), 1210–1213. https://doi.org/10.1126/science.aaf5039

Machhammer, O., Bode, A., & Hormuth, W. (2016). Ökonomisch/ökologische Betrachtung zur Herstellung von Wasserstoff in Großanlagen. *Chemie Ingenieur Technik*, *87*(4), 409–418. https://doi.org/10.1002/cite.201400151

Malico, I., Nepomuceno Pereira, R., Gonçalves, A. C., & Sousa, A. M. O. (2019). Current status and future perspectives for energy production from solid biomass in the European industry. *Renewable and Sustainable Energy Reviews*, *112*, 960–977. https://doi.org/10.1016/j.rser.2019.06.022

Matassa, S., Boon, N., Pikaar, I., & Verstraete, W. (2016). Microbial protein: future sustainable food supply route with low environmental footprint. *Microbial Biotechnology*, *9*(5), 568–575. https://doi.org/10.1111/1751-7915.12369

Matassa, S., Boon, N., & Verstraete, W. (2015). Resource recovery from used water: The manufacturing abilities of hydrogen-oxidizing bacteria. *Water Research*, *68*, 467–478. https://doi.org/10.1016/j.watres.2014.10.028

McKone, T. E., Nazaroff, W. W., Berck, P., Auffhammer, M., Lipman, T., Torn, M. S., Masanet, E., Lobscheid, A., Santero, N., Mishra, U., Barrett, A., Bomberg, M., Fingerman, K., Scown, C., Strogen, B., & Horvath, A. (2011). Grand Challenges for Life-Cycle Assessment of Biofuels. *Environmental Science & Technology*, *45*(5), 1751–1756. https://doi.org/10.1021/es103579c

Meadowcroft, J. (2009). What about the politics? Sustainable development, transition management, and long term energy transitions. *Policy Sciences*, *42*(4), 323–340. https://doi.org/10.1007/s11077-009-9097-z

Millet, P. (2015). PEM Water Electrolysis. In A. Godula-Jopek (Ed.), *Hydrogen Production: Electrolysis* (pp. 63–116). Wiley-VCH. https://doi.org/doi.org/10.1002/9783527676507.ch3

Muñoz, I., & Schmidt, J. H. (2016). Methane oxidation, biogenic carbon, and the IPCC's emission metrics. Proposal for a consistent greenhouse-gas accounting. *International Journal of Life Cycle Assessment*, *21*(8), 1069–1075. https://doi.org/10.1007/s11367-016-1091-z

Muradov, N. Z., & Veziroğlu, T. N. (2005). From hydrocarbon to hydrogen–carbon to hydrogen economy. *International Journal of Hydrogen Energy*, *30*(3), 225–237. https://doi.org/10.1016/j.ijhydene.2004.03.033

Nappa, M., Lienemann, M., Tossi, C., Blomberg, P., Jäntti, J., Tittonen, I. J., & Penttilä, M. (2020). Solar-Powered Carbon Fixation for Food and Feed Production Using Microorganisms - A Comparative Techno-Economic Analysis. *ACS Omega*, *5*(51), 33242–33252. https://doi.org/10.1021/acsomega.0c04926

National Research Council and National Academy of Engineering. (2004). *The Hydrogen Economy: Opportunities, Costs, Barriers, and R&D Needs (2004)*. National Academies Press. https://doi.org/10.17226/10922

Nicita, A., Maggio, G., Andaloro, A. P. F., & Squadrito, G. (2020). Green hydrogen as feedstock: Financial analysis of a photovoltaic-powered electrolysis plant. *International Journal of Hydrogen Energy*, *45*(20), 11395–11408. https://doi.org/10.1016/j.ijhydene.2020.02.062

Njakou Djomo, S., Knudsen, M. T., Parajuli, R., Andersen, M. S., Ambye-Jensen, M., Jungmeier, G., Gabrielle, B., & Hermansen, J. E. (2017). Solving the multifunctionality dilemma in biorefineries with a novel hybrid mass–energy allocation method. *Global Change Biology- Bioenergy*, *9*(11), 1674–1686. https://doi.org/10.1111/gcbb.12461

Notarnicola, B., Sala, S., Anton, A., McLaren, S. J., Saouter, E., & Sonesson, U. (2017). The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges. *Journal of Cleaner Production*, *140*, 399–409. https://doi.org/10.1016/J.JCLEPRO.2016.06.071

Nykvist, N., & Rosén, K. (1985). Effect of clear-felling and slash removal on the acidity of Northern coniferous soils. *Forest Ecology and Management*, *11*(3), 157–169. https://doi.org/10.1016/0378-1127(85)90024-6

Ochsner, A. M., Sonntag, F., Buchhaupt, M., Schrader, J., & Vorholt, J. A. (2015). Methylobacterium extorquens: methylotrophy and biotechnological applications. *Applied Microbiology and Biotechnology*, *99*(2), 517–534. https://doi.org/10.1007/s00253-014-6240-3

Olofsson, J., & Börjesson, P. (2018). Residual biomass as resource – Life-cycle environmental impact of wastes in circular resource systems. *Journal of Cleaner Production*, *196*, 997–1006. https://doi.org/10.1016/j.jclepro.2018.06.115

Park, C., Bigler, F., & Korba, P. (2016). Power-to-Gas Concept for Integration of Increased Photovoltaic Generation into the Distribution. *Energy Procedia*, *99*, 411–417.

https://doi.org/10.1016/j.egypro.2016.10.131

Parkinson, B., Tabatabaei, M., Upham, D. C., Ballinger, B., Greig, C., Smart, S., & McFarland, E. (2018). Hydrogen production using methane: Techno-economics of decarbonizing fuels and chemicals. *International Journal of Hydrogen Energy*, *43*(5), 2540–2555. https://doi.org/10.1016/j.ijhydene.2017.12.081

Parodi, A., Leip, A., De Boer, I. J. M., Slegers, P. M., Ziegler, F., Temme, E. H. M., Herrero, M., Tuomisto, H., Valin, H., Van Middelaar, C. E., Van Loon, J. J. A., & Van Zanten, H. H. E. (2018). The potential of future foods for sustainable and healthy diets. *Nature Sustainability*, *1*(12), 782–789. https://doi.org/10.1038/s41893-018-0189-7

Pesonen, H.-L., Ekvall, T., Fleischer, G., Huppes, G., Jahn, C., Klos, Z. S., Rebitzer, G., Sonnemann, G. W., Tintinelli, A., Weidema, B. P., & Wenzel, H. (2000). Framework for scenario development in LCA. *The International Journal of Life Cycle Assessment*, *5*(1), 21. https://doi.org/10.1007/BF02978555

Peters, M., Köhler, B., Kuckshinrichs, W., Leitner, W., Markewitz, P., & Müller, T. E. (2011). Chemical Technologies for Exploiting and Recycling Carbon Dioxide into the Value Chain. *ChemSusChem*, *4*(9), 1216–1240. https://doi.org/10.1002/cssc.201000447

Phillips, E. (2022). Nanotechnological interventions in biofuel production. In *Handbook of Biofuels* (pp. 593–604). Elsevier. https://doi.org/10.1016/B978-0-12-822810-4.00031-2

Pikaar, I., Matassa, S., Bodirsky, B. L., Weindl, I., Humpenöder, F., Rabaey, K., Boon, N., Bruschi, M., Yuan, Z., van Zanten, H., Herrero, M., Verstraete, W., & Popp, A. (2018). Decoupling Livestock from Land Use through Industrial Feed Production Pathways. *Environmental Science & Technology*, *52*(13), 7351–7359. https://doi.org/10.1021/acs.est.8b00216

Poore, J., & Nemecek, T. (2018). Reducing food’s environmental impacts through producers and consumers. *Science*, *360*(6392), 987–992. https://doi.org/10.1126/science.aaq0216

Quadrelli, E. A., & Centi, G. (2011). Green carbon dioxide. *ChemSusChem*, *4*(9), 1179–1181. https://doi.org/10.1002/cssc.201100518

Quadrelli, E. A., Centi, G., Duplan, J. L., & Perathoner, S. (2011). Carbon dioxide recycling: Emerging large-scale technologies with industrial potential. *ChemSusChem*, *4*(9), 1194–1215. https://doi.org/10.1002/cssc.201100473

Quist, J. (2013). Backcasting and Scenarios for Sustainable Technology Development. In *Handbook of Sustainable Engineering* (pp. 749–771). Springer. https://doi.org/10.1007/978-1-4020-8939-8_52

Rass-Hansen, J., Falsig, H., Jørgensen, B., & Christensen, C. H. (2007). Bioethanol: fuel or feedstock? *Journal of Chemical Technology & Biotechnology*, *82*(4), 329–333. https://doi.org/10.1002/jctb.1665

Rausch, B., Symes, M. D., Chisholm, G., & Cronin, L. (2014). Decoupled catalytic hydrogen evolution from a molecular metal oxide redox mediator in water splitting. *Science*, *345*(6202), 1326–1330. https://doi.org/10.1126/science.1257443

Raven, R. P. J. M. (2005). *Strategic niche management for biomass: a comparative study on the experimental introduction of bioenergy technologies in the Netherlands and Denmark* [[Phd Thesis 1 (Research TU/e / Graduation TU/e), Industrial Engineering and Innovation Sciences]. Technische Universiteit Eindhoven]. https://doi.org/10.6100/IR590593

Ravikumar, D., Sinha, P., Seager, T. P., & Fraser, M. P. (2016). An anticipatory approach to quantify energetics of recycling CdTe photovoltaic systems. *Progress in Photovoltaics: Research and Applications*, *24*(5), 735–746. https://doi.org/10.1002/pip.2711

Ricardo Energy & Environment. (2020). *Analysing the potential of bioenergy with carbon capture in the UK to 2050: Summary for policymakers*. (Report No. TRN 1700/11/2018). https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/911268/potential-of-bioenergy-with-carbon-capture.pdf

Rischer, H., Szilvay, G. R., & Oksman-Caldentey, K.-M. (2020). Cellular agriculture — industrial biotechnology for food and materials. *Current Opinion in Biotechnology*, *61*, 128–134. https://doi.org/10.1016/j.copbio.2019.12.003

Ritala, A., Häkkinen, S. T., Toivari, M., & Wiebe, M. G. (2017). Single Cell Protein—State-of-the-Art, Industrial Landscape and Patents 2001–2016. *Frontiers in Microbiology*, *8*. https://doi.org/10.3389/fmicb.2017.02009

Ritchey, T. (2011). General Morphological Analysis (GMA). In *Wicked Problems – Social Messes* (pp. 7–18). Springer. https://doi.org/10.1007/978-3-642-19653-9_2

Romero, M., & Steinfeld, A. (2012). Concentrating solar thermal power and thermochemical fuels. *Energy & Environmental Science*, *5*(11), 9234. https://doi.org/10.1039/c2ee21275g

Romijn, H., Raven, R., & de Visser, I. (2010). Biomass energy experiments in rural India: Insights from learning-based development approaches and lessons for Strategic Niche Management. *Environmental Science & Policy*, *13*(4), 326–338. https://doi.org/10.1016/j.envsci.2010.03.006

Rosenbaum, R. K., Bachmann, T. M., Gold, L. S., Huijbregts, M. A. J., Jolliet, O., Juraske, R., Koehler, A., Larsen, H. F., MacLeod, M., Margni, M., McKone, T. E., Payet, J., Schuhmacher, M., van de Meent, D., & Hauschild, M. Z. (2008). USEtox—the UNEP-SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in life cycle impact assessment. *The International Journal of Life Cycle Assessment*, *13*(7), 532–546. https://doi.org/10.1007/s11367-008-0038-4

Rudiyanto, B., Illah, I., Pambudi, N. A., Cheng, C.-C., Adiprana, R., Imran, M., Huat Saw, L., & Handogo, R. (2017). Preliminary analysis of dry-steam geothermal power plant by employing exergy assessment: Case study in Kamojang geothermal power plant, Indonesia. *Case Studies in Thermal Engineering*, *10*, 292–301. https://doi.org/10.1016/j.csite.2017.07.006

Saarinen, M., Fogelholm, M., Tahvonen, R., & Kurppa, S. (2017). Taking nutrition into account within the life cycle assessment of food products. *Journal of Cleaner Production*, *149*, 828–844. https://doi.org/10.1016/j.jclepro.2017.02.062

Sala, S., Crenna, E., Secchi, M., & Pant, R. (2017). *Global normalisation factors for the Environmental Footprint and Life Cycle Assessment (JRC Technical Reports)*. (Report No. EUR 28984 EN). Publications Office of the European Union. https://doi.org/10.2760/775013

SalmonBusiness. (2019). *Biotech firm announces that it can produce its single cell protein aquafeed ingredient at a "dramatic" reduction in cost*. Accessed: 10.01.2022. https://salmonbusiness.com/biotech-firm-announces-that-it-can-produce-its-single-cell-protein-aquafeed-ingredient-at-a-dramatic-reduction-in-cost/

Sánchez-Bastardo, N., Schlögl, R., & Ruland, H. (2021). Methane Pyrolysis for Zero-Emission Hydrogen Production: A Potential Bridge Technology from Fossil Fuels to a Renewable and Sustainable Hydrogen Economy. *Industrial & Engineering Chemistry Research*, *60*(32), 11855–11881. https://doi.org/10.1021/acs.iecr.1c01679

Saouter, E., Biganzoli, F., Ceriani, L., Versteeg, D., Crenna, E., Zampori, L., Sala, S., & Pant, R. (2018). *Environmental footprint Update of Life Cycle Impact Assessment Methods : ecotoxicity freshwater, human toxicity cancer, and non-cancer* (EUR 29495). Publications Office of the European Union. https://doi.org/10.2760/611799

Schakel, W., Meerman, H., Talaei, A., Ramírez, A., & Faaij, A. (2014). Comparative life cycle assessment of biomass co-firing plants with carbon capture and storage. *Applied Energy*, *131*, 441–467. https://doi.org/10.1016/j.apenergy.2014.06.045

Schot, J., & Geels, F. W. (2008). Strategic niche management and sustainable innovation journeys: theory, findings, research agenda, and policy. *Technology Analysis & Strategic Management*, *20*(5), 537–554. https://doi.org/10.1080/09537320802292651

Sikkema, R., Dallemand, J. F., Matos, C. T., van der Velde, M., & San-Miguel-Ayanz, J. (2017). How can the ambitious goals for the EU's future bioeconomy be supported by sustainable and efficient wood sourcing practices? *Scandinavian Journal of Forest Research*, *32*(7), 551–558. https://doi.org/10.1080/02827581.2016.1240228

Sillman, J., Nygren, L., Kahiluoto, H., Ruuskanen, V., Tamminen, A., Bajamundi, C., Nappa, M., Wuokko, M., Lindh, T., Vainikka, P., Pitkänen, J.-P., & Ahola, J. (2019). Bacterial protein for food and feed generated via renewable energy and direct air capture of CO2: Can it reduce land and water use? *Global Food Security*, *22*, 25–32. https://doi.org/10.1016/j.gfs.2019.09.007

Sillman, J., Uusitalo, V., Ruuskanen, V., Ojala, L., Kahiluoto, H., Soukka, R., & Ahola, J. (2020). A life cycle environmental sustainability analysis of microbial protein production via power-to-food approaches. *The International Journal of Life Cycle Assessment*, *25*(11), 2190–2203. https://doi.org/10.1007/s11367-020-01771-3

Simbeck, D. R. (2004). CO2 capture and storage—the essential bridge to the hydrogen economy. *Energy*, *29*(9–10), 1633–1641. https://doi.org/10.1016/j.energy.2004.03.065

Smetana, S., Schmitt, E., & Mathys, A. (2019). Sustainable use of Hermetia illucens insect biomass for feed and food: Attributional and consequential life cycle assessment. *Resources, Conservation and Recycling*, *144*, 285–296. https://doi.org/10.1016/j.resconrec.2019.01.042

Smith Stegen, K. (2015). Heavy rare earths, permanent magnets, and renewable energies: An imminent crisis. *Energy Policy*, *79*, 1–8. https://doi.org/10.1016/J.ENPOL.2014.12.015

Smolinka, T., Ojong, E. T., & Garche, J. (2015). Chapter 8 - Hydrogen Production from Renewable Energies—Electrolyzer Technologies. In *Electrochemical Energy Storage for Renewable Sources and Grid Balancing* (pp. 103–128). Elsevier. https://doi.org/10.1016/B978-0-444-62616-5.00008-5

Sonesson, U., Davis, J., Hallström, E., & Woodhouse, A. (2019). Dietary-dependent nutrient quality indexes as a complementary functional unit in LCA: A feasible option? *Journal of Cleaner Production*, *211*, 620–627. https://doi.org/10.1016/j.jclepro.2018.11.171

Spath, P. L., & Mann, M. K. (2001). Life Cycle Assessment of Renewable Hydrogen Production via Wind/Electrolysis. *Milestone Completion Report by U.S National Renewable Energy Laboratory*, *560–35404*. https://www.nrel.gov/docs/fy04osti/35404.pdf

Spiller, M., Muys, M., Papini, G., Sakarika, M., Buyle, M., & Vlaeminck, S. E. (2020). Environmental impact of microbial protein from potato wastewater as feed ingredient: Comparative consequential life cycle assessment of three production systems and soybean meal. *Water Research*, *171*. https://doi.org/10.1016/j.watres.2019.115406

Spolaore, P., Joannis-Cassan, C., Duran, E., & Isambert, A. (2006). Commercial applications of microalgae. *Journal of Bioscience and Bioengineering*, *101*(2), 87–96. https://doi.org/10.1263/jbb.101.87

Squadrito, G., Nicita, A., & Maggio, G. (2021). A size-dependent financial evaluation of green hydrogen-oxygen co-production. *Renewable Energy*, *163*, 2165–2177. https://doi.org/10.1016/j.renene.2020.10.115

Statistics Finland. (2017). *Renewable energy sources produced 45 per cent of electricity and 57 per cent of heat*. Accessed: 04.06.2021. https://www.stat.fi/til/salatuo/2016/salatuo_2016_2017-11-02_tie_001_en.html

Steffen, W., Richardson, K., Rockström, J., Cornell, S. E., Fetzer, I., Bennett, E. M., Biggs, R., Carpenter, S. R., de Vries, W., de Wit, C. A., Folke, C., Gerten, D., Heinke, J., Mace, G. M., Persson, L. M., Ramanathan, V., Reyers, B., & Sörlin, S. (2015). Planetary boundaries: Guiding human development on a changing planet. *Science*, *347*(6223). https://doi.org/10.1126/science.1259855

Steubing, B., de Koning, D., Haas, A., & Mutel, C. L. (2020). The Activity Browser — An open source LCA software building on top of the brightway framework. *Software Impacts*, *3*, 100012. https://doi.org/10.1016/j.simpa.2019.100012

Surindra, M., Caesarendra, W., Prasetyo, T., Mahlia, T., & Taufik. (2019). Comparison of the Utilization of 110 °C and 120 °C Heat Sources in a Geothermal Energy System Using Organic Rankine Cycle (ORC) with R245fa, R123, and Mixed-Ratio Fluids as Working Fluids. *Processes*, *7*(2), 113. https://doi.org/10.3390/pr7020113

Szulc, P., Kasprzak, J., Dymaczewski, Z., & Kurczewski, P. (2021). Life Cycle Assessment of Municipal Wastewater Treatment Processes Regarding Energy Production from the Sludge Line. *Energies*, *14*(2), 356. https://doi.org/10.3390/en14020356

Tamers, M. (2006). *Distinguishing Between Bio-Ethanol and Petroleum Ethanol*. Ethanol Producer Magazine. Accessed: 03.06.2022. https://ethanolproducer.com/articles/2077/distinguishing-between-�bio-ethanol’-and-petroleum-ethanol

Tanzer, S. E., & Ramírez, A. (2019). When are negative emissions negative emissions? *Energy and Environmental Science*, *12*(4), 1210–1218. https://doi.org/10.1039/c8ee03338b

Tecchio, P., Freni, P., De Benedetti, B., & Fenouillot, F. (2016). Ex-ante Life Cycle Assessment approach developed for a case study on bio-based polybutylene succinate. *Journal of Cleaner Production*, *112*, 316–325. https://doi.org/10.1016/j.jclepro.2015.07.090

Technology Collaboration Programme. (2022). Fossil vs biogenic CO2 emissions. In *Accessed: 02.09.21*. IEA Bioenergy. https://www.ieabioenergy.com/iea-publications/faq/woodybiomass/biogenic-co2/

Terlouw, T., Treyer, K., Bauer, C., & Mazzotti, M. (2021). Life Cycle Assessment of Direct Air Carbon Capture and Storage with Low-Carbon Energy Sources. *Environmental Science & Technology*, *55*(16), 11397–11411. https://doi.org/10.1021/acs.est.1c03263

The Royal Society. (2018). *Options for producing low-carbon hydrogen at scale*. (Report No. DES4801_2). https://royalsociety.org/-/media/policy/projects/hydrogen-production/energy-briefing-green-hydrogen.pdf

Timmerberg, S., Kaltschmitt, M., & Finkbeiner, M. (2020). Hydrogen and hydrogen-derived fuels through methane decomposition of natural gas – GHG emissions and costs. *Energy Conversion and Management: X*, *7*, 100043. https://doi.org/10.1016/j.ecmx.2020.100043

Torella, J. P., Gagliardi, C. J., Chen, J. S., Bediako, D. K., Colón, B., Way, J. C., Silver, P. A., & Nocera, D. G. (2015). Efficient solar-to-fuels production from a hybrid microbial–water-splitting catalyst system. *Proceedings of the National Academy of Sciences*, *112*(8), 2337–2342. https://doi.org/10.1073/pnas.1424872112

Treyer, K. (2014). heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014 - GB - electricity, high voltage. *Ecoinvent 3.7 Dataset Documentation*, 9.

Treyer, K. (2019). Electricity production, hydro, reservoir, non-alpine region - FI. *Ecoinvent 3.7 Dataset Documentation*, 6.

Troell, M., Naylor, R. L., Metian, M., Beveridge, M., Tyedmers, P. H., Folke, C., Arrow, K. J., Barrett, S., Crépin, A.-S., Ehrlich, P. R., Gren, Å., Kautsky, N., Levin, S. A., Nyborg, K., Österblom, H., Polasky, S., Scheffer, M., Walker, B. H., Xepapadeas, T., & de Zeeuw, A. (2014). Does aquaculture add resilience to the global food system? *Proceedings of the National Academy of Sciences*, *111*(37), 13257–13263. https://doi.org/10.1073/pnas.1404067111

Trostle, R. (2008). Global Agricultural Supply and Demand: Factors Contributing to the Recent Increase in Food Commodity Prices. *A Report from the Economic Research Service; United States Department of Agriculture*, *WRS-0801*, 30. https://www.ers.usda.gov/webdocs/outlooks/40463/12274_wrs0801_1_.pdf?v=8663.7

Tsoy, N., Prado, V., Wypkema, A., Quist, J., & Mourad, M. (2019). Anticipatory Life Cycle Assessment of sol-gel derived anti-reflective coating for greenhouse glass. *Journal of Cleaner Production*, *221*, 365–376. https://doi.org/10.1016/j.jclepro.2019.02.246

Tsoy, N., Steubing, B., van der Giesen, C., & Guinée, J. (2020). Upscaling methods used in ex ante life cycle assessment of emerging technologies: a review. *International Journal of Life Cycle Assessment*, *25*(9), 1680–1692. https://doi.org/10.1007/s11367-020-01796-8

Tufvesson, L. M., Tufvesson, P., Woodley, J. M., & Börjesson, P. (2013). Life cycle assessment in green chemistry: overview of key parameters and methodological concerns. *The International Journal of Life Cycle Assessment*, *18*(2), 431–444. https://doi.org/10.1007/s11367-012-0500-1

Tuomisto, H. L. (2019). The eco-friendly burger. *EMBO Reports*, *20*(1). https://doi.org/10.15252/embr.201847395

Turkenburg, W., Arent, D. J., Bertani, R., Faaij, A., Hand, M., Krewitt, W., Larson, E. D., Lund, J., Mehos, M., Merrigan, T., Mitchell, C., Moreira, J. R., Sinke, W., Sonntag-O'Brien, V., Thresher, B., van Sark, W., Usher, E., Bilello, D., Chum, H., … Schmid, J. (2012). Renewable Energy. In T. B. Johansson, N. Nakicenovic, A. Patwardhan, & L. Gomez-

Echeverri (Eds.), *Global Energy Assessment (GEA)* (pp. 761–900). Cambridge University Press. https://doi.org/10.1017/CBO9780511793677.017

Turkenburg, W., Schöne, S., Metz, B., & Meyer, L. (2016). *De klimaatdoelstelling van Parijs: Wat Betekent Een Maximale Temperatuurstijging Van "ruim Beneden 2 Graden, En Streven Naar Anderhalve Graad"?.* https://www.uu.nl/sites/default/files/de_klimaatdoelstelling_van_parijs_-_wctssbmlm_-_15mrt2016.pdf

Udvardi, M., Brodie, E. L., Riley, W., Kaeppler, S., & Lynch, J. (2015). Impacts of Agricultural Nitrogen on the Environment and Strategies to Reduce these Impacts. *Procedia Environmental Sciences*, *29*, 303. https://doi.org/10.1016/j.proenv.2015.07.275

United Nations Environment Programme. (2011). *Global Guidance Principles for Life Cycle Assessment Databases: A Basis for Greener Processes and Products*. https://www.lifecycleinitiative.org/wp-content/uploads/2012/12/2011 - Global Guidance Principles.pdf

Ursua, A., Gandia, L. M., & Sanchis, P. (2012). Hydrogen Production From Water Electrolysis: Current Status and Future Trends. *Proceedings of the IEEE*, *100*(2), 410–426. https://doi.org/10.1109/JPROC.2011.2156750

van der Giesen, C., Cucurachi, S., Guinée, J., Kramer, G. J., & Tukker, A. (2020). A critical view on the current application of LCA for new technologies and recommendations for improved practice. *Journal of Cleaner Production*, *259*. https://doi.org/10.1016/j.jclepro.2020.120904

van der Laak, W. W. M., Raven, R. P. J. M., & Verbong, G. P. J. (2007). Strategic niche management for biofuels: Analysing past experiments for developing new biofuel policies. *Energy Policy*, *35*(6), 3213–3225. https://doi.org/10.1016/j.enpol.2006.11.009

van Loo, S., & Koppejan, J. (2007). *The handbook of combustion and co-firing biomass* (2nd ed.). Earthscan Ltd.

Verbong, G., & Geels, F. (2007). The ongoing energy transition: Lessons from a socio-technical, multi-level analysis of the Dutch electricity system (1960–2004). *Energy Policy*, *35*(2), 1025–1037. https://doi.org/10.1016/j.enpol.2006.02.010

Viebahn, P., Scholz, A., & Zelt, O. (2019). The Potential Role of Direct Air Capture in the German Energy Research Program—Results of a Multi-Dimensional Analysis. *Energies*, *12*(18), 3443. https://doi.org/10.3390/en12183443

Volova, T. G., & Barashkov, V. A. (2010). Characteristics of proteins synthesized by hydrogen-oxidizing microorganisms. *Applied Biochemistry and Microbiology*, *46*(6), 574–579. https://doi.org/10.1134/S0003683810060037

Von Der Assen, N., Jung, J., & Bardow, A. (2013). Life-cycle assessment of carbon dioxide capture and utilization: Avoiding the pitfalls. *Energy and Environmental Science*, *6*(9), 2721–2734. https://doi.org/10.1039/c3ee41151f

Wahlroos, M., Cross, S., & Syri, S. (2014). Prospects for biomass use in large power plants in the EU-27 and the role of combined heat and power production. *International Conference on the European Energy Market, EEM*, *May*. https://doi.org/10.1109/EEM.2014.6861250

Wallace, A. G., & Symes, M. D. (2018). Decoupling Strategies in Electrochemical Water Splitting and Beyond. *Joule*, *2*(8), 1390–1395. https://doi.org/10.1016/j.joule.2018.06.011

Weiss, M., Haufe, J., Carus, M., Brandão, M., Bringezu, S., Hermann, B., & Patel, M. K. (2012). A Review of the Environmental Impacts of Biobased Materials. *Journal of Industrial Ecology*, *16*, 169–181. https://doi.org/10.1111/j.1530-9290.2012.00468.x

Wernet, G., Bauer, C., Steubing, B., Reinhard, J., Moreno-Ruiz, E., & Weidema, B. (2016). The ecoinvent database version 3 (part I): overview and methodology. *The International Journal of Life Cycle Assessment*, *21*(9), 1218–1230. https://doi.org/10.1007/s11367-016-1087-8

WMO, (World Meteorological Organization). (2014). Scientific Assessment of Ozone Depletion: 2014. *World Meteorological Organization, Global Ozone Research and Monitoring Project-Report*, *55*, 416. https://csl.noaa.gov/assessments/ozone/2014/citations.html

Wolf, M.-A., Pant, R., Chomkhamsri, K., Sala, S., & Pennington, D. (2012). *The International Reference Life Cycle Data System (ILCD) Handbook: Towards more sustainable production and consumption for a resource-efficient Europe (JRC Reference Reports)*. (Report No. EUR 24982 EN). Publications Office of the European Union. https://doi.org/10.2788/85727

World Biogas Association. (2019). Global potential of biogas. In *World Biogas Association* (Issue June). https://www.worldbiogasassociation.org/wp-content/uploads/2019/09/WBA-execsummary-4ppa4_digital-Sept-2019.pdf

XE. (2022). *Currency Währungsrechner*. Accessed: 22.08.2022. https://www.xe.com/de/currencyconverter/convert/?Amount=180&From=GBP&To=EUR

Yu, J. (2014). Bio-based products from solar energy and carbon dioxide. *Trends in Biotechnology*, *32*(1), 5–10. https://doi.org/10.1016/j.tibtech.2013.11.001

Zampori, L., & Pant, R. (2019). Suggestions for updating the Product Environmental Footprint (PEF) method. In *JRC Technical Reports*. https://doi.org/10.2760/424613

10 Appendix 1

10.1 Calculations

10.1.1 Nutrient utilisation and electrolysis efficiency level

In the primary reference MP LCI (Järviö et al., 2021) used for this study, there are some inconsistencies regarding the FU. The publication refers to a FU of 1 kg of MP product before packing with a 5% moisture and 65% protein content at the factory gate. Yet, some LCI data (Järviö et al., 2021) relates to dried biomass with 0% moisture content. In some cases, it was unclear if the input data refers to 5% or 0% moisture content. This data mismatch thus needed evaluation, summarised in the following paragraph.

In table 2 (Järviö et al., 2021), a value of 14.13 kWh of electricity for the electrolyser is given for 79% efficiency. Yet, this efficiency value does not match table S9, where 13.45kWh produces 1kg MP with 5% moisture and 65% protein content, while table S9 does not refer to a nutrient utilisation level. However, as 13.45 is approximately 95% of 14.13, it was assumed that input data in table 2 refers to the production of dried biomass with 0% moisture content. Thus, tables 2 and S9 refer to the same scenario and, therefore, to the same efficiency levels while only differing in their moisture content. The H2, O2, and CO2 outputs, and CO2 input displayed in table S9, are consequently based on 79% electrolysis efficiency and 99% nutrient utilization. This evaluation confirmed that 1% of CO2 and H2 in table S9 leave the fermentation process without being utilised.

10.1.1.1 Nutrient utilisation level

Table S9 only provides the amount of water for the fermentation process without stating how much is needed for the electrolysis process. This amount required determination, as it defined the H_2 and O_2 input.

All the below values refer to a nutrient utilisation level of 99% to produce 1kg MP with 95%moisture content. For calculations regarding the lower nutrient utilisation levels are in appendix 2.

Amount of CO2

The amount of CO2 was given in table S9 (Järviö et al., 2021), which was 1.76kg CO2/kg MP at 99% nutrient efficiency level.

Amount of H2

According to figure 13A, 401 kg of water is needed for the electrolysis process to produce 160kg MP with 5% moisture content; therefore, 401/160 = 2.51kg is required for 1kg MP. As H2O is made up of 2 Hydrogen atoms (molar mass = 1g/mol) and 1 Oxygen atom (molar mass = 16g/mol), the molar mass of H2O (water) is 18g/mol.

Thus, 2.51kg of water is needed, of which 2 parts are Hydrogen and 16 are Oxygen. Dividing water input by the sum of the atomic ratio gives: 2.51kg/18g/mol = 139.44mol, of which H2 = 2g/mol*139.44mol = 0.28kg and 16g/mol*139.44mol = 2.23kg is O2. These values were assumed to be based on 99% nutrient efficiency, which seemed to be the default in the study's supplementary information (Järviö et al., 2021)

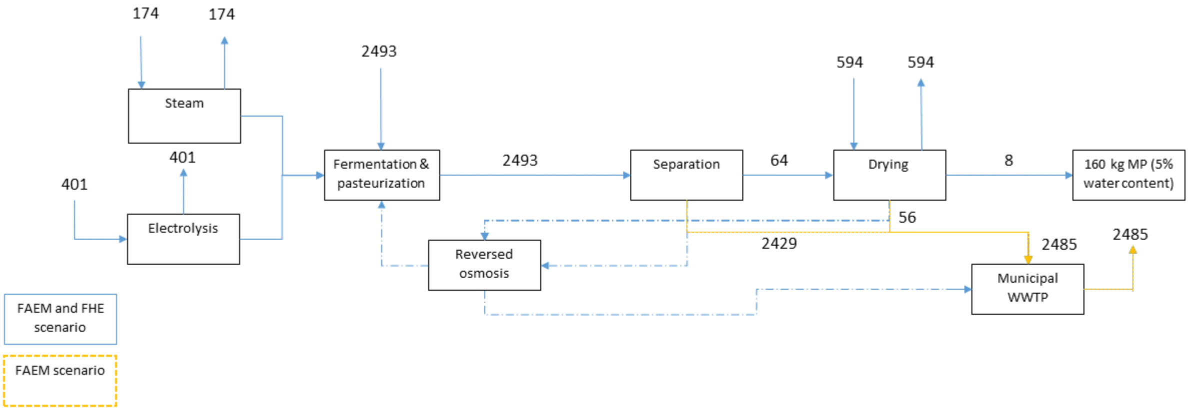


Figure 13A. Water flow chart in kg to produce 160kg MP (Järviö et al., 2021; SI 1)

Amount of O2 (cut off)

This paragraph determines the amount of O2 that could be utilised as a by-product of MP production. Table S9 (Järviö et al., 2021) shows that 1.39kg O2/kg MP leave the fermentation process unused, given a nutrient utilisation level of 99%. The origin of this stream comes from the water input into the electrolysis and the the CO2 input for the fermentation. Additionally, the O2 left over from the NH3 production was determined. As 24.04kg NH3 are needed to produce 160kg MP with 5% moisture content (Järviö et al., 2021), this gives an amount of 0.15kg NH3/kg MP. The molar mass of N is 14g/mol, thus NH3 has a molar of 17g/mol. Thus, the amount of H2 is: 0.15kg/17g/mol*3g/mol = 0.03kg H2. As water's molar mass is 18g/mol, with 2 parts H2 and 16 parts O2, 0.03kg/2*16 = 0.21kg O2. Therefore, a total of 1.39kg+0.21kg = 1.6kg O2 could theoretically be used as a by-product.

Amount of NH3

The NH3 was modelled as part of the nutrient solution. It was thus not directly scaled to match the input to the FU. The value of liquid NH3 was retrieved from supplementary information 2 of the reference LCI (Järviö et al., 2021).

10.1.1.2 Electrolysis efficiency level

The electrolysis efficiency is related to the H2 output, making it comparable to other publications in the field. On the other hand, the electrolysis efficiency level in table 2 (Järviö et al., 2021) refers to the FU. Based on table 2 (Järviö et al., 2021), 13.45kWh are needed to produce 1kg of MP at 79% efficiency. Thus, the ratio between 1kg MP and the required amount of H2 is 1kg MP/0.28kg H2 = 3.57. An efficiency of 79%, can therefore also be expressed as 3.57*13.45kWh/kg MP = 48.04kWh$_e$/kg H2. This value is in the range of future predictions for AEL's efficiency levels in 2050 (Delpierre et al., 2021).

An efficiency level of 60% or 18.6kWh/kg MP, shown in table 2 (Järviö et al., 2021), was also based on dry biomass with a moisture content of 0. Considering a 5% moisture level the efficiency is 0.95*18.6kWh/kg MP = 17.67kWh/kg MP; in relation to H2 output, this value is

3.57*17.67kWh/kg MP = 63.10kWh$_e$/kg H2, which is the value used in the pilot-scale scenario in 2020.

10.1.2 Allocation factors of multifunctional process

The ecoinvent database allocated the process "heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014" between heat and electricity respectively (Treyer, 2014). In reality, this is one co-production process, yet the database divided into two processes with emissions allocated accordingly. In this study, in addition to heat and electricity, the BCHP process also produced CO2. Thus, to solve the multifunctional process, firstly, the allocation factor between heat and electricity had to be determined.

The allocation factor for electricity and heat are given as 66.27 and 33.73, respectively (Treyer, 2014); thus, their ratio is 33.73/66.27 = 0.51. The relations between 1MJ and 1kWh is 1MJ=1000kWh/3600 = 0.28kWh. Out of a total efficiency level of 0.6 for the CHP process, electrical efficiency is 0.15, and thermal efficiency is 0.45 (Treyer, 2014); therefore, their ratio is 0.15/0.45 = 0.33. Thus, the allocation factor for 1MJ$_{th}$ in relation to 1kWh$_e$ = 1*0.28*0.33*0.51 = 0.05. This value is confirmed when compared to the flows to and from the co-production, such as the non-fossil CO2 output. For 1MJ$_{th}$, this output is 0.07kg, and for 1kWh$_e$, 1.53kg (Treyer, 2014), while the relation between the two values is 0.07/1.53 = 0.05, the same as the allocation factor calculated above.

The total amount of non-fossil CO2 captured per kWh$_e$ and MJ$_{th}$ was defined based on a 90% capturing rate (Schakel et al., 2014): (1.53kg + 0.07kg)*0.9= 1.44kg CO2.

10.1.2.1 Allocation factors based on a CO2 capturing price of 120€/t captured

Given a CO2 capturing price of 0.12€/kg (see section 4.2.4) and assuming an average cost of 0.117€/kWh$_e$, the ratio of the co-production process for heat, electricity, and CO2 was determined. The CO2 capturing price per kWh$_e$ and MJ$_{th}$ is 0.12€/kg*1.44kg = 0.17€. Thus the price ratio between the CO2 capturing price and the price of 1kWh$_e$ is 0.17/0.117 = 1.44. Therefore, a formula relating 1MJ$_{th}$ and 1.44kg CO2 to 1kWh$_e$, based on the above analysis is:

$0.05 * x + 1.00 * x + 1.44 * x = 1.00$

$\rightarrow x = 0.40$

And thus the allocation factors are:

→ **1MJ$_{th}$:** 0.40*0.05 = **0.02**

→ **1kWh$_e$:** 0.40*1.00 = **0.40**

→ **1.44kg CO2:** 0.40*1.44 = **0.58**

10.1.2.2 Allocation factors based on a CO2 capturing price of 280€/t captured

Given a price of 0.28€/kg, based on the highest price predictions of 240-325€/t CO2 captured (see section 4.2.4) and assuming an average cost of 0.117€/kWh$_e$, the ratio of the co-production process for heat, electricity, and CO2 was determined. The CO2 capturing price per kWh$_e$ and MJ$_{th}$ is 0.28€/kg*1.44kg = 0.40€. Thus the price ratio between the CO2 capturing price and the price of 1kWh$_e$ is 0.40/0.117 = 3.45. Therefore, a formula relating 1MJ$_{th}$ and 1.44kg CO2 to 1kWh$_e$, based on the above analysis is:

$0.05 * x + 1.00 * x + 3.45 * x = 1.00$

$\rightarrow x = 0.22$

And thus the allocation factors are:

→ **1MJ$_{th}$:** 0.22*0.05 = **0.01**

→ **1kWh$_e$:** 0.22*1.00 = **0.22**

→ **1.44kg CO2:** 0.22*3.45 = **0.77**

10.1.2.3 Allocation factors based on a CO2 capturing price of 25€/t captured

Given a price of 0.025€/kg, based on price predictions of 11-38€/t CO2 captured (see section 4.2.4) and assuming an average cost of 0.117€/kWh$_e$, the ratio of the co-production process for heat, electricity, and CO2 was determined. The CO2 capturing price per kWh$_e$ and MJ$_{th}$ is 0.025€/kg*1.44kg = 0.04€. Thus, the price ratio between the CO2 capturing price and the price of 1kWh$_e$ is 0.04/0.117 = 0.31. Therefore, a formula relating 1MJ$_{th}$ and 1.44kg CO2 to 1kWh$_e$, based on the above analysis is:

0.05*x + 1.00*x + 0.31*x = 1.00

→ x = 0.74

→ **1MJ$_{th}$:** 0.05*0.74 **= 0.03**

→ **1kWh$_e$:** 1.00*0.74 **= 0.74**

→ **1.01kg CO2:** 0.31*0.74 **= 0.23**

10.2 Sankey diagrams of CA for MP production in 2050

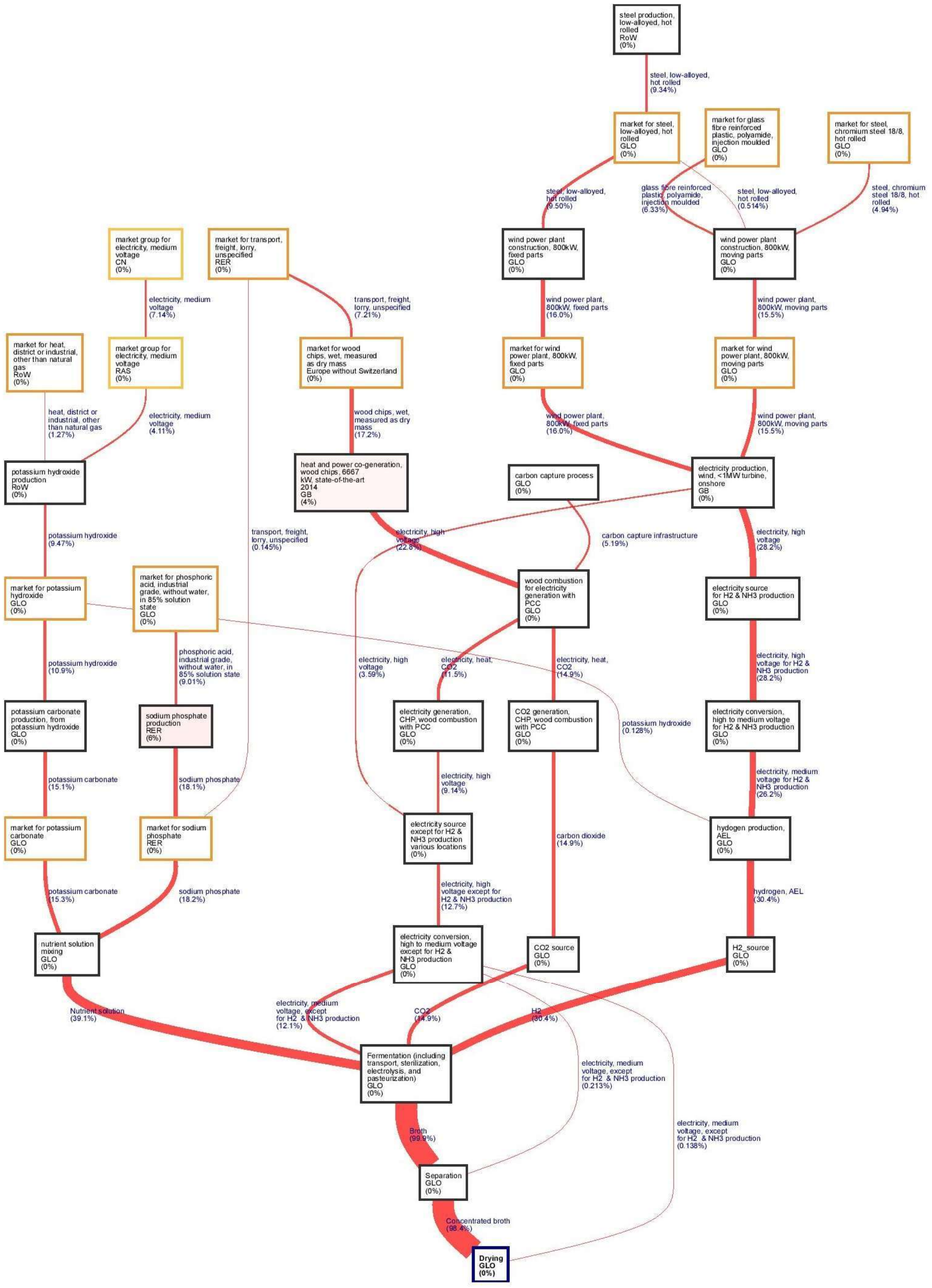


Figure 14A. CA Bio 2050, climate change

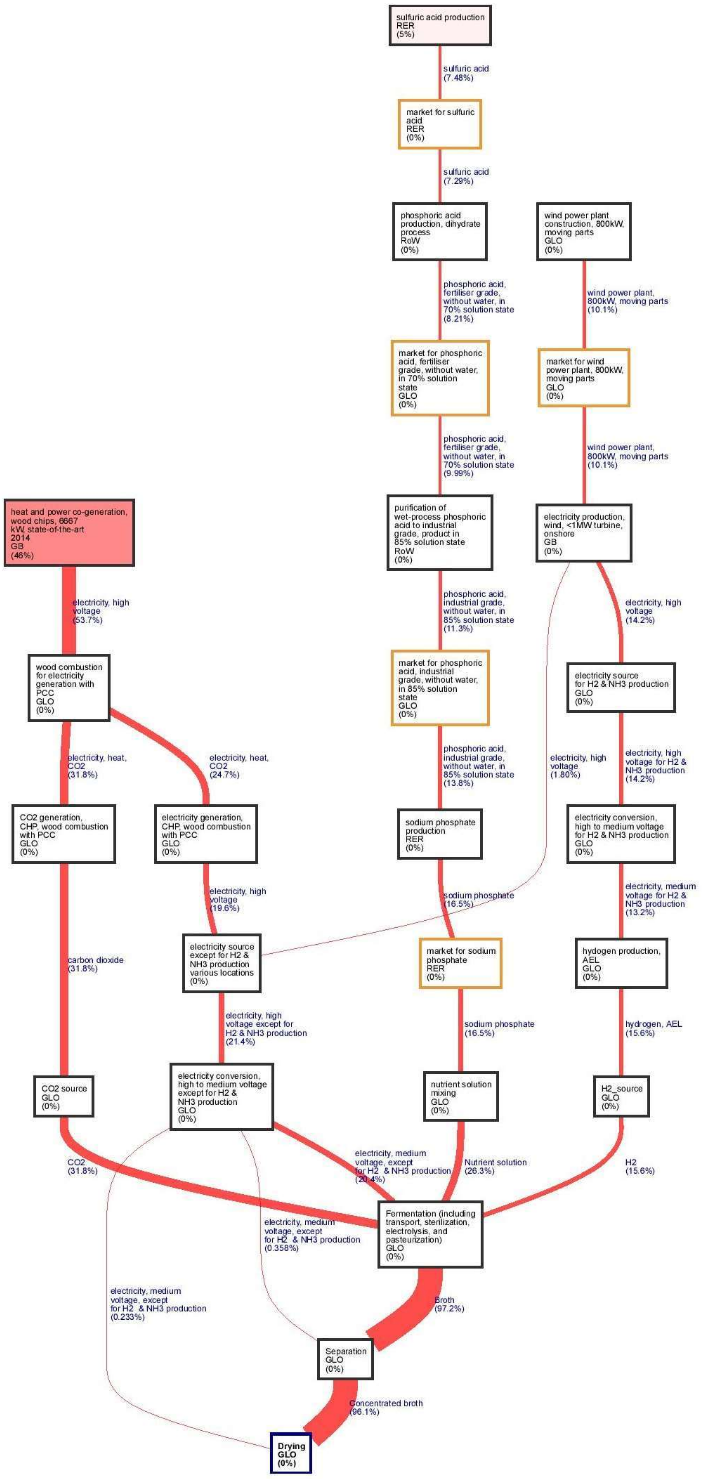


Figure 15A. CA Bio 2050, freshwater & terrestrial acidification

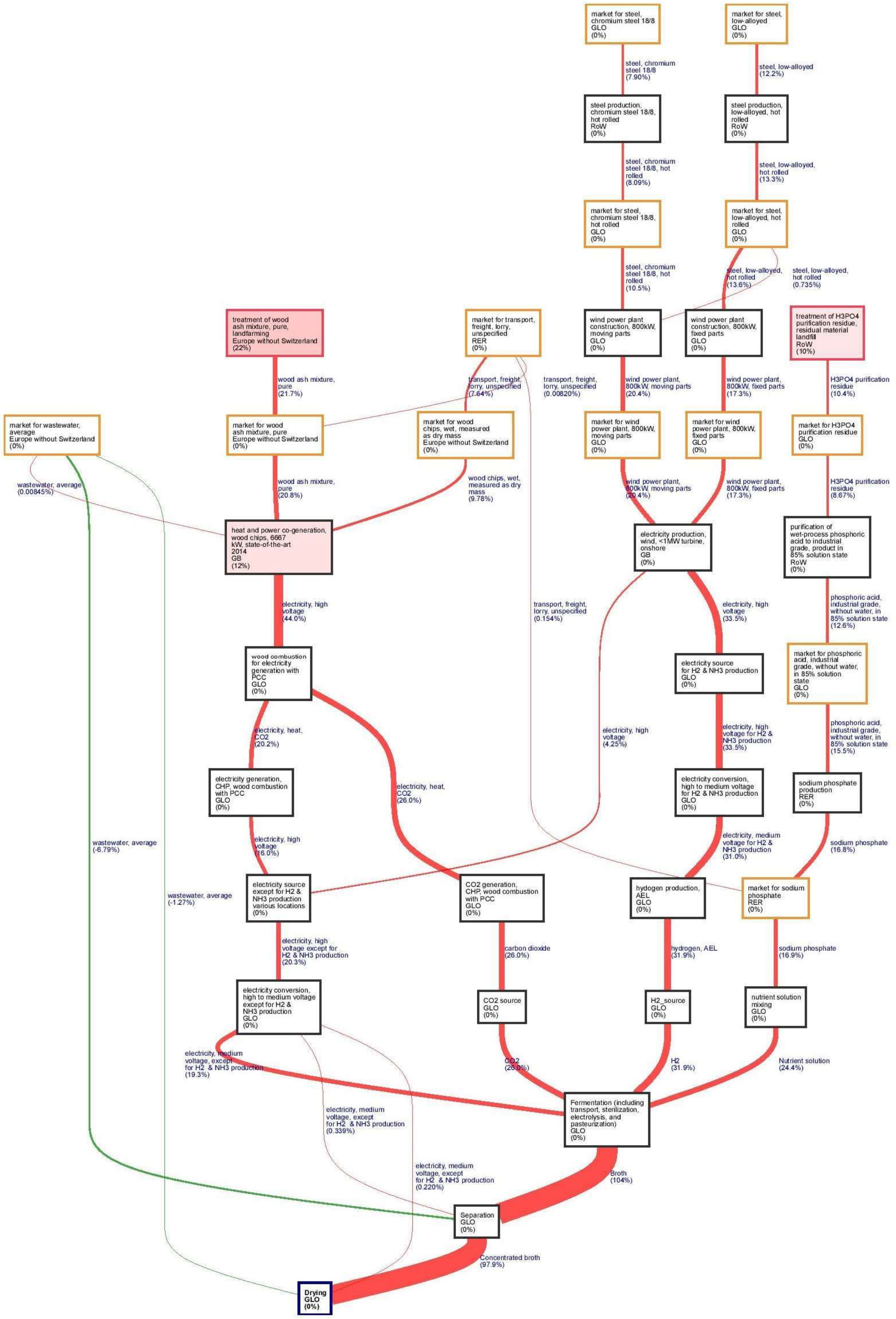


Figure 16A. CA Bio 2050, freshwater ecotoxicity

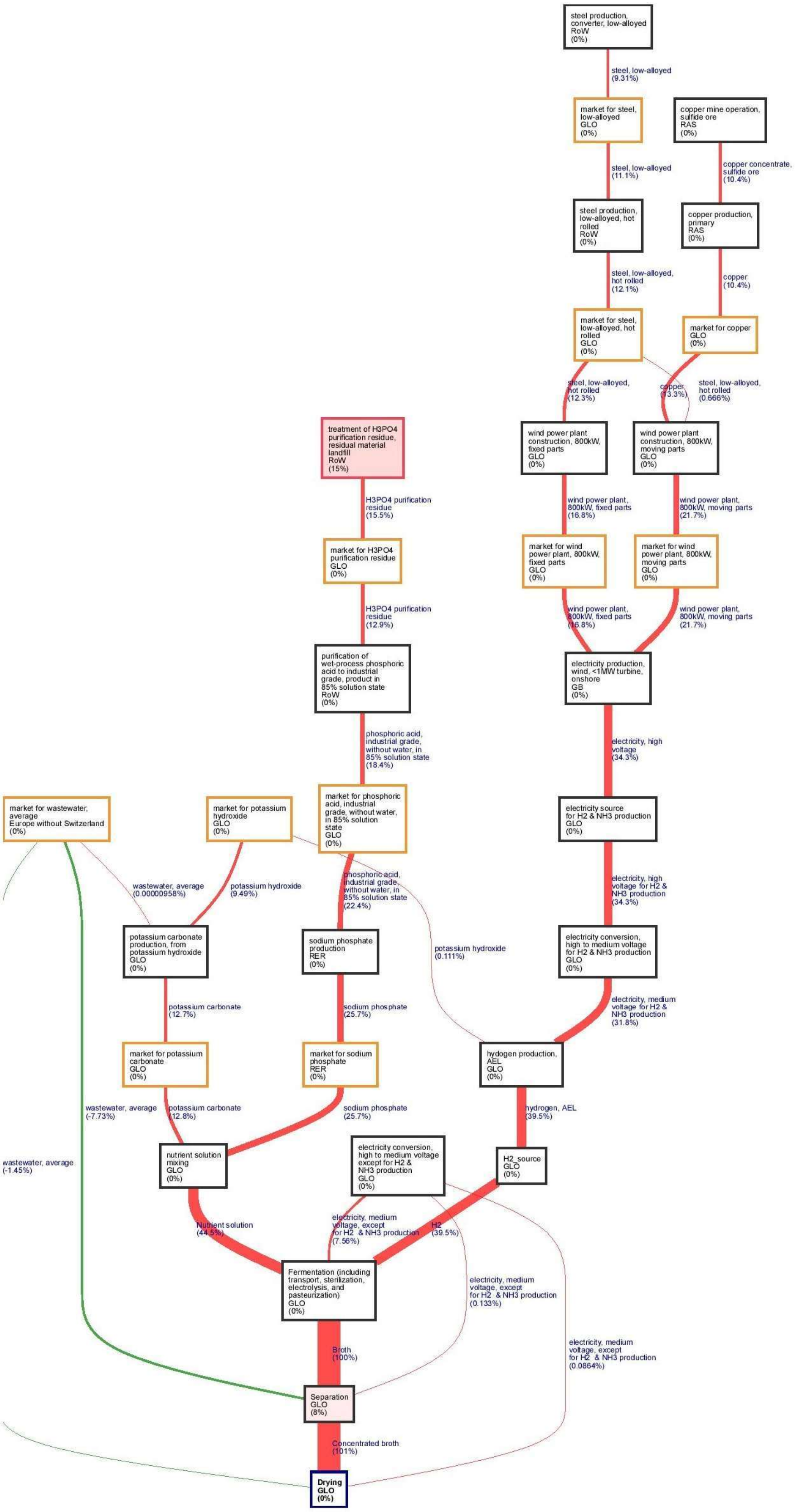


Figure 17A. CA Bio 2050, freshwater eutrophication

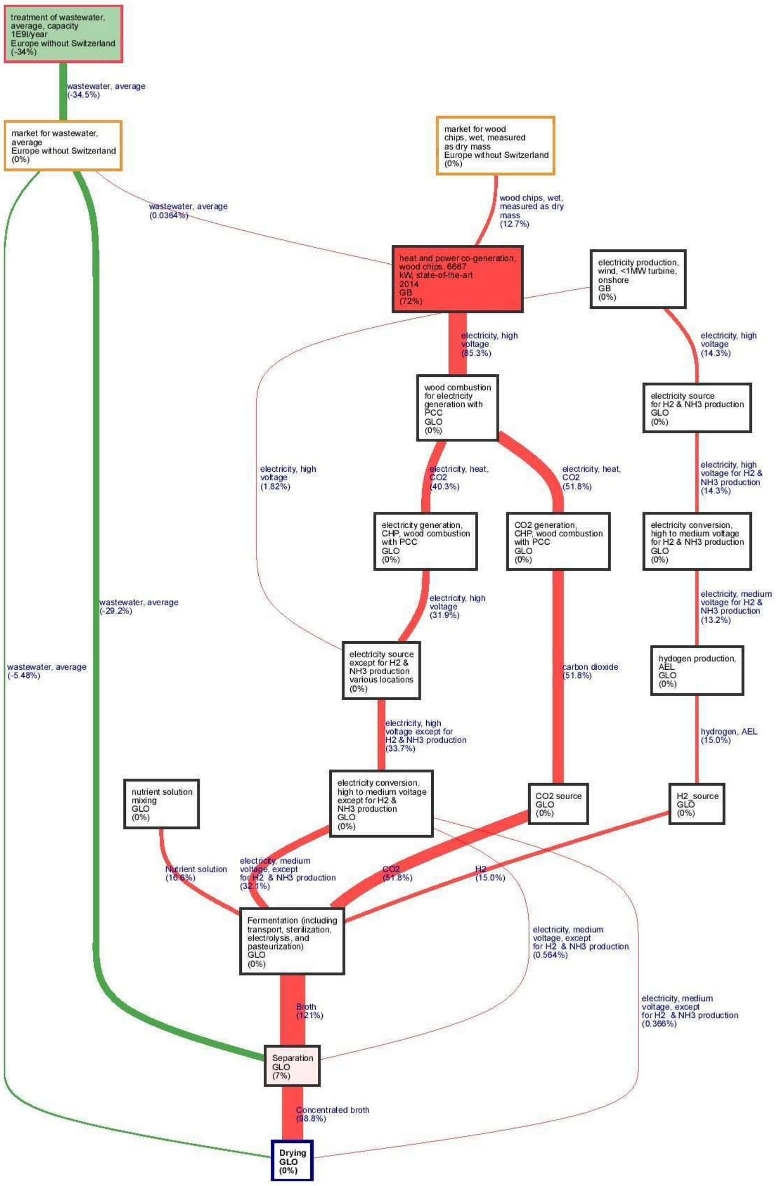


Figure 18A. CA Bio 2050, marine eutrophication

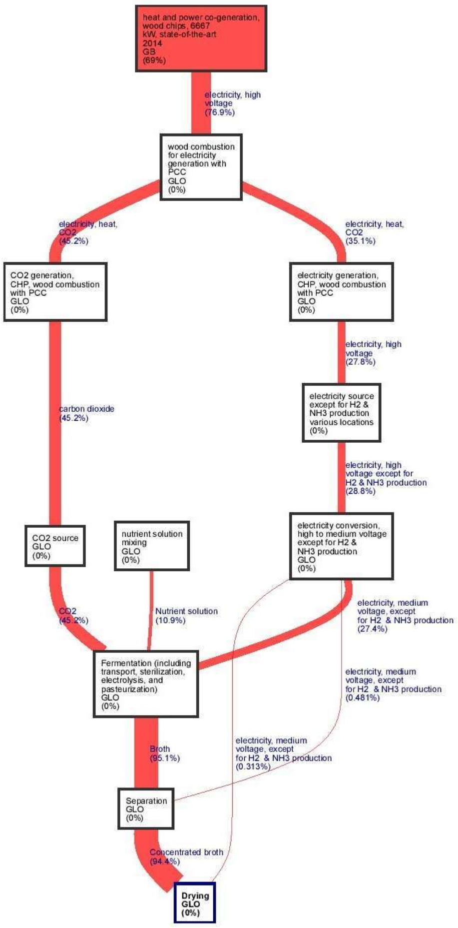


Figure 19A. CA Bio 2050, terrestrial eutrophication

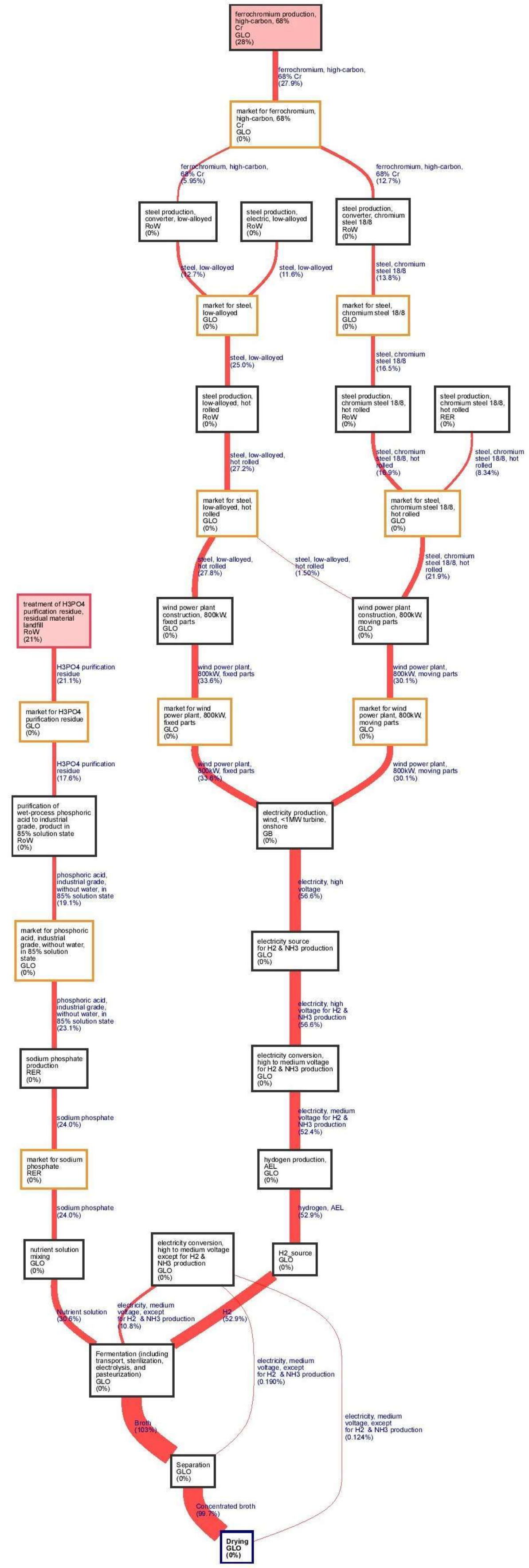


Figure 20A. CA Bio 2050, carcinogenic effects

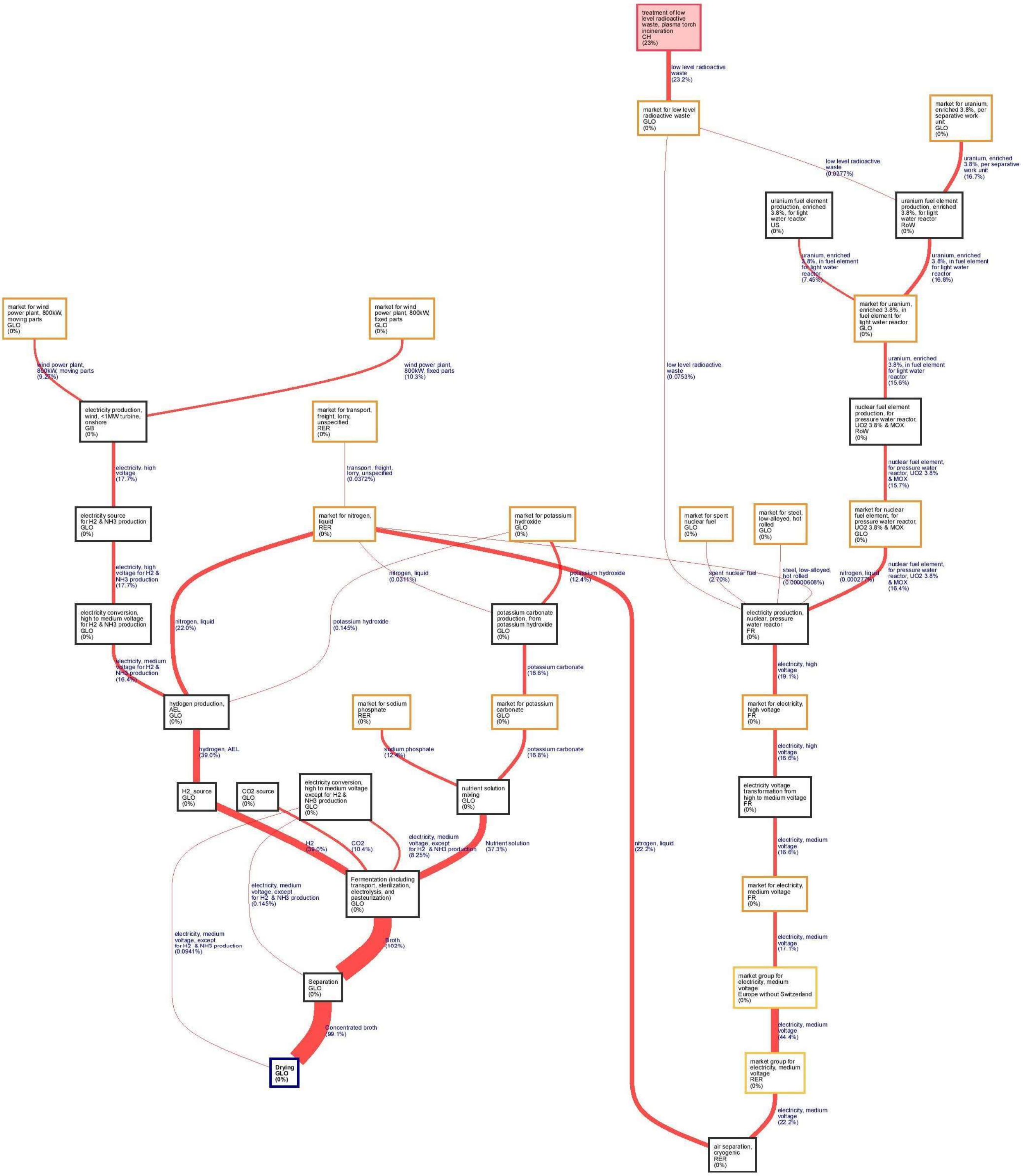


Figure 21A. CA Bio 2050, ionising radiation

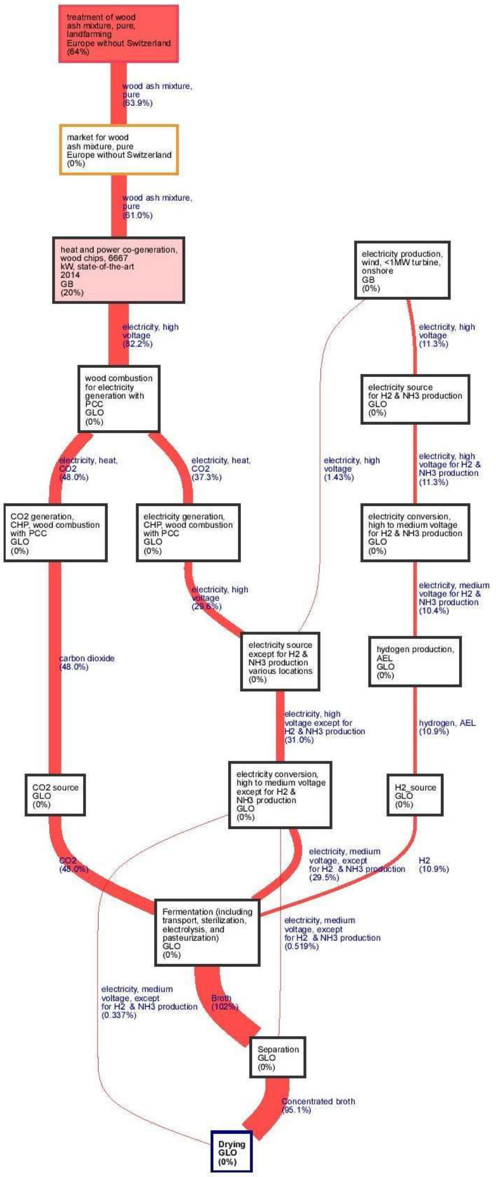


Figure 22A. CA Bio 2050, non-carcinogenic effects

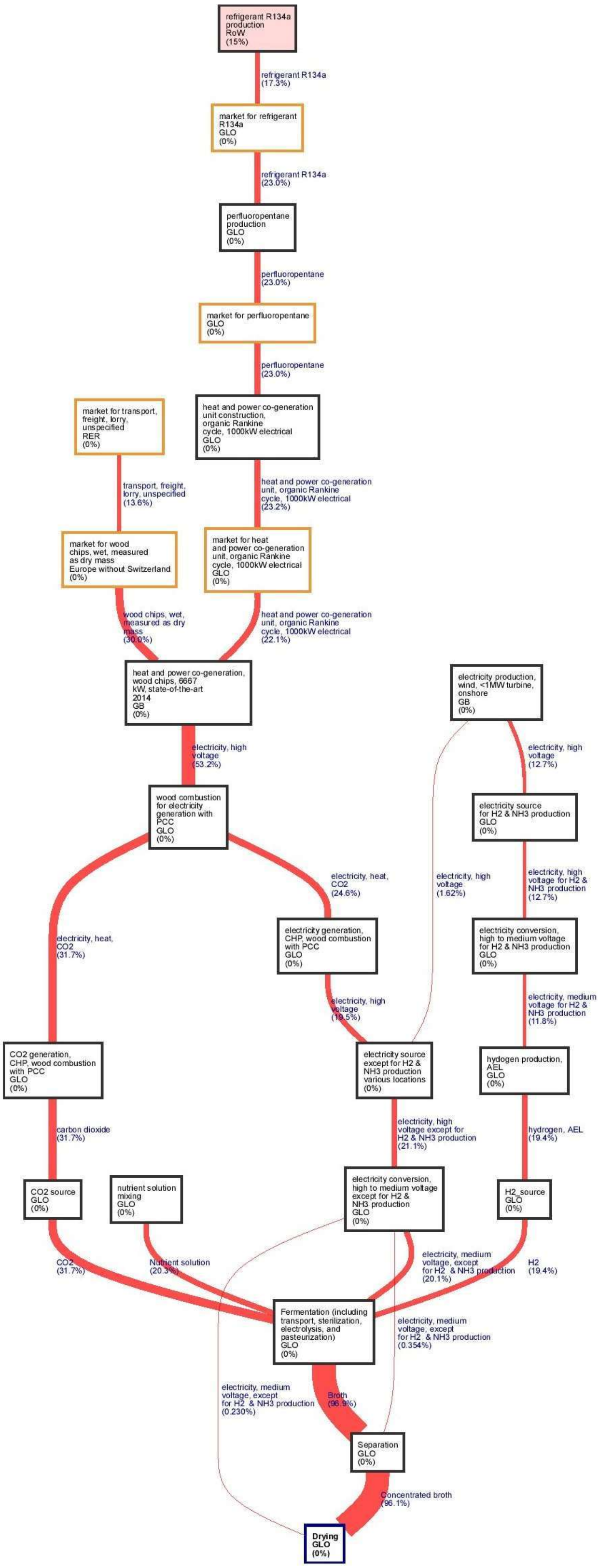


Figure 23A. CA Bio 2050, ozone layer depletion

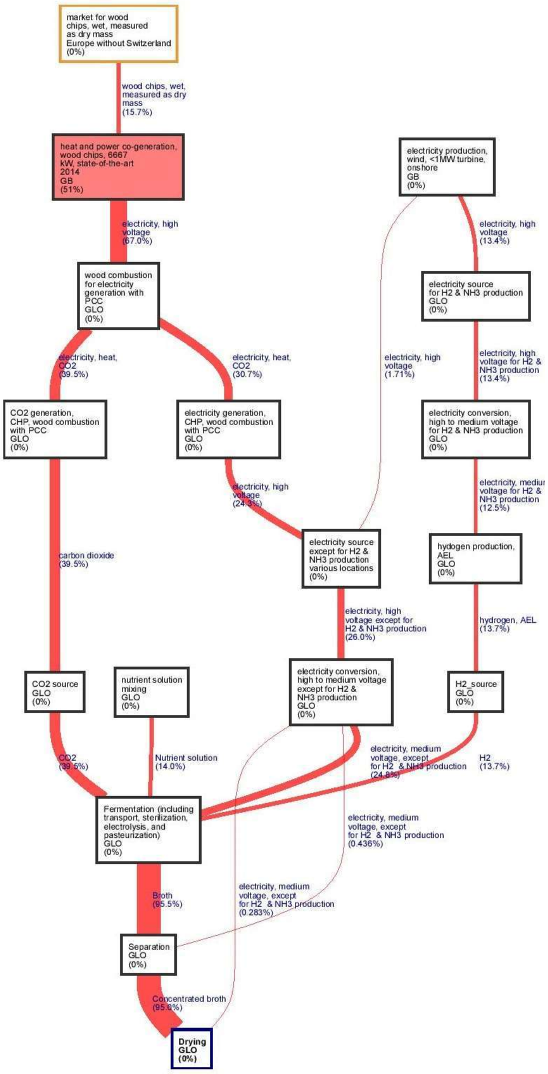


Figure 24A. CA Bio 2050, photochemical ozone creation

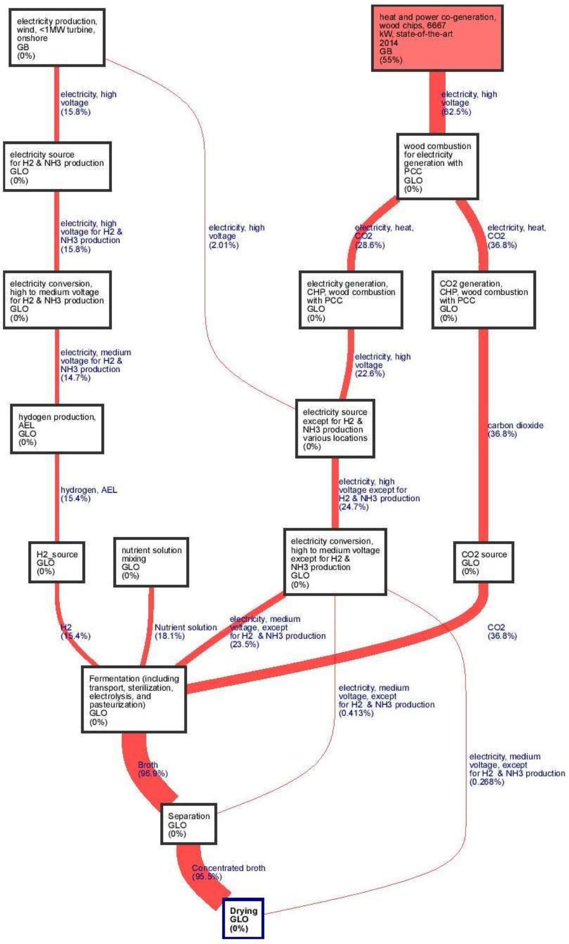


Figure 25A. CA Bio 2050, respiratory effects, inorganics

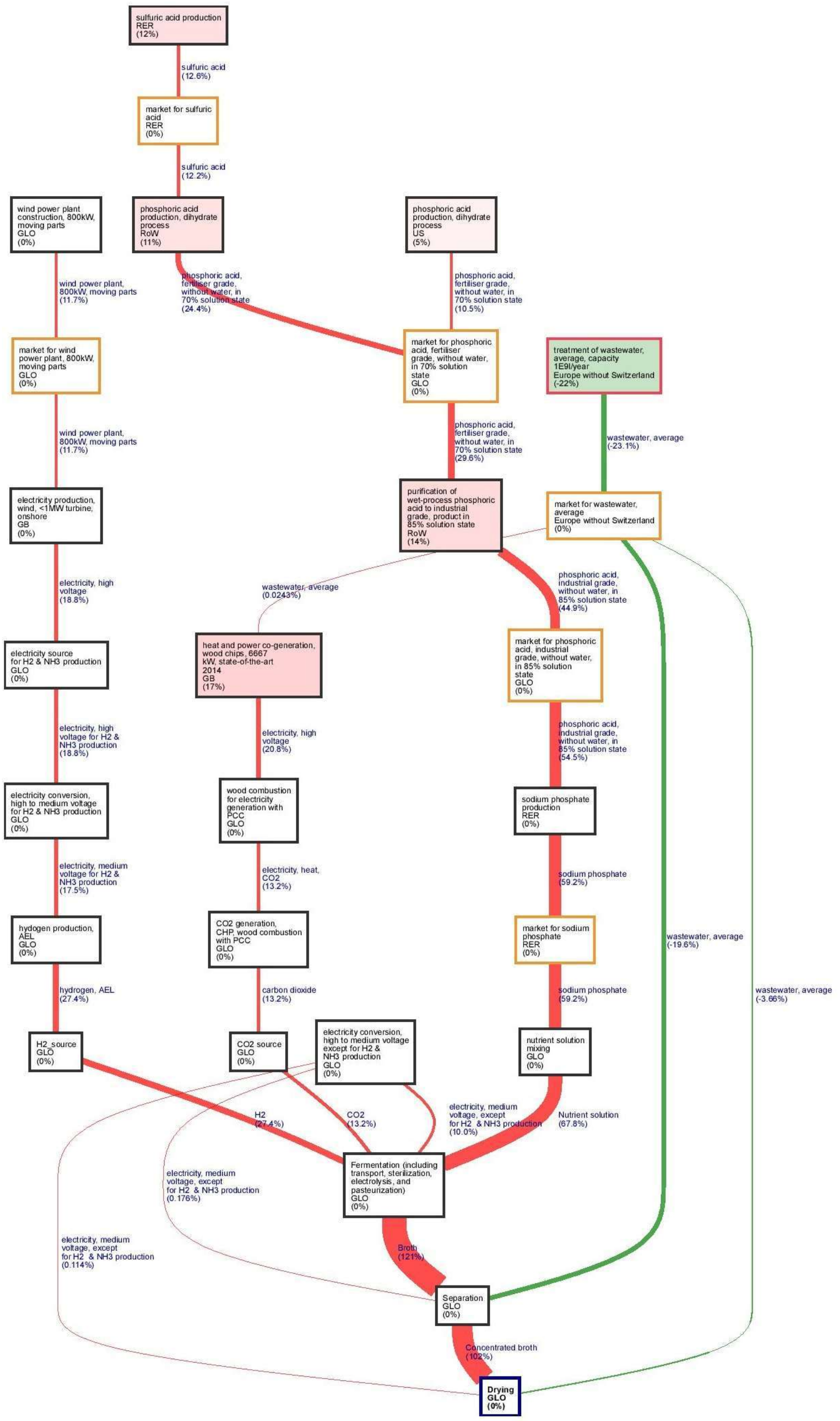


Figure 26A. CA Bio 2050, dissipated water

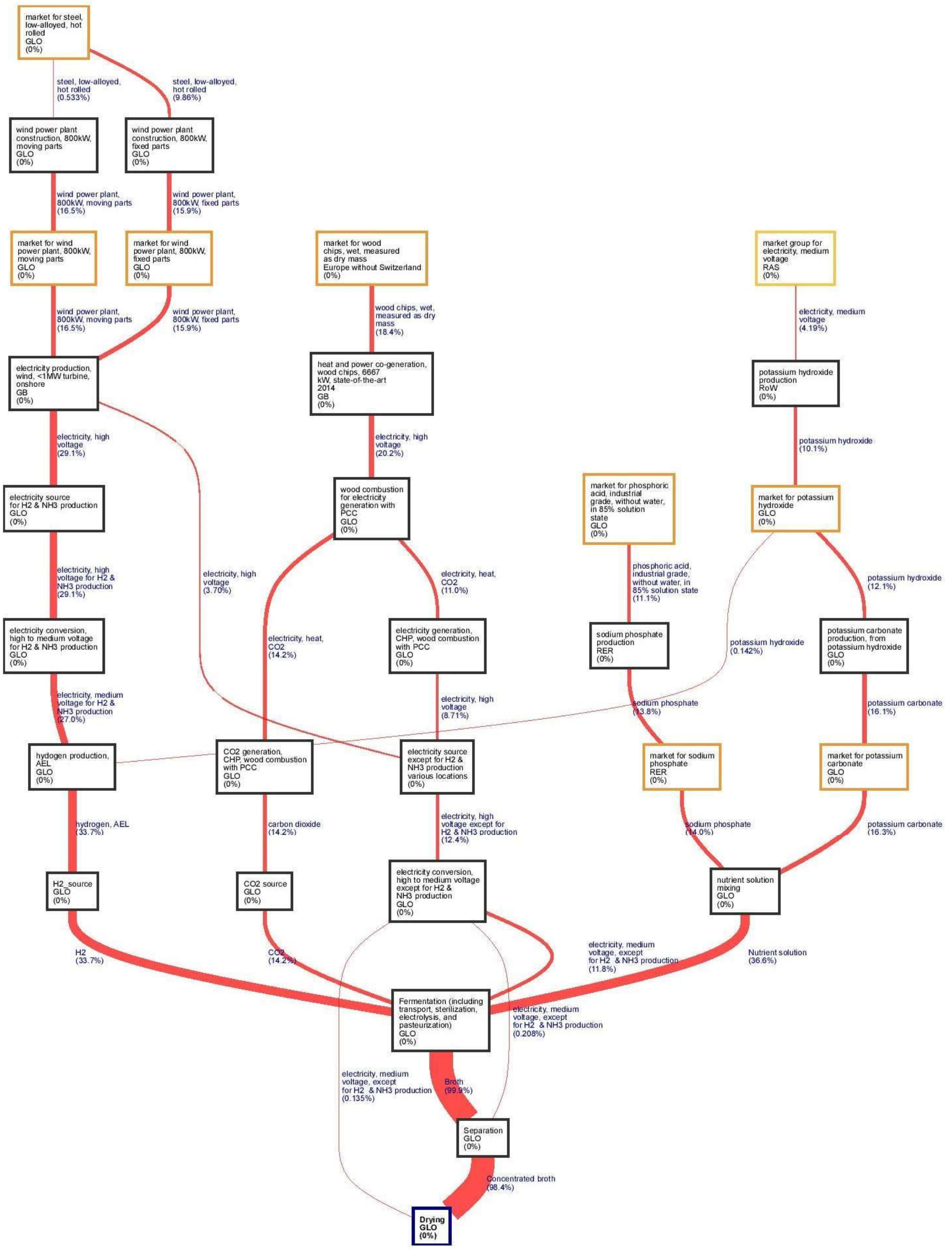


Figure 27A. CA Bio 2050, resources, fossils

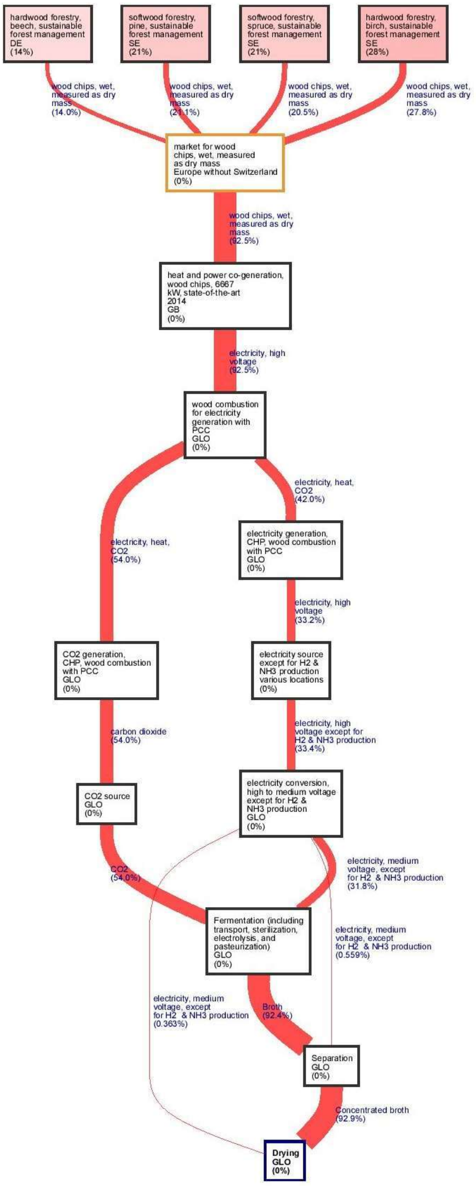


Figure 28A. CA Bio 2050, land use

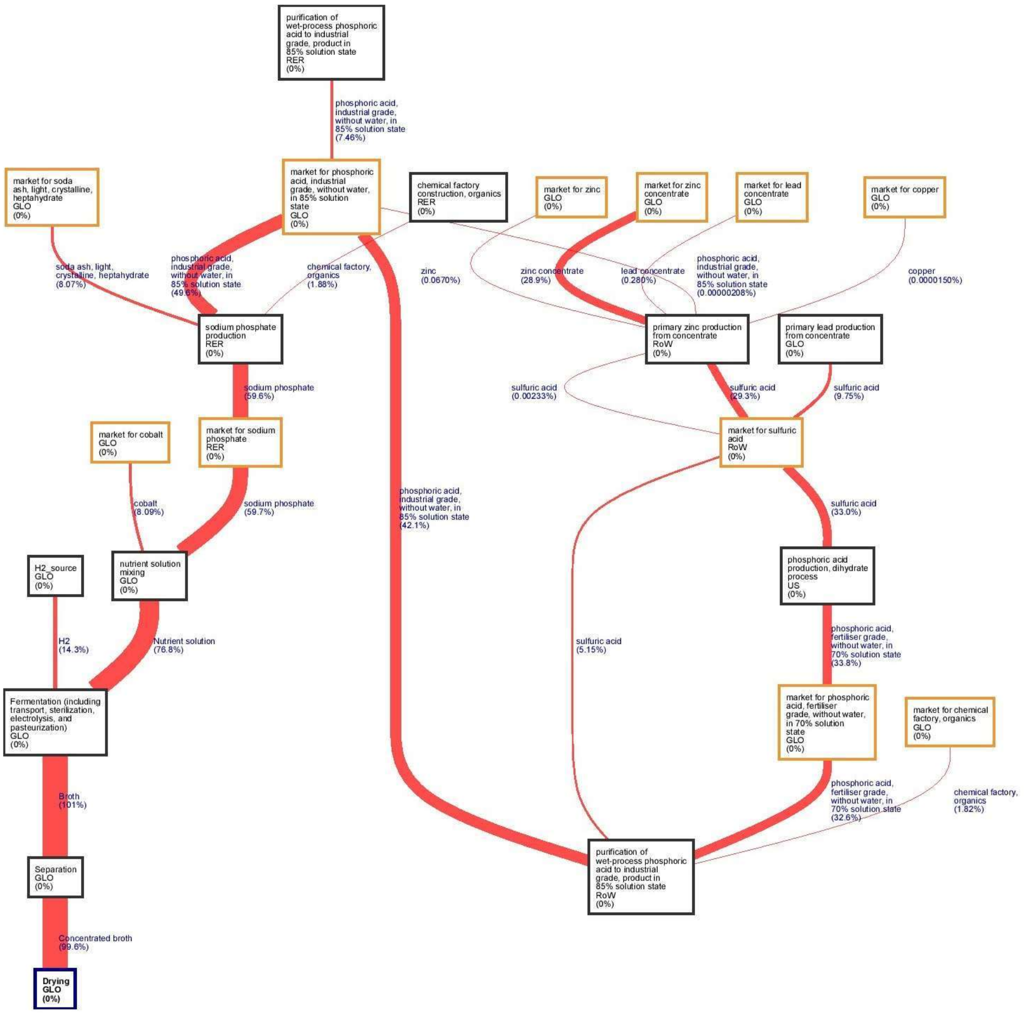


Figure 29A. CA Bio 2050, minerals and metals

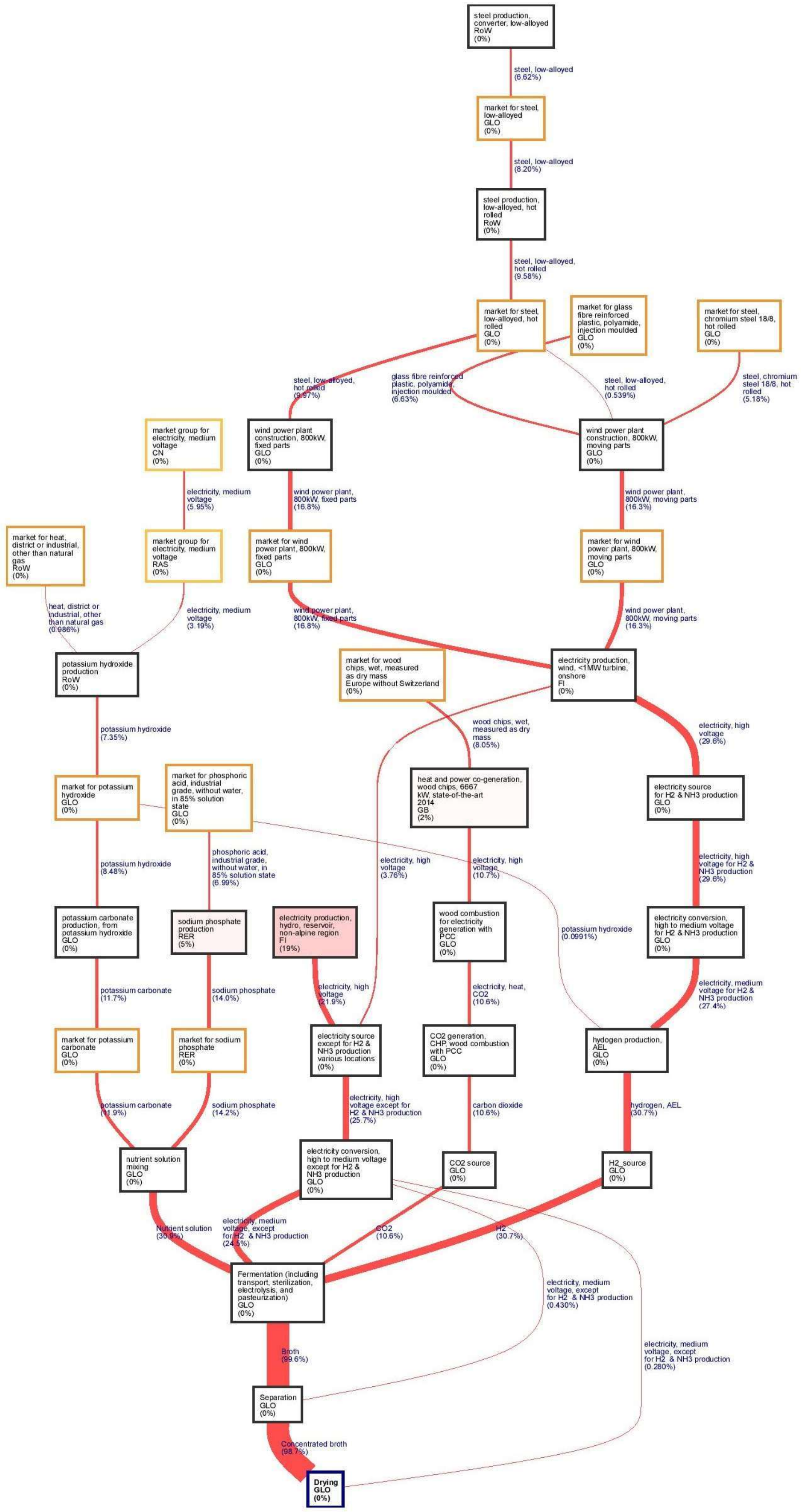


Figure 30A. CA Hydro 2050, climate change

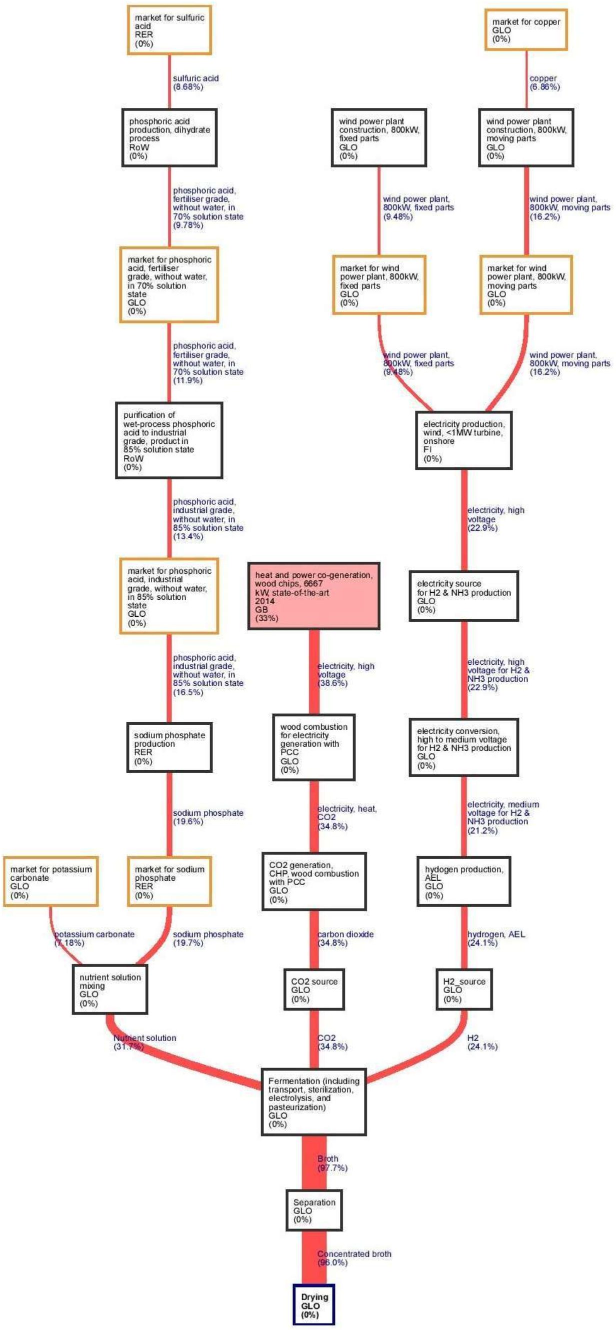


Figure 31A. CA Hydro 2050, freshwater & terrestrial acidification

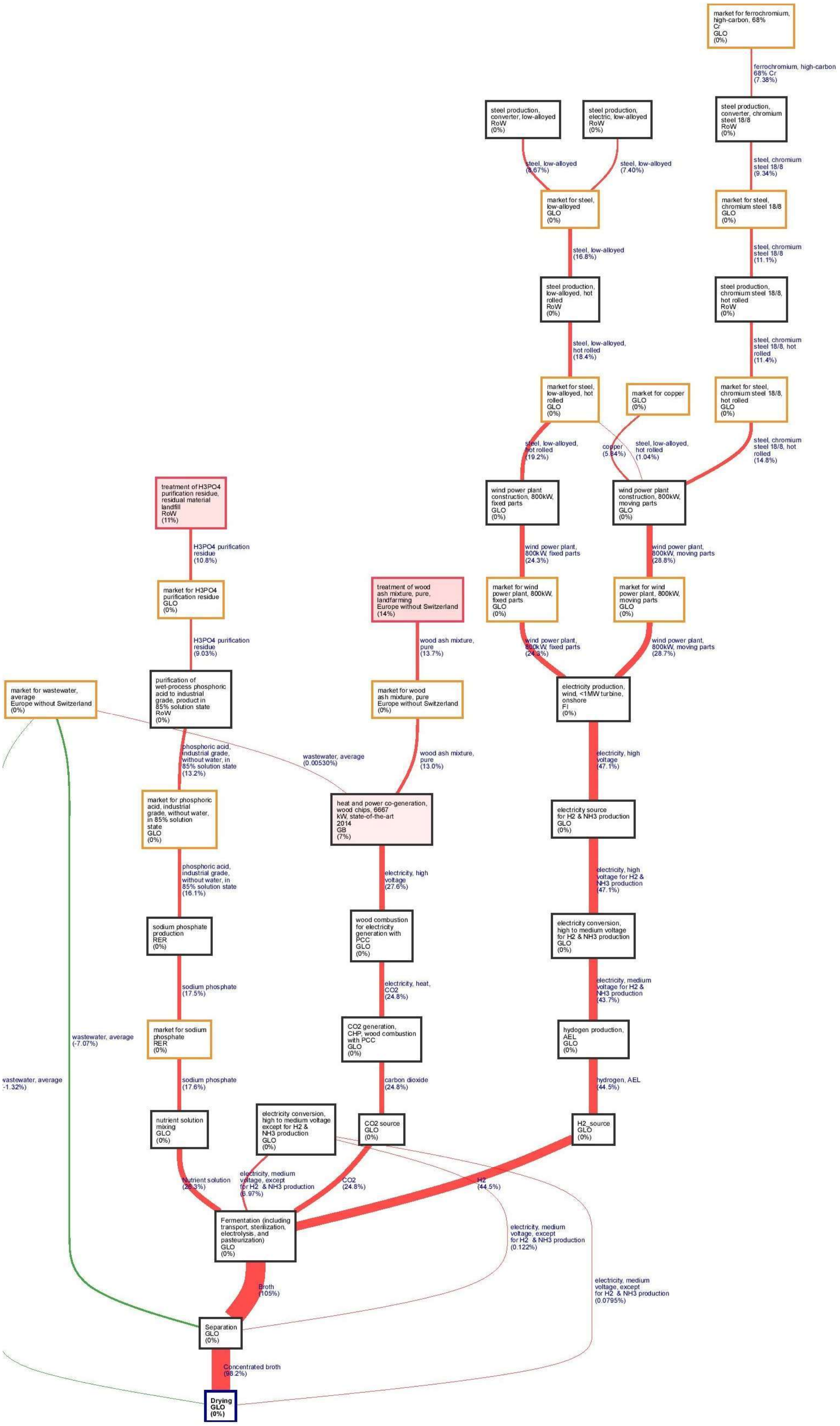


Figure 32A. CA Hydro 2050, freshwater ecotoxicity

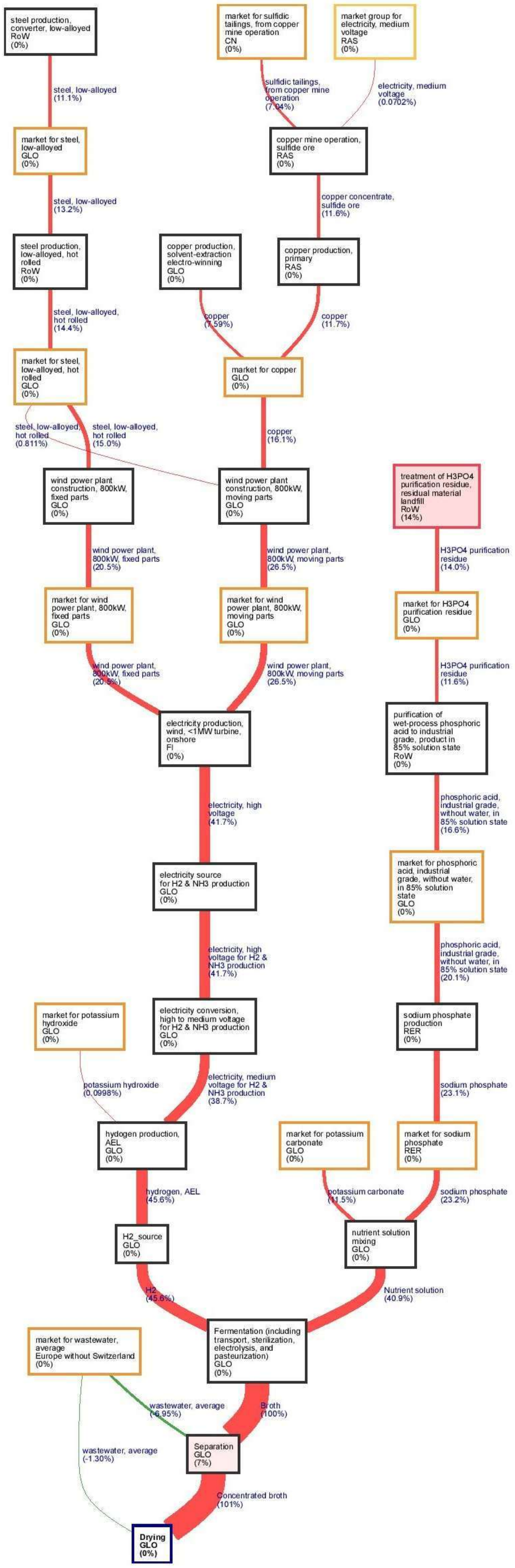


Figure 33A. CA Hydro 2050, freshwater eutrophication

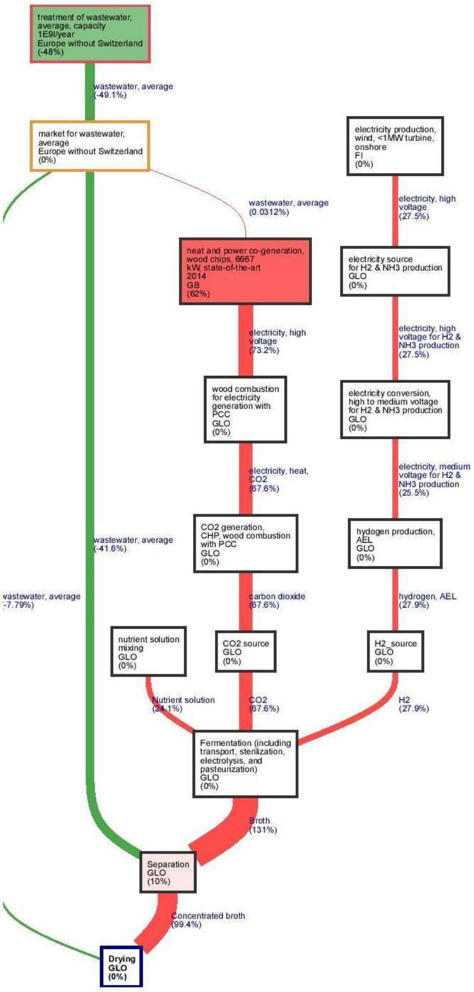


Figure 34A. CA Hydro 2050, marine eutrophication

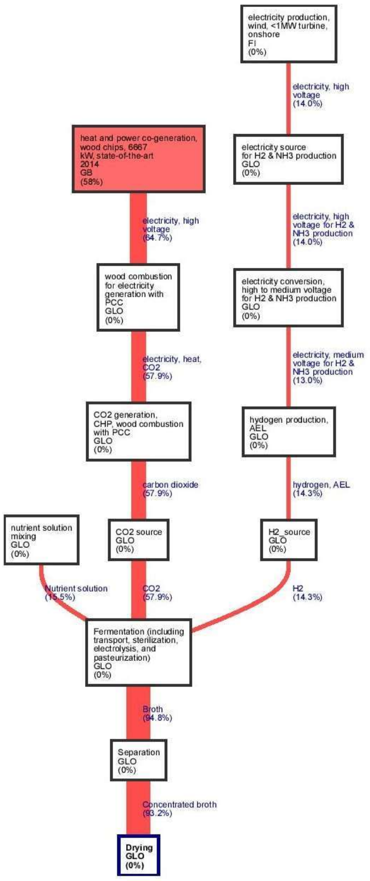


Figure 35A. CA Hydro 2050, terrestrial eutrophication

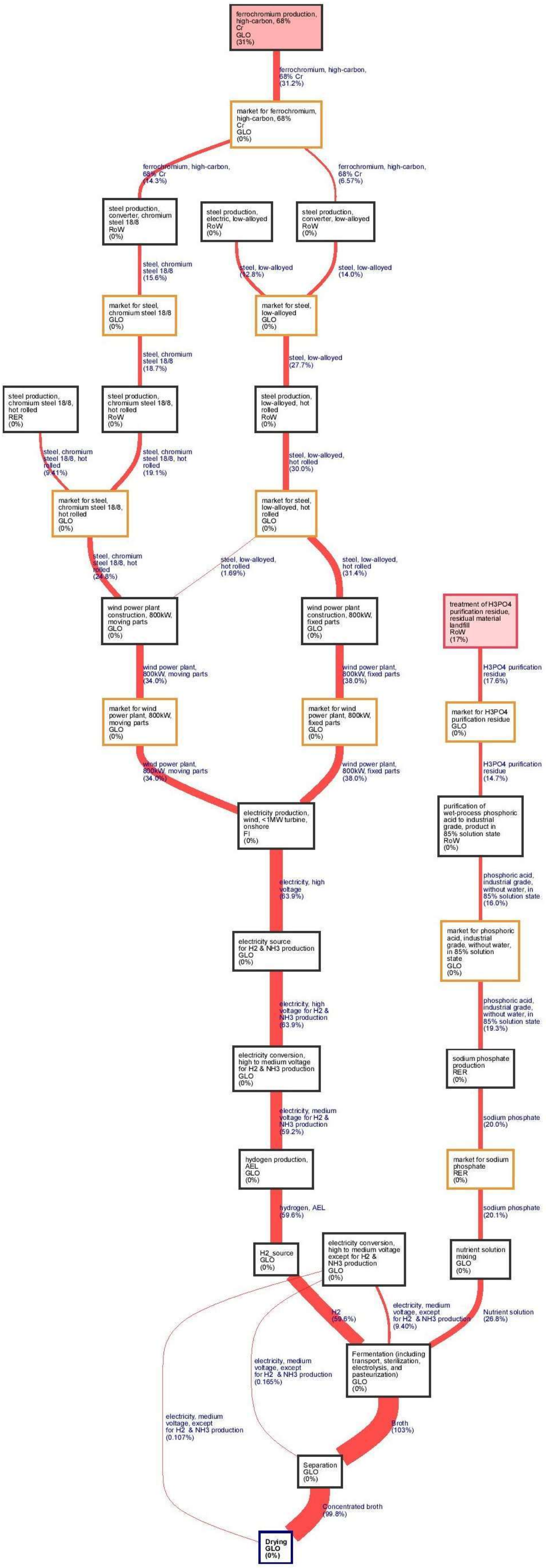


Figure 36A. CA Hydro 2050, carcinogenic effects

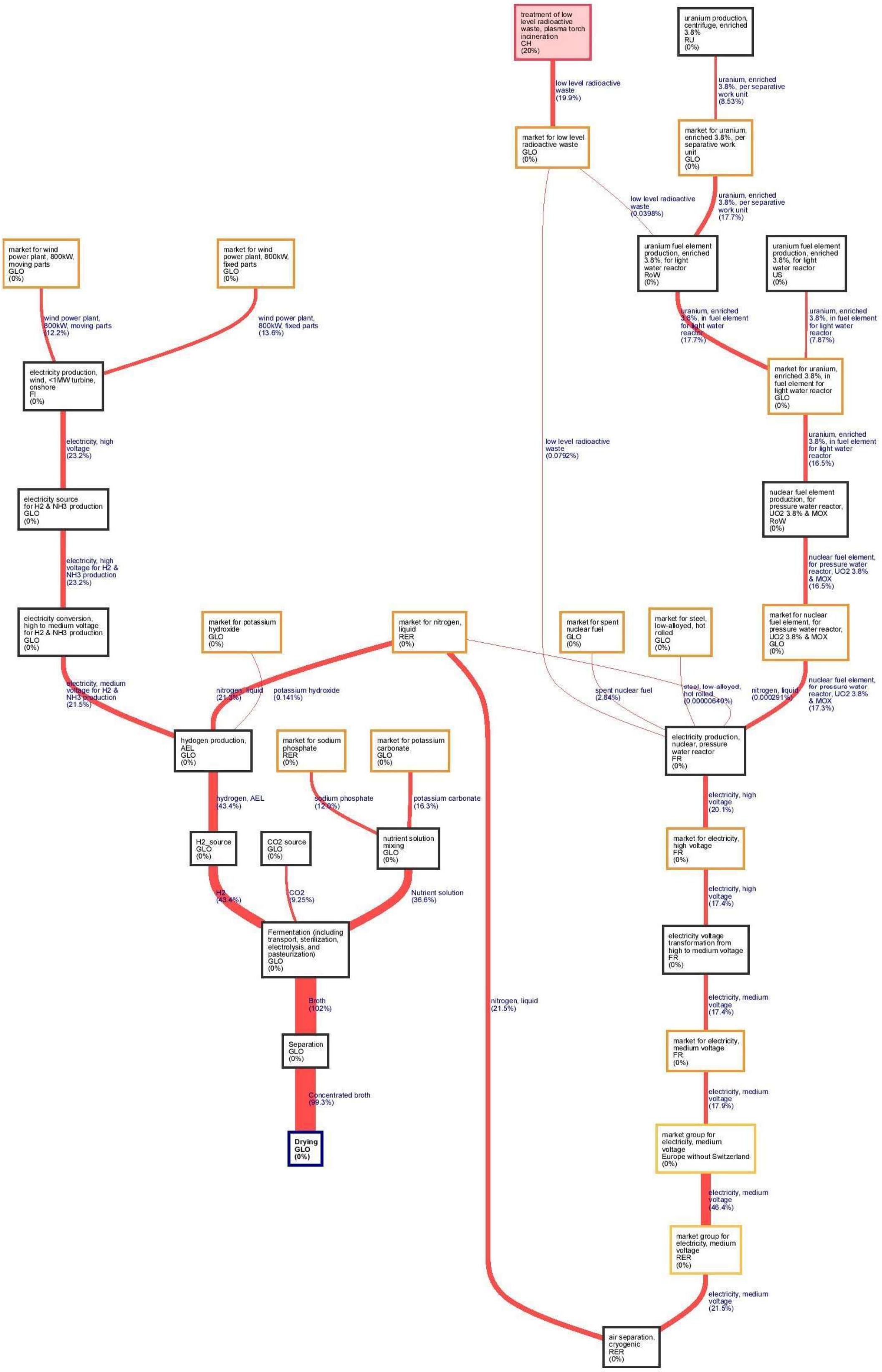


Figure 37A. CA Hydro 2050, ionising radiation

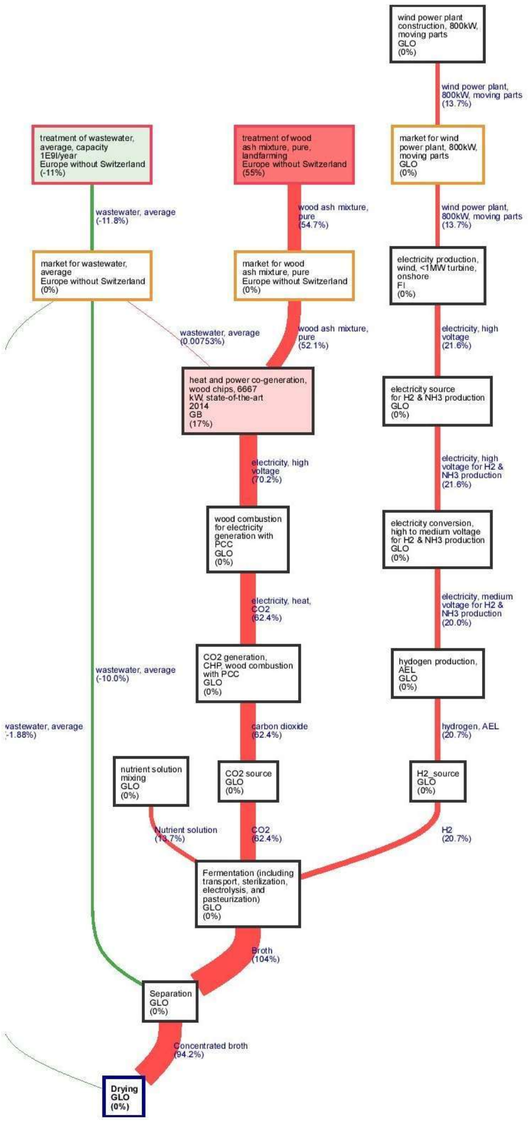


Figure 38A. CA Hydro 2050, non-carcinogenic effects

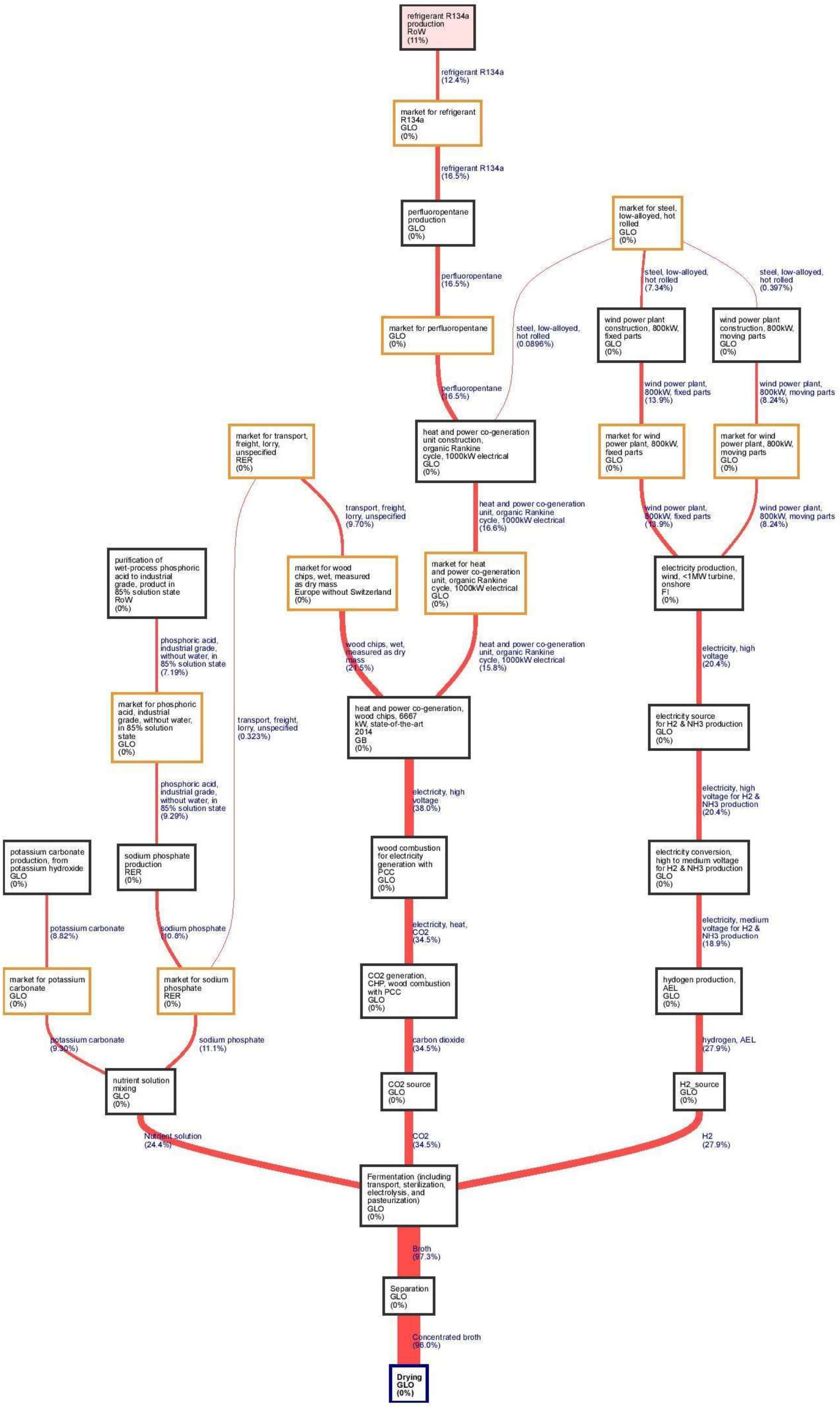


Figure 39A. CA Hydro 2050, ozone layer depletion

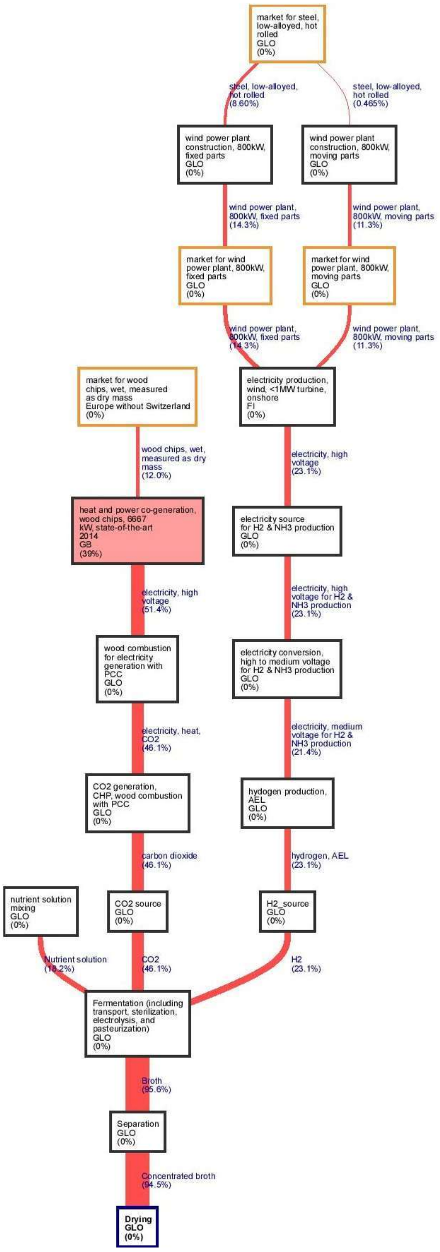


Figure 40A. CA Hydro 2050, photochemical ozone creation

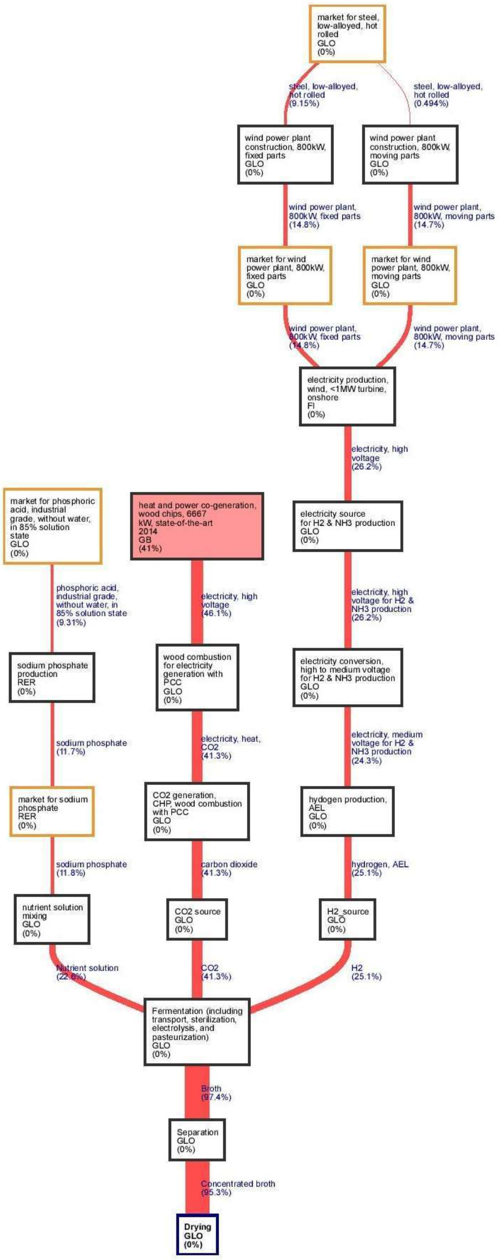


Figure 41A. CA Hydro 2050, respiratory effects, inorganics

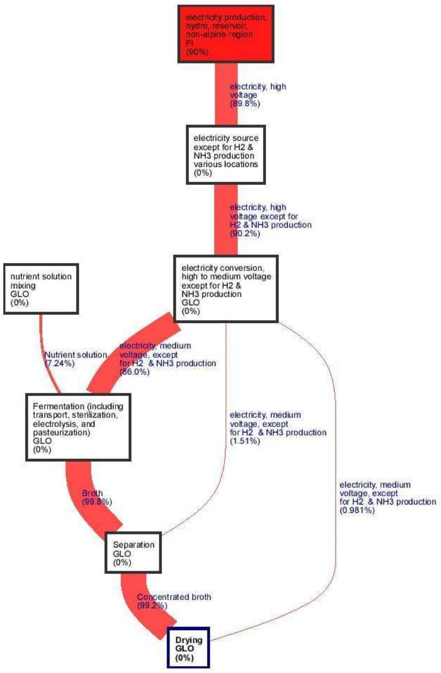


Figure 42A. CA Hydro 2050, dissipated water

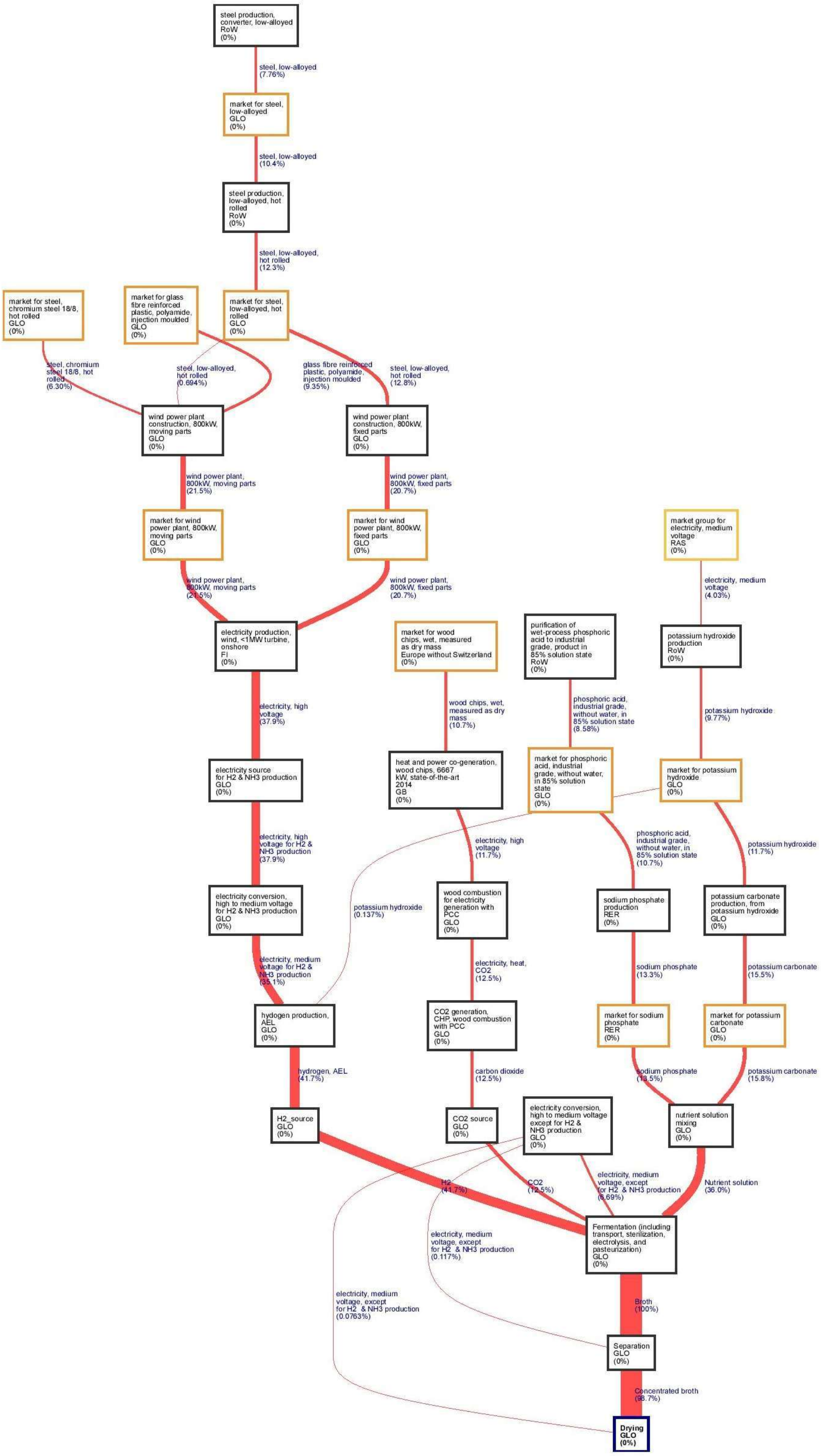


Figure 43A. CA Hydro 2050, resources, fossils

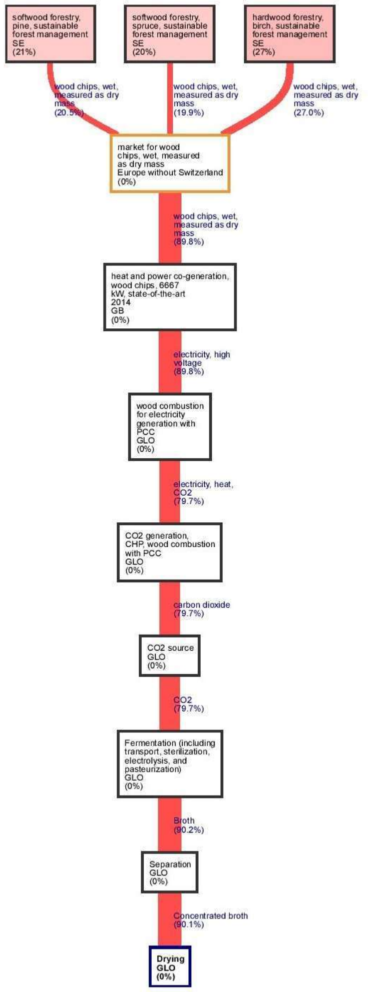


Figure 44A. CA Hydro 2050, land use

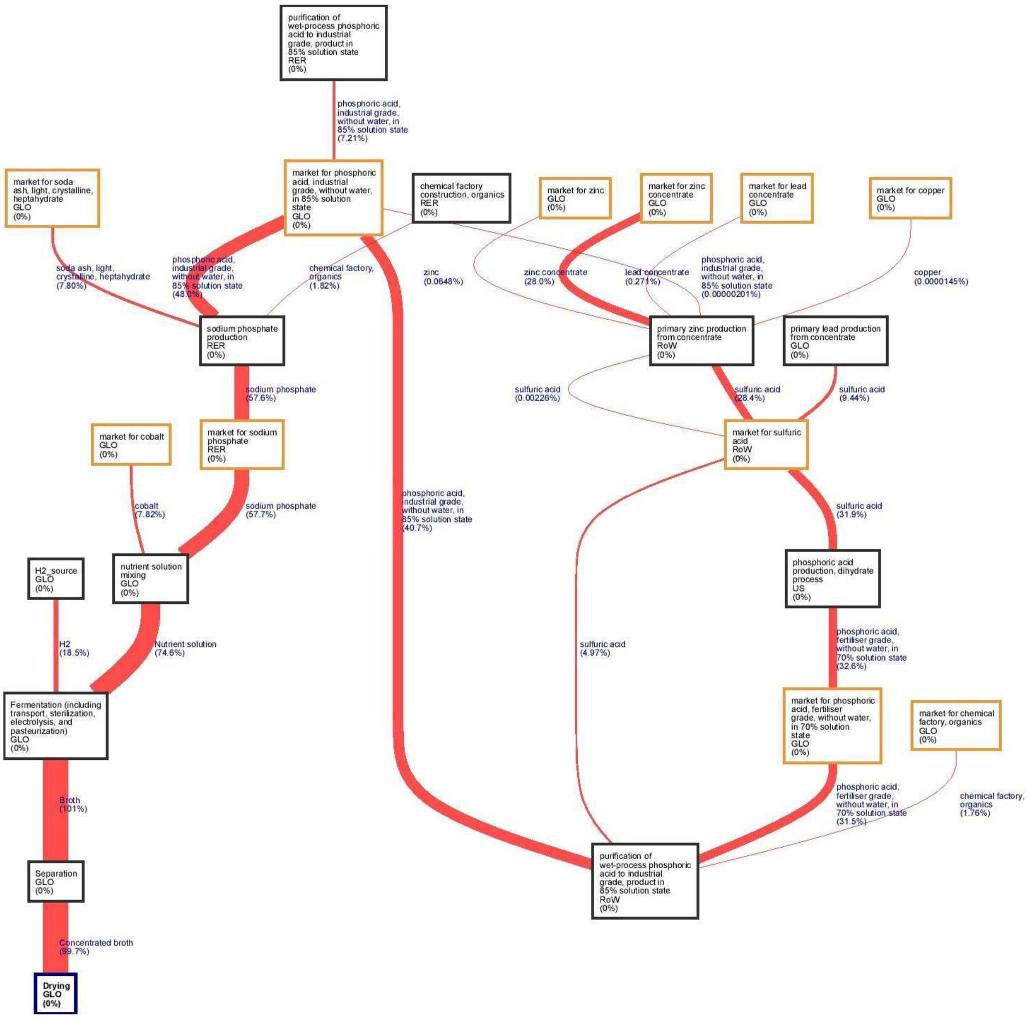


Figure 45A. CA Hydro 2050, minerals and metals

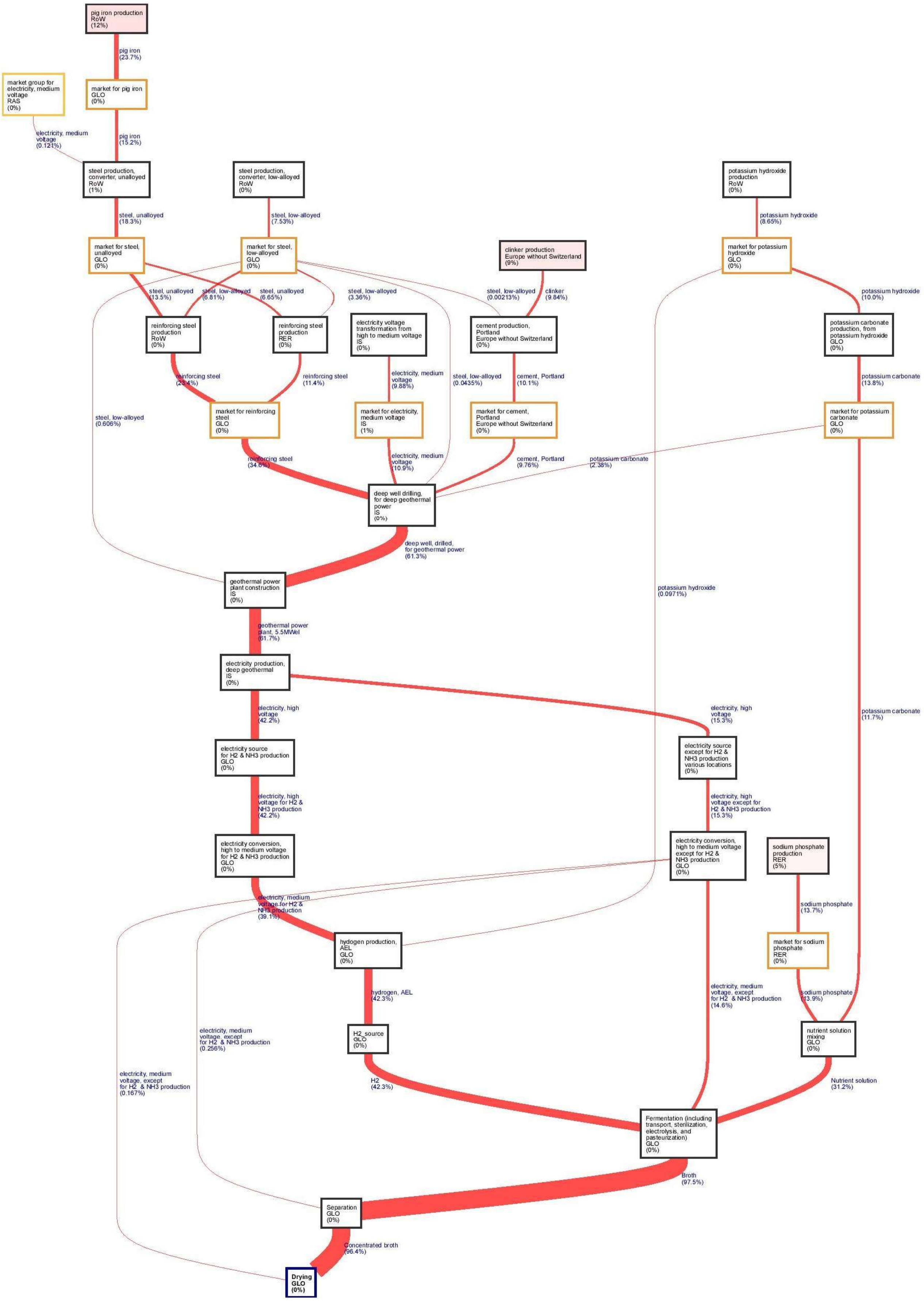


Figure 46A. CA Geo 2050, climate change

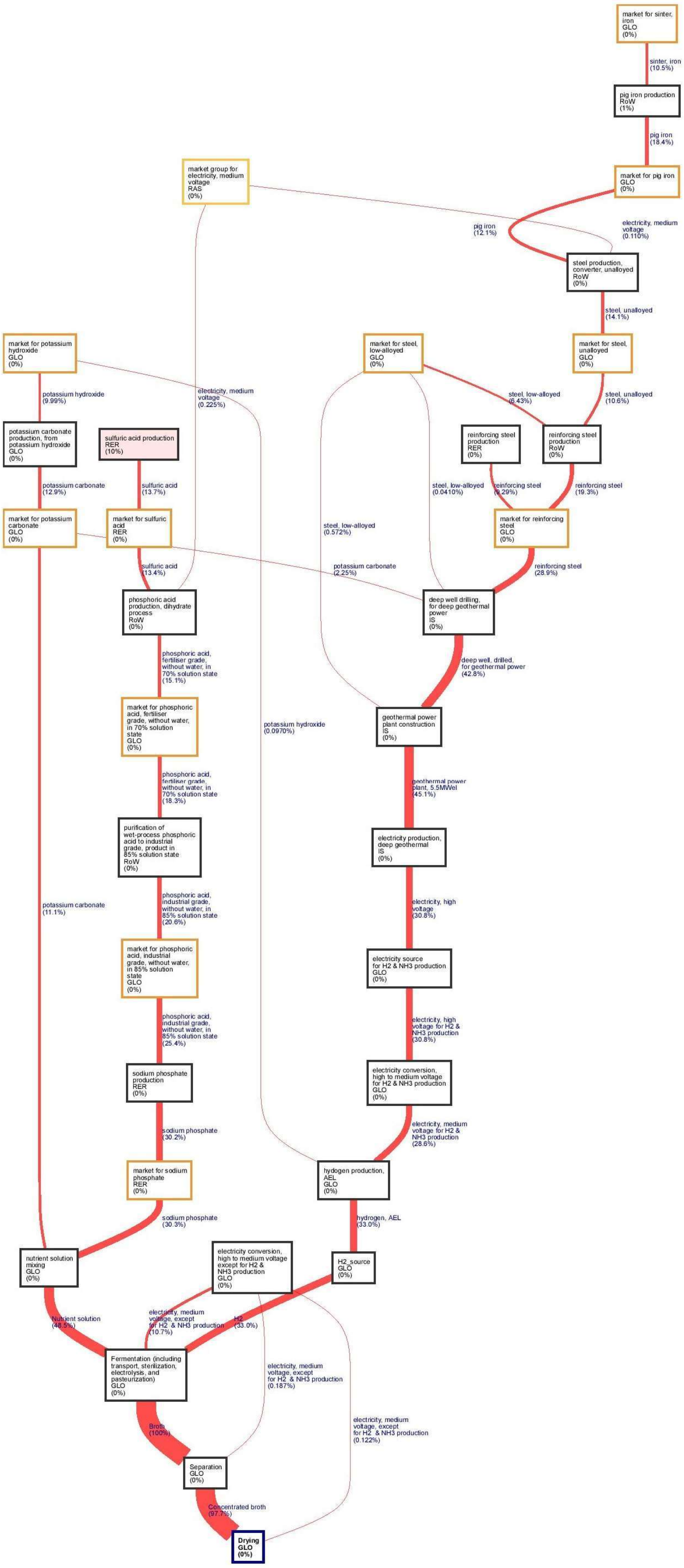


Figure 47A. CA Geo 2050, freshwater & terrestrial acidification

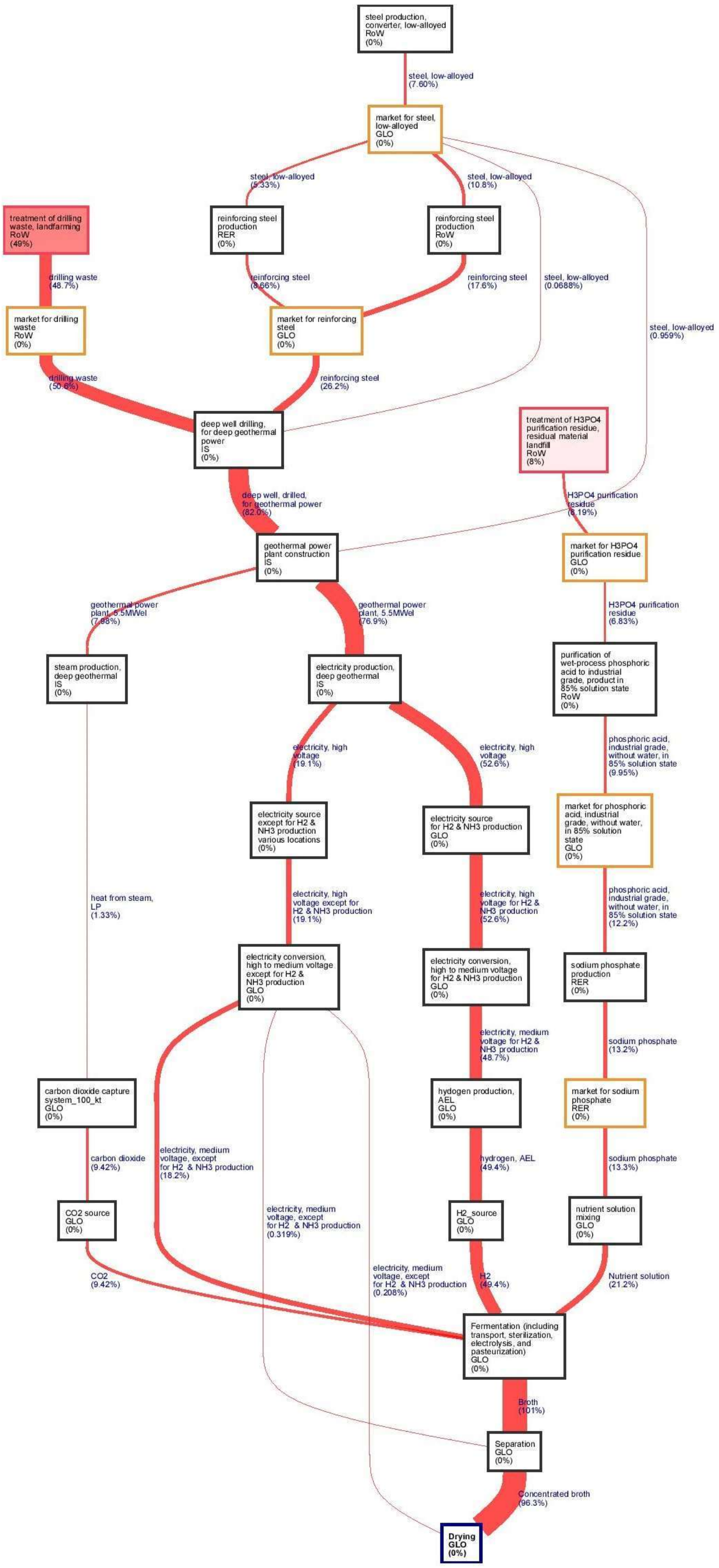


Figure 48A. CA Geo 2050, freshwater ecotoxicity

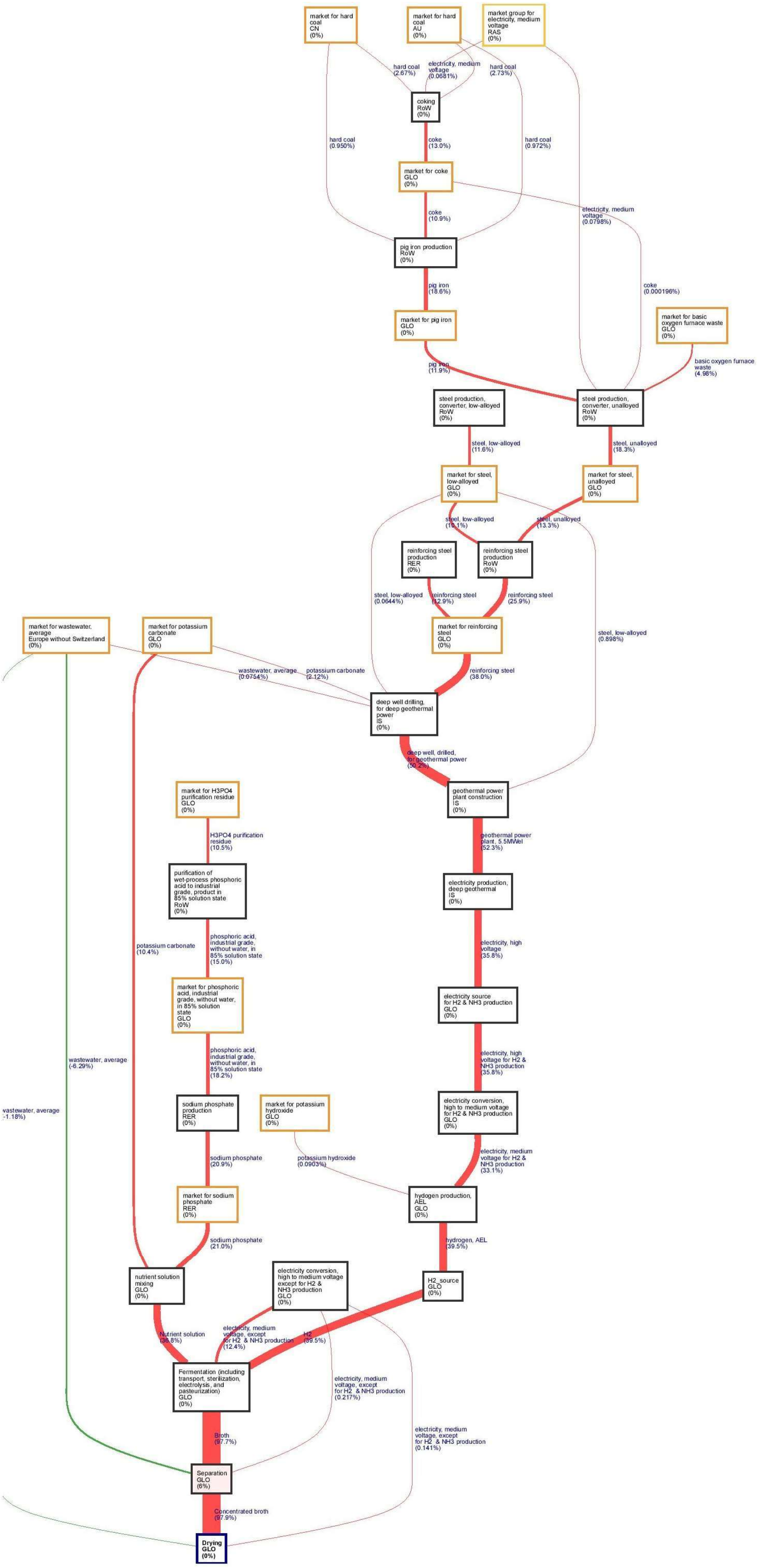


Figure 49A. CA Geo 2050, freshwater eutrophication

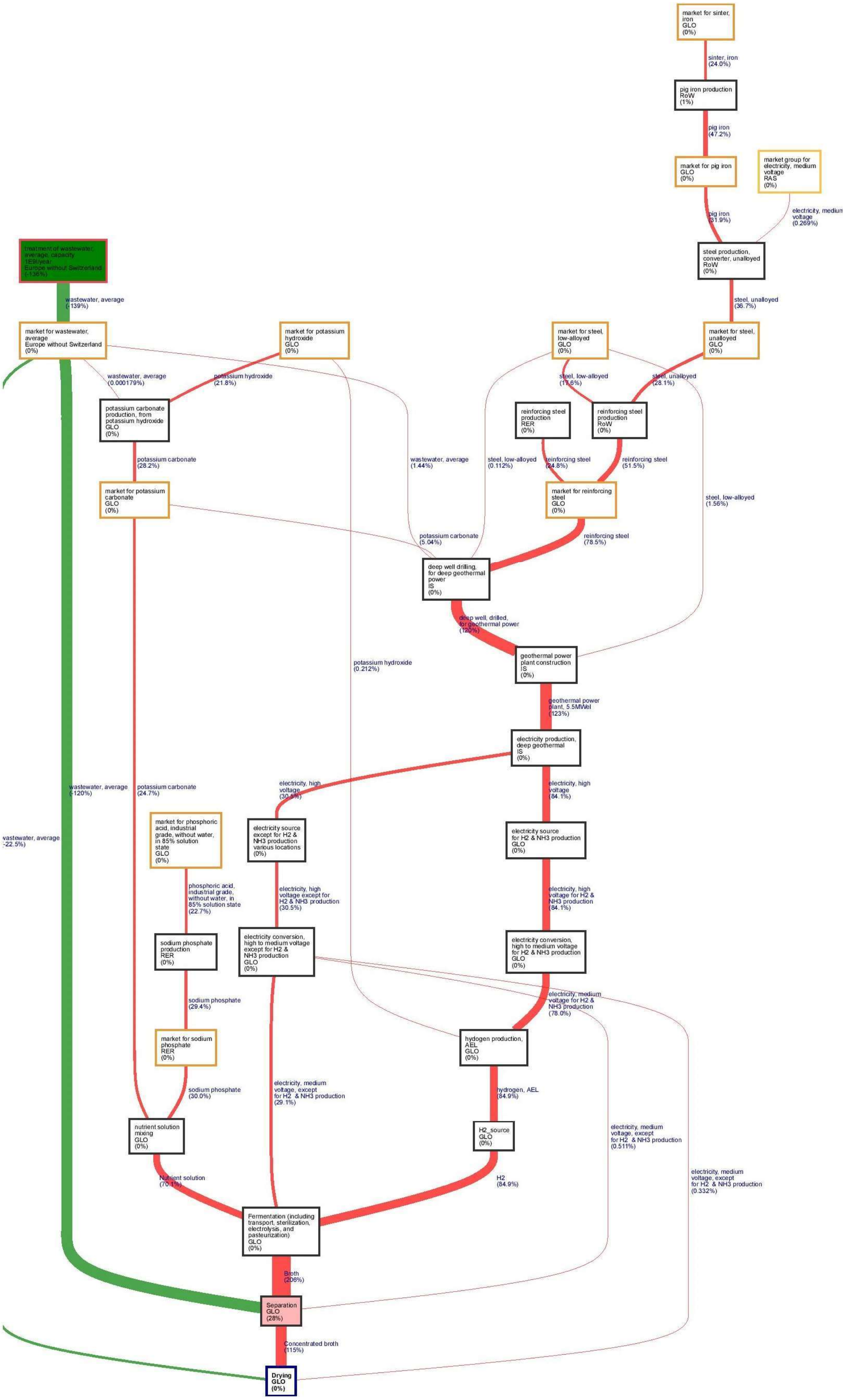


Figure 50A. CA Geo 2050, marine eutrophication

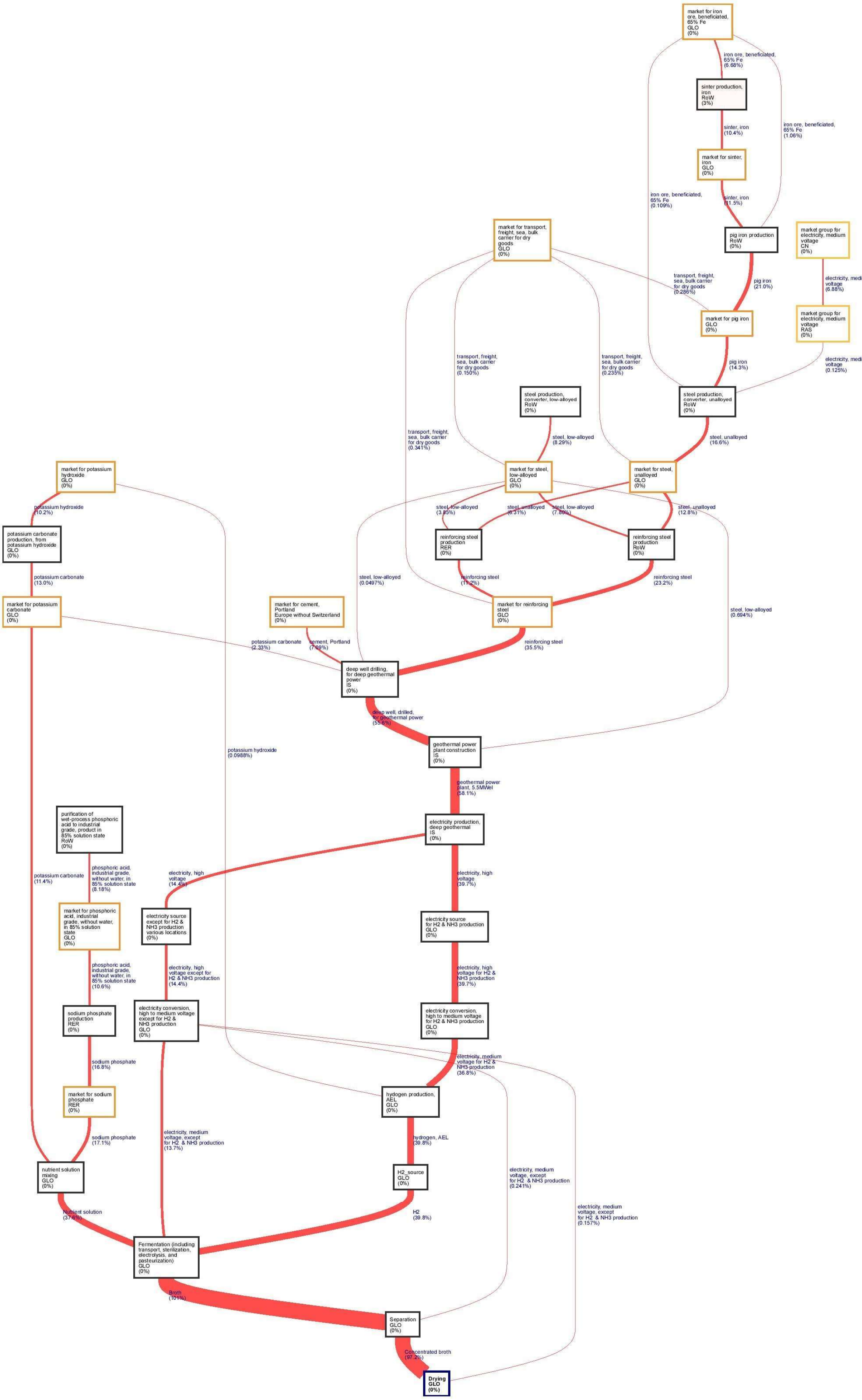


Figure 51A. CA Geo 2050, terrestrial eutrophication

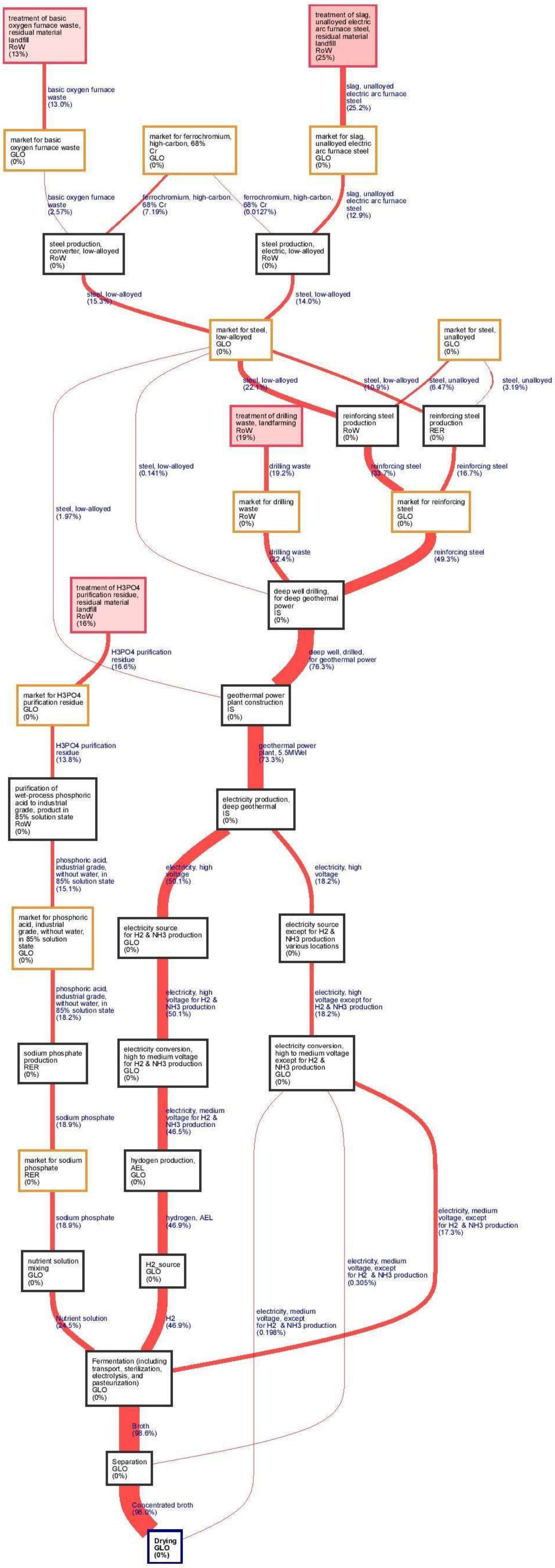


Figure 52A. CA Geo 2050, carcinogenic effects

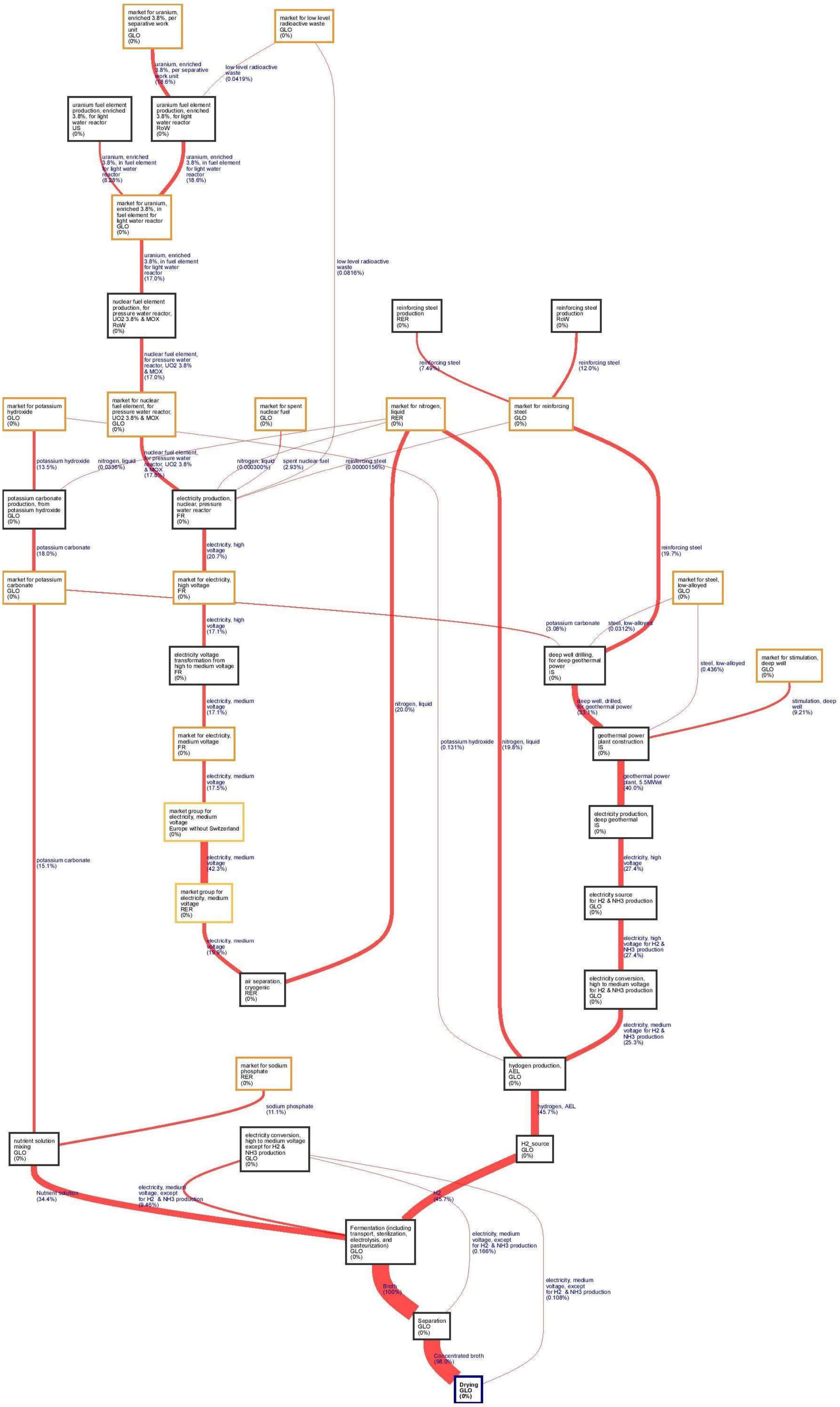


Figure 53A. CA Geo 2050, ionising radiation

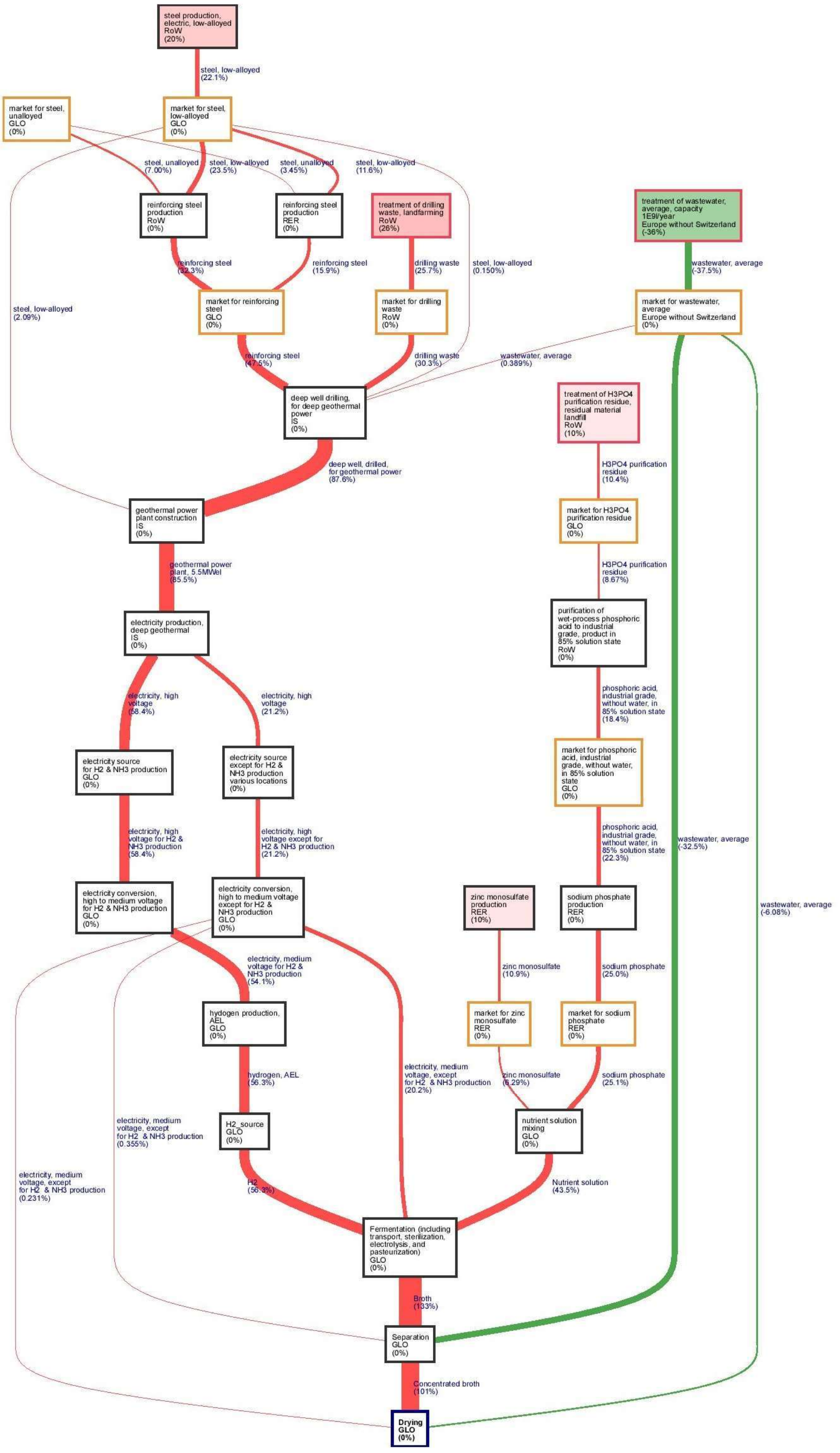


Figure 54A. CA Geo 2050, non-carcinogenic effects

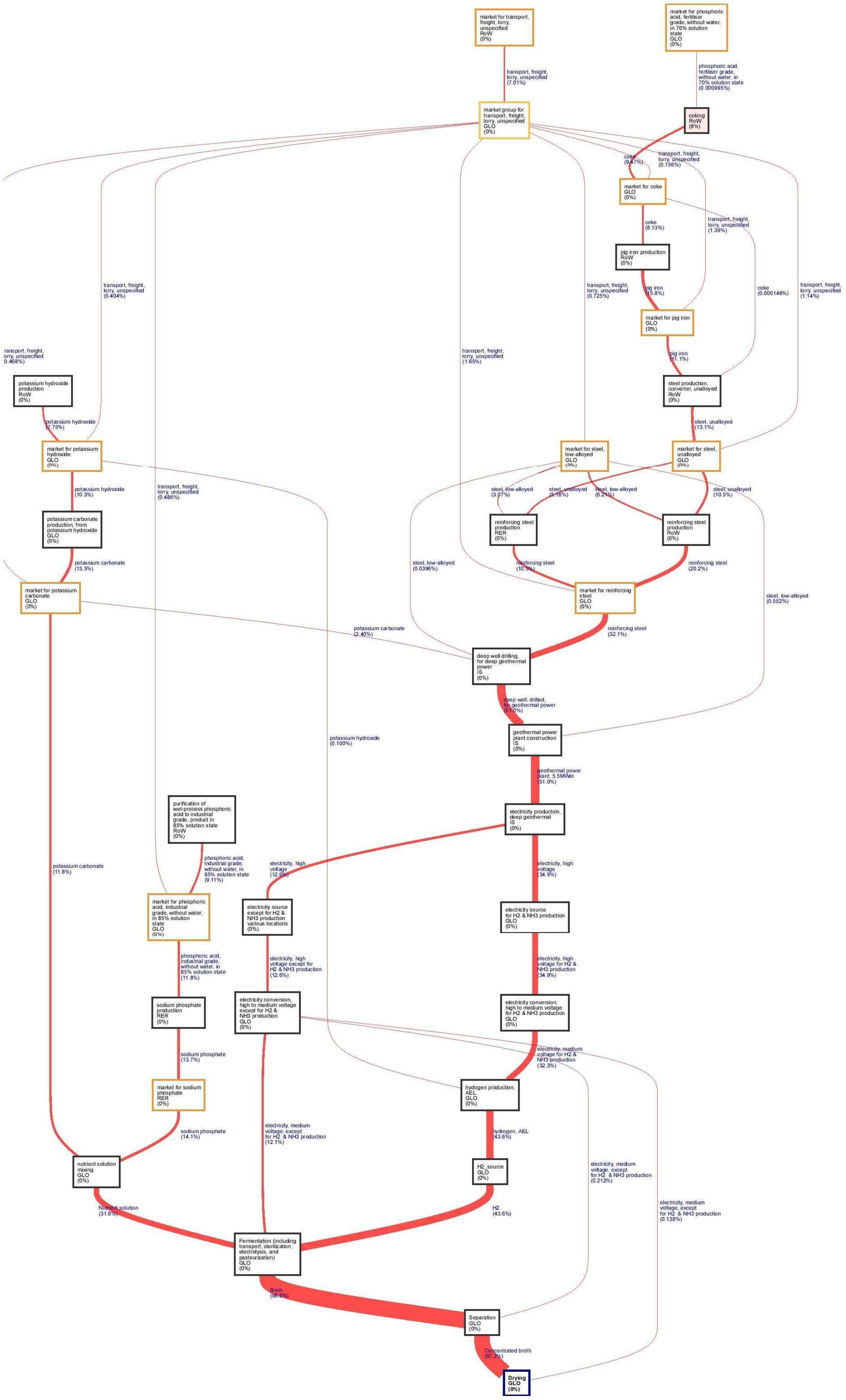


Figure 55A. CA Geo 2050, ozone layer depletion

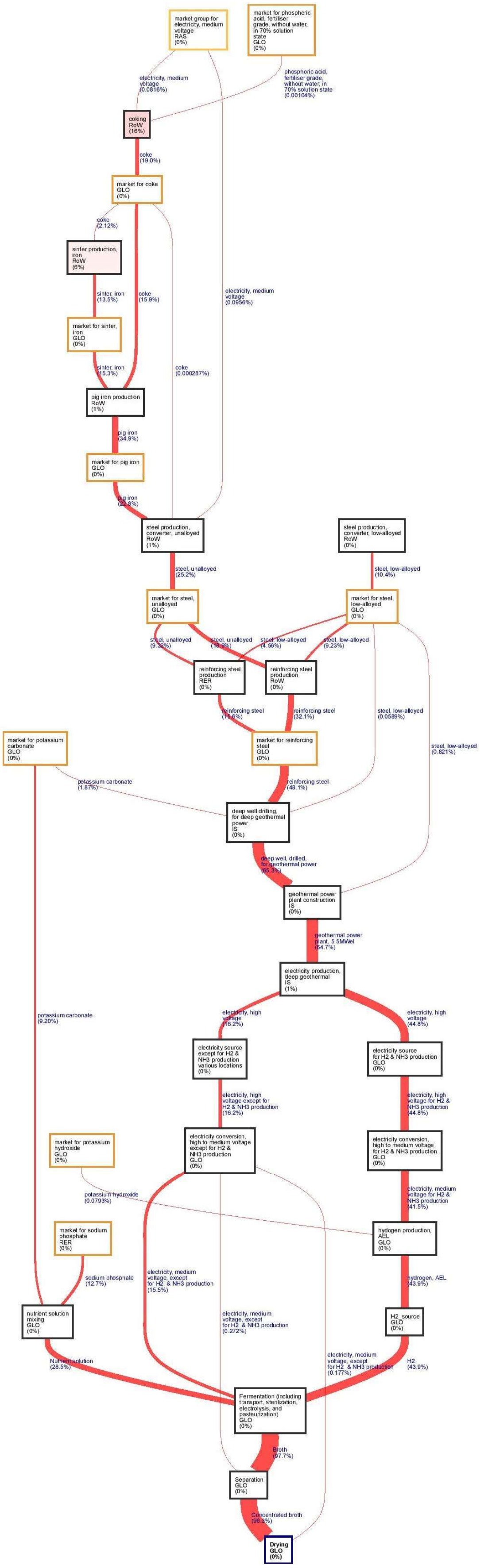


Figure 56A. CA Geo 2050, photochemical ozone creation

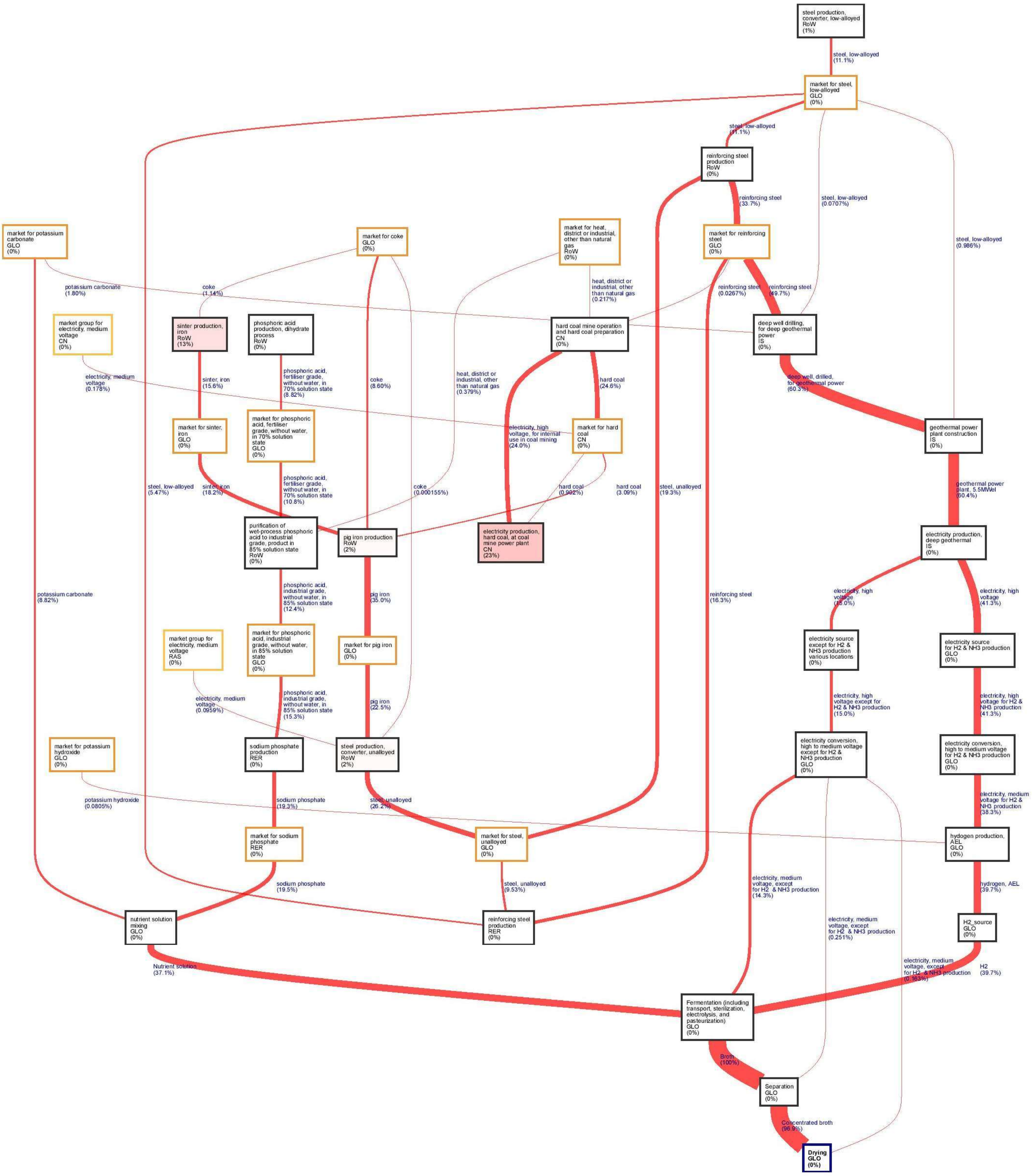


Figure 57A. CA Geo 2050, respiratory effects, inorganics

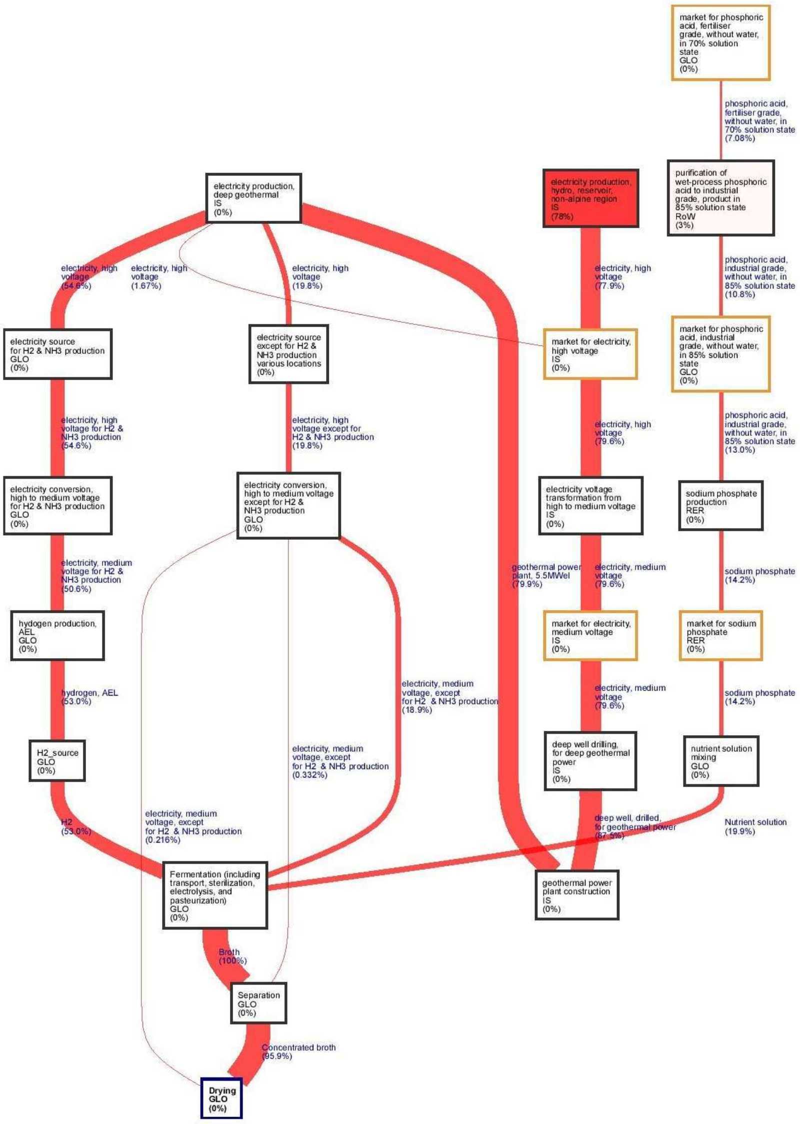


Figure 58A. CA Geo 2050, dissipated water

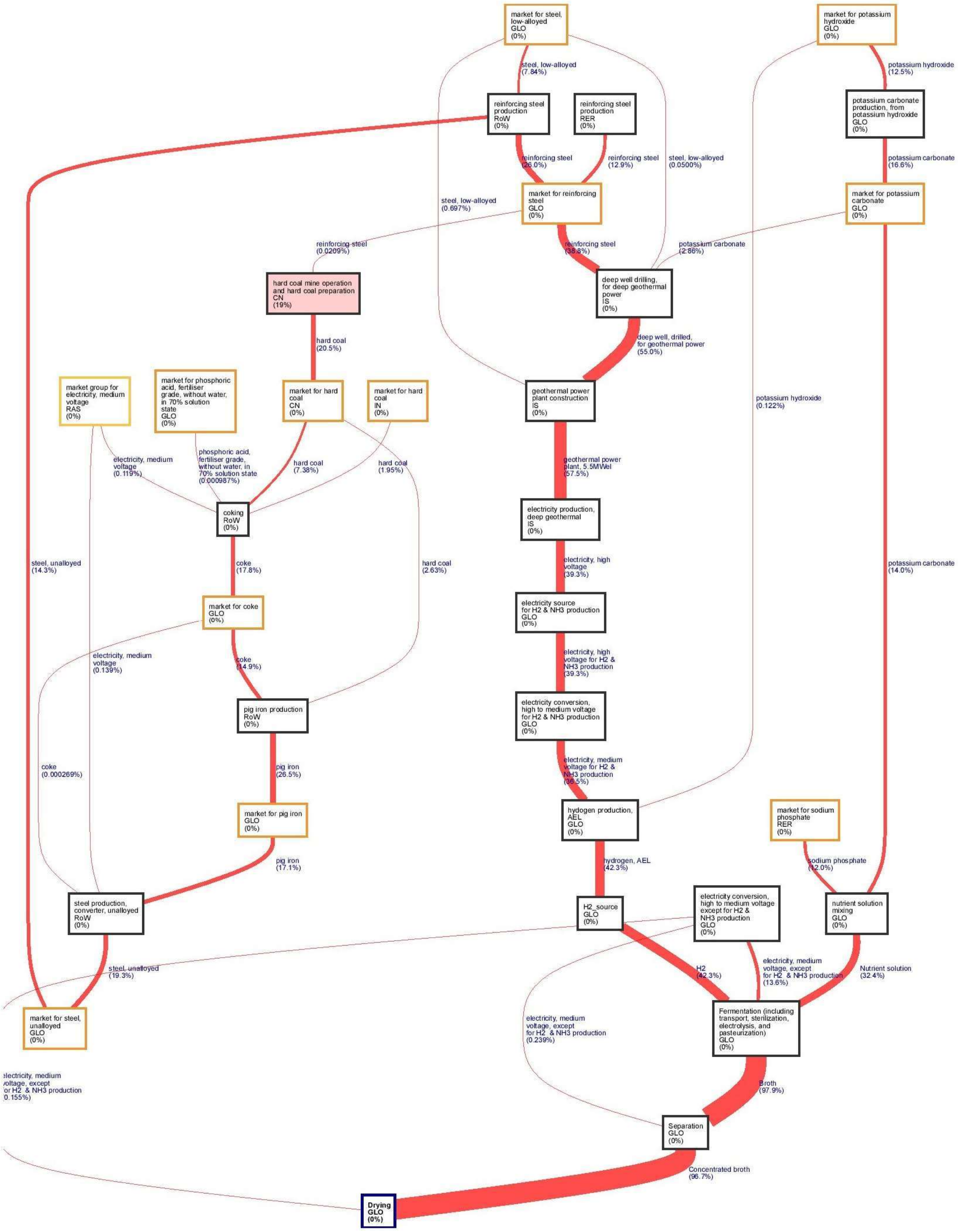


Figure 59A. CA Geo 2050, resources, fossils

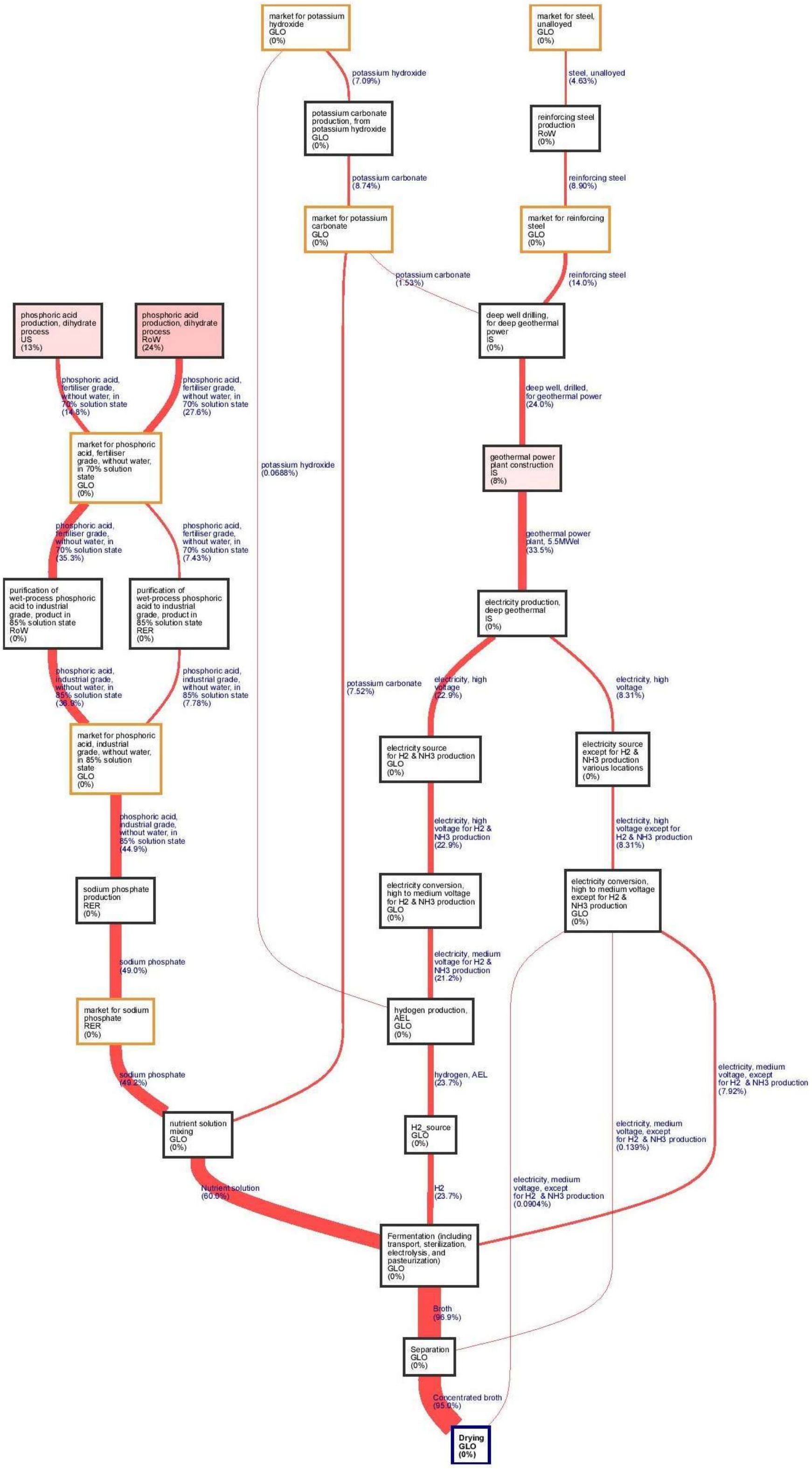


Figure 60A. CA Geo 2050, land use

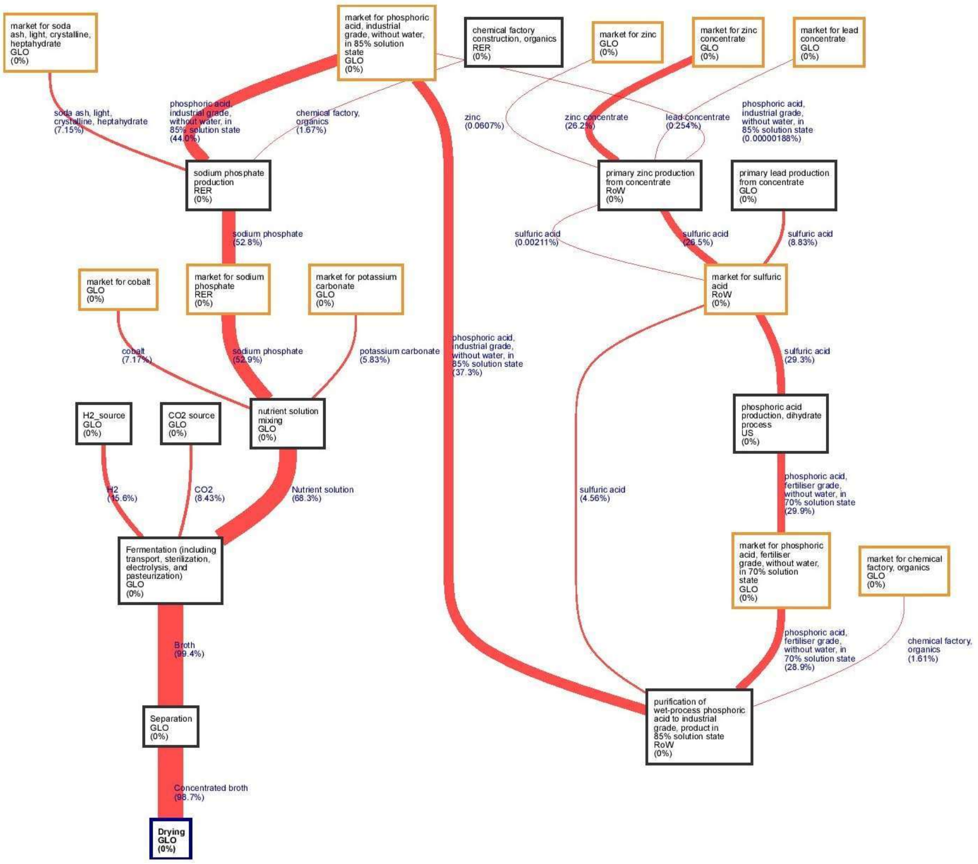


Figure 61A. CA Geo 2050, minerals and metals