



Sustainable electricity transition? The direct and indirect environmental impacts along the global supply chain of renewable and non-renewable sources of electricity

A greenhouse gas and material footprint analysis of the European Union's electricity consumption

by

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Abstract: This thesis studies the direct and indirect environmental impacts of the European Union's final consumption of electricity between 1995 and 2019. Using *Exiobase 3*, an environmentally extended multiregional input-output database, the greenhouse gas and material footprints for an aggregated 23 environmental stressors have been derived, indicating that renewable sources of electricity on average have lower greenhouse gas and similar material intensities as non-renewable electricity sources. By applying a production layer decomposition it has become evident that renewable sources exert higher greenhouse gas pressures in upstream activities, whereas both renewables and non-renewables show similar patterns of material usage throughout the whole global supply chain. Finally, an applied policy intervention model showed that depending on the policies for a transition to a more renewable electricity mix, the overall environmental impacts vary, but generally the environmental pressures are reduced.

Keywords: Electricity Transition, Renewable Electricity, Input-Output Analysis, European Union, Environmental Impact Analysis, Global Supply Chains

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List of abbreviations

CBA	Consumption-Based Accounting
EE-MRIOT	Environmentally Extended Multiregional Input-Output Table
EU	European Union
FD	Final Demand
FP	Footprint
GHG	Greenhouse Gases
GSC	Global Supply Chain
IOA	Input-Output Analysis
PBA	Production-Based Accounting
PLD	Production Layer Decomposition
RMC	Raw Material Consumption

1. Introduction

In 2009 the European Union announced their flagship initiative “Europe 2020” to reduce greenhouse gas emissions by 20% compared to the levels in 1990, increase the share of renewable energy sources to 20% and increase energy efficiency by 20% (CEC, 2010). Hence, transitions towards renewables became an important indicator for an improved European path towards sustainability. With new and more ambitious targets from the “Clean Energy for all Europeans” initiative the EU is trying to become the global leader in sustainable growth by fostering capacities and investment in the use of renewables and green technologies by exploiting the interconnection and scale networks of the EU (Lowitzsch, Hoicka, & Van Tulde, 2020). While most supranational policies focus on energy and energy transitions, most end users are interested in the transformed primary energy, otherwise known as electricity (Cullen & Allwood, 2010). While primary energy is further used for transport and heating, solely focusing on the electricity sector is motivated by the fact that an electrification of the heating and transport sector is required to reduce the environmental impacts, with that transition only making sense if the electricity generation itself is transitioning towards renewable solutions (Mac Domhnail & Ryan, 2020).

At the same time, interconnectivity and dependencies between different sectors have become more important resulting in the share of global environmental pressures embodied in international trade growing from 23% to 32% between 1995-2011 (Wood et al., 2018a). Therefore, environmental consequences should be tackled via a consumption perspective accounting for pressures exerted along the complete global supply chain (Wood et al., 2018a; Ansari, Haider & Khan, 2020). Global supply chains (GSC) represent a sequence of all suppliers involved in the production of a final good tracing all production steps from downstream to upstream (Matthews, Weber & Hendrickson, 2008). Upstream defines activities close to the extraction of material and resources, hence the beginning of the supply chain, whereas downstream are the value adding activities to create a final product, such as power generation in the case of the electricity sector (Singer & Donoso, 2008). Generally, GSCs are a two-sided sword in terms of their environmental impact. Due to the facilitation of global production processes, new environmentally friendly goods such as solar panels can be produced faster and cheaper, whereas the increased trade through a scale effect will automatically have negative impacts on the environment (World Bank, 2019).

Improved methods and data have allowed to continuously improve the analysis of environmental pressures via the creation of various footprints (FP), taking the direct and indirect pressures into account for a specific product or product group (Minx et al., 2009). Direct pressures are emissions or materials included in the power generation activity itself, whereas indirect pressures measure the intermediate deliveries and services to enable and sustain the power generation facilities, ranging from the manufacturing of sub-components to the transportation of raw materials to the power station (Hertwich et al., 2015). More general studies were applied to the energy sector such as Allan et al. (2007) for a

Scottish energy transition towards renewables sources or reducing global carbon emissions by improving energy efficiency (Minx et al., 2009). However, limited studies have tried to understand the environmental consequences of electricity generation in a globalized context. The only extensive studies conducted, breaking down the electricity sector into renewable and non-renewables sources of electricity, were analysing the water FP (Mekonnen, Gerbens-Leenes & Hoekstra, 2015), as well as the air pollutants in distinct phases of electricity (Proops et al., 1996). Bjelle et al. (2021) find that transitioning towards a more renewably sourced electricity mix is seen as a viable option to decarbonize the economy due to the large technological potential to reduce future emissions. Yet, due to the importance of dematerialization (Barrett & Scott, 2012) the material extraction further needs to be considered (Wiedmann et al., 2015), especially in the case of power generation, with a relatively high up-front investment taking place for renewable sources of electricity (Hertwich et al., 2015). With an increasing globalized world and increasing demands for electricity, the question arises:

What are the environmental impacts of electricity consumption of renewable and non-renewable sources along the global supply chain?

Thus, this study aims to improve the understanding of the direct and indirect consequences of electricity production along the complete global supply chains. In order to derive insights into the posed question, this work makes use of a newly available update of the environmentally extended input-output table *Exiobase 3* (Stadler et al., 2021). The database allows to breakdown the electricity sector into 12 distinct electricity products, defined by the source of primary energy used, ranging from solar photovoltaic to coal. This thesis contributes to the academic debate by improving the limitations of singular product life-cycle analysis in measuring all involved environmental impacts by applying three distinct input-output methods to the electricity sector. First, by aggregating the categories into renewable and non-renewable sources, standardised consumption-based material, based on two subcategories metal ores and minerals, and greenhouse gas FPs are calculated for the electricity consumption of the European Union. Optimally a combination of both production-based and consumption-based accounting methods would be used (Liu et al., 2018), but given the scope of this work the focus will be put on assigning the emission emitted and material extracted responsibility to the final consumption. Additionally, by taking the FP intensities the different power generation technologies can be compared overtime. Second, to advance the academic debate on electricity transition the analysis further decomposes the production layers of both renewable and non-renewable sources, allowing to assign at which stage along the supply chains the environmental pressures are exerted. This contributes to a better understanding of the supply chain complexity, the indirect consequences of an electricity transition, as well as making a differentiation between renewable and non-renewable sources. Third and final, due to the higher expected up-front investment costs of renewable sources of electricity, this work aims to provide a policy analysis tool, allowing to compare the environmental consequences of different electricity mixes. This is done by creating a what-if scenario model in which non-renewable electricity is reduced and

compensated by renewable sources, allowing to better understand the tension between current higher environmental impact and future gains from a changing electricity mix.

The rest of the paper is organised as follows. Section 2 will provide an overview of the relevant literature and the theory, concluding with the derived hypotheses. Section 3 will introduce *Exiobase 3* the database used for the analysis, including its structure and limitations. Section 4 will introduce the methodology used in environmentally extended input-output analysis and the methods applied in the empirical analysis of section 5, where the insights from the results will be discussed in detail, before concluding the work in section 6.

2. Literature Review

The literature review will provide an overview of the electricity generation sector as well as the transition to renewable electricity sources. As environmental pressures continue to increase and climate change mitigation has become a top priority in policy and academia with the United Nations Kyoto Protocol (Neij & Åstrand, 2006), energy and electricity policy will be discussed. Furthermore, the current findings on the environmental impacts of the electricity mix are presented, as well as the academic approaches on how the former can be measured and analysed in the context of a globalised economy. Finally, based on theoretical approaches and findings from the literature, the hypotheses of this thesis are presented.

2.1. Electricity, Transitions and Policies

Electricity generation or electricity production can be defined as the process of converting primary energy sources into electricity that can be converted for final consumption, accounting for about 30% of total energy consumption (Cullen & Allwood, 2010). Renewable sources are defined as hydropower, geothermal, solar, wind, tidal and wave, biomass and waste, while non-renewable sources include coal, gas, nuclear and petroleum sources (OECD, 2021). In this context, electricity transition refers to a switch in the electricity mix from non-renewable to renewable sources of electricity. The clear pressure for such a transition becomes apparent when one considers studies such as Ang, Zhou and Tay (2011), which analyse the potential economic, social and environmental benefits of switching the electricity mix. By applying a benchmarking analysis, the authors show that improving global electricity generation efficiency levels to developed country standards would result in an 8% reduction in global carbon emissions. More importantly, the potential reduction in carbon emissions is even greater if countries decide to switch to non-fossil fuel electricity generation, implying that even countries with already low emissions have a large potential for future reductions by switching to non-fossil fuel electricity generation (Ang, Zhou & Tay, 2011). Therefore, achieving carbon emission reductions in the power sector requires a radical transition to renewable sources of electricity (Foxon, 2013).

Borenstein (2012) argues that the reason for the transition to renewable electricity is the unpriced externalities caused by fossil fuel pollution. The author goes on to state that the pricing of both renewable and non-renewable technologies, including their market and environmental prices, is extremely difficult and is generally judged by "the preference of the decision maker" (Borenstein, 2012, p. 68). In the case of electricity generation, Foxon (2013) notes that any transition will be a political process due to the trade-offs between energy security, economic prosperity, affordability and reducing environmental impacts. Therefore, understanding the environmental externalities of all technologies in power generation is key to predicting the impacts of an electricity transition and setting the right policy framework.

The 2010 Industrial Development Report (UNIDO, 2011) highlights three potential strategies to reduce the environmental impact of industrial energy use through: 1) increasing energy efficiency, 2) changing fuel sources, and 3) improving production technology. In the EU, the European Energy and Climate Change Package in 2009 (CEC, 2010) introduced energy transition policies that put pressure on national governments to meet the target of 20% of total energy from renewable sources and also to set their own emission reduction targets. Such national policies include the UK's commitment to close coal-fired power plants (Foxon, 2013), Germany's *Energiewende* away from nuclear towards renewables (Quitow et al., 2016) or Scotland's plan to close both coal and nuclear power plants simultaneously (Allan et al., 2007). National policies have been shown to lead to a large divergence in the share of renewable electricity in different member states (Mac Domnhail & Ryan, 2020). Therefore, the EU is increasingly taking a top-down approach and trying to harmonise renewable electricity efforts to achieve updated targets such as 27% of total energy from renewable sources (del Rio et al., 2017). A large part of the Europe 2020 strategy is to enable sustainable growth by improving dematerialisation and reducing the environmental impacts caused by economic growth, which is why the EU Commission launched its flagship initiative 'Resource Efficient Europe' (CEC, 2010). The new initiative seeks to decouple economic growth from resource use by moving to a low-carbon economy, as well as to grow within planetary boundaries. Finally, in 2019, the EU adopted a new initiative called Clean Energy for All Europeans, which sets new and more ambitious targets for the energy sector in line with the Paris Agreement. The targets include incentives for households to use more electricity from renewable sources and the creation of a legal framework that encourages further investment in green energy (Lowitzsch et al., 2020). Nevertheless, there are only a limited number of specific policies for renewable technologies. This may be due to large differences in the effectiveness and prices of renewable energy technologies, fears about the security of electricity supply and a variety of barriers to rapid electricity transitions.

Potential barriers to fast electricity transitions have been studied by Neij and Astrand (2006), who define both solar and wind power generation as socio-technical systems with a high technology intensity, requiring large investments in turbines and solar cells, collectors and batteries. This makes them

modular technologies, which favours the ability to scale up and switch power more quickly. Although scalable, both technologies are dependent on geography for efficient use, albeit to a lesser extent than hydropower and geothermal. The transmission of electricity from remote locations to the point of final demand is thus a barrier to scaling renewable sources (Bornstein, 2012). This makes the power sector interesting for analysis due to its interdependence with other infrastructure investments such as a European grid integration and energy storage capacities (Haas et al., 2015). Mac Domhnaill and Ryan (2020) found that improving the electricity grid interconnection and increasing environmental pressure from emissions are important drivers for increased renewable electricity development in the European Union. Improving the European grid and electricity network has therefore become a priority in EU policy towards a green economy (CEC, 2010, Lowitzsch et al., 2020). However, Mac Domhnaill and Ryan (2020) go on to argue that investment in renewable energy is strongly linked to a country's wealth, as only richer countries are able to afford the initial costs. This was originally seen as a barrier in the case of solar power generation (Trieb, Langniss & Klaiss, 1997). However, with increasing production capacity, better integration into grid systems and improved commercialisation, the authors hypothesise that the advantages of mass production of photovoltaic solar power, combined with increasing environmental pressure, will prevail. Indeed, innovation has been driven by rapid investment in research, resulting in continuous improvement in photovoltaic cell technology and huge improvements in production costs and solar cell efficiency (Hosenuzzaman et al., 2015). In general, Bjelle et al. (2021) expect a future decrease in GHG emissions due to rapidly improving technologies, which is especially true for the electricity sector. This offers the EU a unique opportunity to become the global leader in renewable electricity technology (European Commission, 2020). However, while the transition to renewable energy is increasingly promoted by the EU, there seems to be a bias to simply measure direct CO₂ emissions in the electricity sector (Lowitzsch et al., 2020).

2.2. Environmental Consequences of Electricity Generation

In general, all sources of electricity generation are harmful to the environment. These include factors such as harm to birds from wind turbines and threats to wildlife habitat from the construction of solar power plants, or simply oil spills from oil and gas production. However, due to the relevance of electricity generation and economic growth (Karanfil & Li, 2015), this paper analyses the environmental impacts of electricity generation by looking at resource extraction, focusing on the two material categories of metal ores and minerals, as well as the greenhouse gases emitted, in order to derive policy interventions that do more good than harm (Bornstein, 2012). The potential environmental impacts of the electricity sector were investigated by Hertwich et al. (2015) by applying a life cycle assessment in a scenario-based comparison. The authors find that renewable electricity technologies generally require higher upfront investments in infrastructure as well as higher material requirements per unit of electricity generated. This is particularly true for bulk materials such as iron and copper for solar photovoltaic and wind power plants. Although the upfront investment of

renewables is estimated to be high, the future scenarios and electricity generation lead to decarbonisation and are thus an important part of climate protection. In this context, Haas et al. (2015) further emphasise the importance of taking a holistic view in future research when analysing the environmental impacts of electricity transition due to the interaction between different technologies and their production as well as the complexity of supply chains. Today's international trade structure with increasing trade of intermediate products requires an analysis of the environmental costs of products such as electricity generation in the context of GSCs (Tukker & Dietzenbacher, 2013). This is also evidenced by the fact that the growth rate of globalisation has overtaken the growth of GDP, so that the environmental impact of increasing consumption is nowadays more often taking place abroad (Tukker, Giljum and Wood, 2018). Possible ways to analyse the direct and indirect environmental impacts of electricity generation are presented in the next section.

2.2.1. Environmental Impact Assessment

Historically, the first analyses to take GSC considerations into account looked more closely at carbon emissions from the production of goods, but only a limited number of studies and protocols " pursuing the broadest scope boundaries including a full range of their supply chain emissions " (Matthews et al., 2008, p. 1). Currently, two broad methodologies are used to consider the interrelationships in understanding the environmental impacts of consumption, namely life cycle analysis and input-output analysis.

2.2.1.1. *Life-Cycle Assessments*

Life cycle assessments (LCA) have been applied by environmental scientists to specific products, projects or industries for years and allow tracking the specific requirements of a product throughout its life cycle, from raw material extraction to production and recycling (Hertwich et al., 2015). Given the large supply chains, tracking all direct and indirect environmental impacts for a given product becomes a data-intensive affair, so often not all intermediate supplies can be accounted for (Joshi, 1999). This is commonly known as truncation error, where LCAs truncate upstream supply chains in complex value chains to facilitate analysis (Giljum, Bruckner & Martinez, 2015). A specific case study analysing the environmental costs of solar energy technologies was conducted by Tsoutsos, Frantzeskaki and Gekas (2005), who find significant environmental benefits compared to traditional electricity sources. Nevertheless, the authors show that the production of solar electricity from photovoltaics in particular is energy-intensive and requires a large amount of materials to produce the components such as the photovoltaic cell. Thus, although solar power is considered a low GHG emitting technology, the depletion of natural resources needs to be taken into account. Another LCA by Mahmud, Huda, Farjana and Lang (2020) examines the environmental impacts of renewable electricity sources in the US. The authors found that most emissions occur during the extraction of materials and their transport, the manufacture of components and the installation of the electricity generation system. The authors rated solar photovoltaic systems as the best option from an end-to-end perspective due to low GHG emissions.

Hydropower, on the other hand, is energy-intensive due to the pumping of water into storage and therefore more emissions-intensive, while biomass was found to be the most materially expensive renewable source. Finally, a similar pattern was found over the life cycle for all renewable electricity generation plants, starting with the extraction of raw materials from mines, the transport of these materials to the manufacturing plants that produce the components needed for the power plants, the transport of these materials to the electricity plants, the installation, the electricity generation and finally the disposal of the waste (Mahmud et al., 2020). Overall, LCAs measure very specific products for a very specific geographical area, which makes it difficult to generalise the results and understand global consequences.

2.2.1.2. Input-Output Analysis

To circumvent the problem of truncation and the limited geographical scope of LCAs, input-output analysis (IOA) can be applied to include all supply chains for a given sector or product (Matthews et al., 2008). This is done by calculating the marginal demand or marginal environmental impact resulting from increased final demand (FD), derived from environmentally extended multi-regional input-output tables (EE-MRIOT), which combine all sectors in all countries into a single table (Leontief, 1970; Miller & Blair, 2009). Using such methods, the electricity generation process can be further divided into three different phases, namely construction, operation and decommissioning, as done by Proops et al. (1996). By tracking air pollution through all economic activities, they found that in the case of the UK, the greatest reduction in emissions can be achieved in the operational phase of power plants. In their analysis, they show that already in 1996 economic activity is heavily dependent on inter-industry trade, so that the electricity sector is dependent on steel, non-ferrous metals, electronics, chemicals and other industries. Similarly, Jafar, Al-Amin and Siwar (2008) analysed the environmental impact of changing the electricity mix in power generation in Malaysia. By applying an environmentally extended input-output framework, the authors found that depending on the diversification strategy away from gas towards coal and hydropower, total CO₂, SO₂ and N₂O emissions would increase if no simultaneous efficiency improvement is achieved. An extensive IOA study on the water consumption of electricity generation was conducted by Mekonnen, Gerbens-Leenes and Hoekstra (2015). The analysis of electricity based on different sources showed a large heterogeneity between the different categories. Here, wind, solar photovoltaic and geothermal have the lowest water consumption, while biomass and hydropower have the highest, with electricity from fossil fuels and nuclear power in between the extremes. The results suggest that the transition to renewable energy based on wind, solar and geothermal, away from fossil sources, will have the greatest impact on the global water FP (Mekonnen et al., 2015). Allan et al. (2007), analysing the potential Scottish transition to renewable energy sources, conclude that it is important firstly to disaggregate electricity generation technologies and secondly to adopt a systems approach to consider direct and indirect environmental, social and economic impacts.

Input-output analysis has enabled global assessments, however, they are limited by the fact that they do not cover the end of life of products, namely recycling and reuse.

Optimally, a combination of LCA and IOA would be appropriate; the closest theoretical method that incorporates aspects of both is footprint (FP) analysis, which is presented in the following sections. Prior to this, an important aspect of environmental impact studies is introduced: the allocation of responsibility for emitted emissions and extracted material. Currently, there are two accounting methods, a production perspective and a consumption perspective.

2.2.2. Consumption- and Production-Based Accounting

To date, international standards set by environmental agencies and protocols have focused on capturing the direct impacts of production, thus ignoring the international impacts of a nation's production (Lenzen et al, 2003). In the case of complex value chains, deciding who is responsible for the environmental damage associated with the production of the final product is the subject of much debate on GHG and material accounting (Matthews et al, 2008). Based on an extensive literature review, two accounting methods are gaining acceptance in relation to GHG mitigation policy: Production-based accounting (PBA) and consumption-based accounting (CBA). PBA is a GHG accounting method recognised by the Intergovernmental Panel on Climate Change (IPCC) that accounts for emissions produced within the territory of a given national jurisdiction (Liu, Huang, Baetz & Zhang, 2018). CBA, on the other hand, is a method based on multiregional input-output analysis, which assigns responsibility for produced emissions to a nation's consumption, i.e. a nation's final demand (FD) (Liu et al., 2018). Understanding the difference between CBA and PBA makes it possible to examine whether carbon or material leakage occurs between countries or regions, where leakage refers to the outsourcing of polluting activities or production steps to other countries (Liu et al., 2018). This means that domestically improved technological progress does not automatically lead to improved sustainability, due to the mix of foreign and domestic products that are consumed (Wieb et al., 2019). Choosing the right accounting method could therefore have far-reaching consequences, such as meeting the EU's Green Deal ambitions of reducing GHG emissions by 55% by 2030 or achieving climate neutrality in 2050 (European Commission, 2020).

Peters, Minx, Weber and Edenhofer (2011) have initiated an important debate regarding the tracking of emissions embodied in trade. They show that when CBA is applied, developed countries have outsourced their emissions to developing countries, thereby reducing their territorial emissions. The authors call this phenomenon emission leakage and introduce the concept of “balance of emissions embodied in trade”, clearly showing a “spatial disconnect between the point of consumption and the emissions in production” (Peters et al., 2011, p. 8907). While most debates refer to carbon or greenhouse gas emissions, the accounting debate can also be extended to the extraction of materials and other resources. Kander, Jiborn, Moran and Wiedmann (2015), in response to Peters et al. (2011), argue that

the debate on CBA and PBA should be resolved by finding a measure that incentivises countries' policies to address global emissions. According to them, CBA does not assess a country's efforts to make its exports more carbon-efficient, as the trade balance would indicate emissions leakage due to the import of more carbon-intensive products. Therefore, the authors propose the use of a technology-adjusted CBA that takes into account the different carbon efficiencies in the export sector to see how national policies affect global emission levels (Kander et al, 2015). The new measure was used to analyse whether there has been an actual decoupling of economic growth and emissions in the UK and Sweden, and showed that no absolute decoupling is observed (Jiborn et al., 2018).

2.2.3. Greenhouse Gases and Material footprints

Following the definition of Minx et al. (2009) FPs represent the direct and indirect environmental pressures exerted to satisfy a final demand of a product, product group or an economic activity assessing the whole year. The timeframe of a year is based on standard national assessment boundaries and the data coverage per year of most known EE-MRIOT (Minx et al., 2009). As such a FP is a consumption-based method taking complete GSCs into account. In this work, two FPs are predominantly used: the GHG FP and the material FP. While there is a wide range of FPs, their application also varies considerably.

One of the most commonly known FPs is the so-called carbon FP, measuring the CO₂ emissions for a specific FD. Arto and Dietzenbacher (2014) used the method to analyse the total global GHG FP. The author found that the main driver of the global carbon FP was changes in consumption levels, while improvements in technology and changes in the composition of consumption reduced global levels the most. Similarly, Liu, Guo and Xiao (2019) found that improved technologies and innovations led to improved emission intensities and thus reduced carbon FPs. They also point out that the aggregate sector of electricity, gas and water supply was the main driver of carbon FP growth between 1995 and 2009. While Minx et al. (2009) found, by applying a two-factor decomposition of the carbon trajectories of product groups, that electricity generation experienced a decline in absolute CO₂ emissions between 1992-2004, the largest shift of all sectors caused by technological change was partially offset by increasing FD for electricity. Giljum, Bruckner and Martinez (2015) analyse raw material consumption (RMC), also known as material FP (MF), between 1997 and 2007 and find that while domestic material consumption has stabilised or even declined, national RMCs have increased. This clearly shows that upstream material consumption in international trade needs to be included in a national assessment. This goes hand in hand with Anasari, Haider and Khan (2020), who used a material FP to show that increased energy consumption increases the global material FP, even though some countries have managed to reduce their domestic material extraction. Both studies thus use the material FP as an indicator of pressure on natural resources. Furthermore, Wiedmann et al. (2015) show that the current headline indicators for resource efficiency in the EU's flagship initiatives do not take upstream raw materials into account and thus truncate the supply chain, leading to misleading assessments. While

material flow analyses have often been conducted for national aggregates, Giljum et al. (2016) aimed to improve the policy applicability of FP assessments by applying a production layer decomposition for 163 industries. Their analysis showed an increasing dependence of the EU on direct material imports as well as on materials embodied in imported products, with extraction of materials within the EU declining dramatically between 1995 and 2011. Their analysis provides interesting findings, such as the high material FP of the services sector of about 25%, which is due to the complex supply chain structures and relatively higher material extractions in upstream activities. In general, Giljum et al. (2016) show that supply chains have become more complex over this period, with the importance of more upstream production steps increasing, especially for metal ores, while minerals tend to remain close to the final downstream activities.

While GHG and material FPs can be calculated and analysed separately, Barrett and Scott (2012) bring the approaches together to see the impact of dematerialisation and thus increasing resource productivity on GHG emissions and to measure the macroeconomic effectiveness of potential policies. By applying a range of scenarios, the authors find that "consumption-side strategies are able to influence a wider range of emissions" (Barrett & Scott, 2012, p. 306). After analysing a wide range of FPs, Wood et al. (2018a) note that material extraction is the only indicator that grew not only relative to GDP but also in absolute terms, so no decoupling is apparent, distinguishing it from other environmental impacts such as GHGs. Not focusing solely on GHG emissions can therefore provide a more comprehensive picture of the environmental damage caused by electricity consumption.

2.3. Hypothesis

Given the high interdependence of the European economies, reforms in one country will affect the economy in another one (CEC, 2010). Thus, the hypothesis will considerate the EU's final consumption of electricity. While solar photovoltaics and wind are considered to be the best alternative to fossil-fuel electricity, due to their scalability (Hosenuzzaman et al., 2015; Mahmud et al., 2020), the focus will be put on making a clear overall distinction between all renewable and non-renewable sources, as will be explained in section 3.1.. Materials studied in this thesis include two sub-categories, namely metal ores and non-metallic minerals. The choice is motivated by the fact that metal ores make up roughly 10% of global material extraction and minerals are the material with the highest extraction growth rates (Haas et al., 2015). As both types have a relatively low level of circularity studying the dependence on these materials will allow to judge the environmental impacts of power generation. Finally, this work will apply the internationally recognised consumption-based-accounting methodology (Liu et al., 2018), thus taking a FD rather than a production perspective in understanding the environmental consequences of the EU's electricity mix.

Based on the findings of prior environmental impacts studies the distinction between greenhouse gases and material in the electricity sector is important. Given the higher initial investment rates and the

building of the infrastructure for renewable electricity sources (Haas et al., 2015) and taking an EU-FP approach, the first hypothesis is derived as:

H1: Renewable electricity sources have higher material FP intensities and lower greenhouse gas FPs intensities than non-renewable electricity sources.

Given the higher complexity of producing electricity based on wind, solar, or other forms of renewables, most of the emissions and materials included are deemed to take place in the manufacturing and production part of the supply chain, as has been seen for the case of solar photovoltaics (Tsoutsos et al., 2005). Given the improvements in technology overtime, as well as the higher rate of globalization the next hypothesis is derived as:

H2: Renewable electricity sources exert a higher dependence on emissions and materials included in more upstream production steps of global supply chains, with an increasing effect over time.

Previous research has shown that there are two important aspects to consider when analysing energy transformations. First, environmental costs may be hidden in layers further down the value chains, and second, upfront investment costs may lead to initial increases in environmental impacts due to the expansion of production capacities (Borenstein, 2012; Hertwich et al., 2015). Consumption-based policies can exert sufficient pressure leading to changes in the production structure via the type of products demanded (Wood et al., 2018b). Given the large potential to reduce emissions by improving the emissions intensity of the electricity sector (Liu et al., 2019), and following EU and national policies regarding the transition to a new electricity generation mix, the final hypothesis is derived as follows:

H3: Changing the electricity consumption composition by transitioning towards renewable electricity sources will reduce the EU's environmental FP.

3. Data

3.1. Exiobase 3

The database used for this analysis is *Exiobase 3*, an environmentally extended multi-regional input-output table (EE-MRIOT), specifically developed in order to assess the EU's environmental impacts (Giljum et al., 2016). EE-MRIOTs merge bilateral trade data via monetary intersectoral flows into a single global table keeping the international trade structure intact (Giljum et al., 2016). Therefore, MRIOTs depict the whole global economy and are avoiding the common problem of double counting traded goods in imports and exports (Giljum et al., 2016). Hence, a key feature of global EE-MRIOT is that they guarantee global consistency, in a sense that by taking the producer or consumer perspective the aggregated emissions or resource extractions will add up to the same global total, thus guaranteeing a solid accounting framework (Tukker, Giljum & Wood, 2018). Thus far, an important limitation of EE-MRIOT was the high level of aggregation necessary to create concise tables matching international trade, environmental and other economic data. The aggregation might not have posed a problem for understanding global trade patterns, yet in the case of environmental analysis a detailed sectoral disaggregation is necessary, such as breaking down the electricity generation sector into the sub-components ranging from fossil-fuel to nuclear and wind power generation (Tukker & Dietzenbacher, 2013). With EE-MRIOT generally being a track record of a single year, to better understand time-trends the disaggregated tables need to be extended to a time-series, allowing to understand the dynamics of environmental pressure over time (Minx et al., 2009). While there is an increasing number of EE-MRIOT such as the WIOD, EORA, GTAP and Exiopol with different extensions, geographical coverage, computation methods and time-scopes (Tukker & Dietzenbacher, 2013), up to this day, the only database fulfilling all three desired requirements is *Exiobase 3* (Stadler et al., 2021). *Exiobase 3* thus “has the largest potential to provide a consistent accounting framework to calculate a variety of different FP indicators” (Tukker, Giljum & Wood, 2018, p.489).

Exiobase 3 is a result from the EU funded DESIRE project, which was set up with the goal of creating a new database, which allows to conduct environmental impact assessments of the EU compatible with the System of Environmental Economic Accounting (Stadler et al., 2013; Wood et al., 2015).¹ Contrary to earlier versions, the construction of *Exiobase 3* followed a top-down approach first harmonizing global trade data and then imposing these structures on the supply and use tables of the 49 country and regions included, making them consistent with UN macroeconomic statistics (Tukker, Giljum & Wood, 2018). By following this approach, the problem of truncation of traditional LCAs is avoided by including all available supply chains (Giljum et al., 2015). For environmental analysis the underlying national monetary supply-use data and trade structures further need to be matched with the

¹ DESIRE – Development of a System of Indicators for a Resource efficient Europe is a project of the European Union as part of the seventh framework programme, in which the continuation and improvement of the existing versions of Exiobase 1 and 2 were funded (Tukker, Giljum & Wood, 2018).

technological and environmental statistics from a range of environmental accounts from a variety of external sources, assigning the environmental indicators to specific product categories (Stadler et al., 2013).² The development of *Exiobase 3* further introduced constant prices, thus allowing to make a time-series comparison covering the period of 1995-2011 (Tukker, Giljum & Wood, 2018). With the newest available update 3.8.1 the available time-series was extended up to 2019, based on now-casted data extrapolating the latest available data points for different macroeconomic accounts with the original tables and the continuously updated environmental extensions (Tukker, Giljum & Wood, 2018; Stadler et al., 2021).³

The structure of *Exiobase 3* for a simplified version of three regions in a single year is represented in *Figure 1*. The database follows the general layout of EE-MRIOT consisting of the intermediate input matrix, or interindustry trade matrix, in monetary terms represented in red. More specifically the values of each element represent the monetary unit equivalent in million € (Stadler et al., 2021). The matrix is inherently quadratic with the dark-coloured squares representing the domestic input-output tables for each region and the light-coloured squares the bilateral trade tables between the respective regions. In the context of *Exiobase 3* each square would represent a matrix of 200 rows times 200 columns representing the intermediate trade between each product category.⁴ Intermediate inputs in the case of electricity can contain anything from transport equipment or sub-components of a solar cell. The second main element, in green, represents the FD (FD) from the three regions for the domestically produced products (dark-green squares) and imported products (light-green squares). In this example, the FD categories household, non-governmental organisation and government consumption were summed into a single vector for each region.⁵ The blue matrix represents factor inputs such as wages or taxes in order to calculate total value added in the production. The purple matrix represents the environmental extensions of production. For each product in each region the environmental indicators are listed in their respective physical units, in the case of GHG in kilogram of emission equivalents and for materials in kilotons extracted. Finally, the yellow matrix represents the environmental extension from final use, thus including the direct environmental pressure exerted by consumers (Stadler et al., 2013).

The final yearly tables between 1995 and 2019 cover roughly 95% of the total global economy with a main focus on detailed country level data for the 28 EU members, 16 major economies including China, the USA, India and Brazil, and five aggregated rest-of-the-world regions (Stadler et al., 2013; Stadler

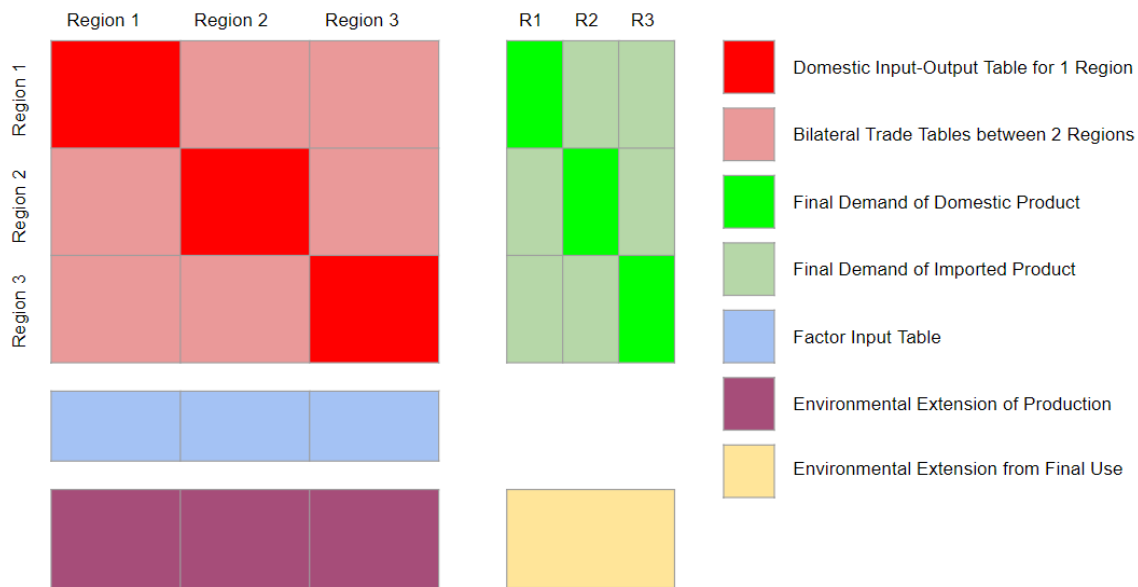
² For more details on the exact computation methods, the harmonization of the monetary trade tables with the environmental accounts and the underlying calculations please refer to Stadler et al. (2013).

³ All new versions of *Exiobase 3* are freely and publicly accessible on the hosting platform zenodo: <https://zenodo.org/record/4588235>

⁴ *Exiobase 3* is available in two formats: an industry by industry and a product by product table, in this thesis the product by product table was preferred due to the disaggregation of the electricity generation.

⁵ The final demand categories in *Exiobase 3* consist of households, non-profit organisations, government, gross fixed capital formation, changes in inventories, changes in valuables, as well as exports. The sum for the analysis was taken for the first three categories, representing the domestic actual consumption.

et al., 2021). To improve the readability, the results in section 5 will be aggregating the 28 EU countries into a single region, allowing for a complete assessment of the EU. The economic sector of interest is the power generation sector which is disaggregated in the following 12 sources of electricity production: coal, gas, nuclear, hydro, wind, petroleum and other oil derivatives, biomass and waste, solar photovoltaic, solar thermal, tide wave and ocean, and geothermal (Stadler et al., 2021). The focus thus lies on the final power generation products excluding the two other energy categories transport and heating (Mac Domhnail & Ryan, 2020).



Author's own illustration based on Stadler et al. (2013)

Figure 1 - Illustration of a simplified input-output table: Exiobase 3

The environmental satellite account data of interest in this analysis consists of three categories: greenhouse gases, metal ores and minerals.⁶ While materials normally are decomposed into biomass, fossil fuels, minerals and metal ores, this work is solely focusing on metal ores and minerals, as introduced in section 2.3. With carbon dioxide (CO₂), methane (CH₄) and nitrous oxide (N₂O) making up the largest share of the six GHGs defined in the Kyoto protocol (Liu, Guo & Xiao, 2019) they will be used to analyse GHG in this work. While originally presented in kg of emissions, in this work the unit was transformed into kilotons of emissions to improve readability. Metal ores include the following categories: Bauxite and aluminium, copper, gold, iron, lead, nickel, other non-ferrous metal, PGM, silver, tin, uranium and thorium, and zinc. Minerals consists of: Building stones, chemical and fertilizer minerals, clays and kaolin, gravel and sand, limestone and chalk, other minerals, salt, and slate. *Exiobase 3* would further allow to disaggregate all materials into used and unused, again in order to simplify the analysis only the used materials in either production or final use will be considered.

⁶ Exiobase 3 includes a total of 1112 unique environmental indicators measured in their respective units, allowing for detailed environmental science assessments (Stadler et al., 2013).

3.2. Data Constraints

A disadvantage of EE-MRIOT is the dependence on available international data and the environmental extensions, by applying multiple assumptions and accounting methods, consequently the final tables deviate slightly from national statistics (Giljum et al., 2015). As EE-MRIOT are based on a large amount of different sources needing to be harmonized, these tables generally result in some uncertainty. This becomes evident if the same indicators are derived based on different databases, resulting in different outcomes, thus more consistent and robust tables are desired, achievable through improved standardization of the data harmonization process (Rueda-Cantuche et al., 2018). Nevertheless, the degree of precision and uncertainty of EE-MRIOT is deemed acceptable and representative for explorative studies in a globalized context (Hawkins et al., 2007). A second disadvantage of working with *Exiobase 3* is that the tables are static, thus not differentiating between stocks and flows of goods, emissions or materials. While a time-series is available, one cannot model consumer-behaviour based on utilities and elasticities, resulting in what-if scenarios in which the intervention scenario is compared with a baseline scenario to draw comparison (Wood et al., 2018b). This is further due to the assumption of linearity in the construction of EE-MRIOT, in which constant returns to scale are taken as given, so to assign increased inputs to outputs consistently (Hawkins et al., 2007). Finally, while the time series is conducted using constant prices and the macroeconomic accounts continue until 2019, the availability or rather the delay of publication of environmental extension data resulted in the fact that depending on the category the data had to be now-casted, using a large set of estimation methods, for the most recent years. Given the active and frequent updates of the versions, newly available real-data points are incorporated into the final tables on a continuous basis, improving the data validity with each version. The results must therefore be carefully interpreted taking the mentioned limitations into consideration.

Nevertheless, the key advantages of *Exiobase 3* compared to other databases with its detailed level of products, the large environmental accounts, the time-series ranging from 1995-2019 as well as the EU centricity still makes it the most relevant and reliable data available for the proposed analysis. A final important advantage is that *Exiobase 3* follows the standardised EE-MRIOT as seen in *Figure 1* and thus allows to apply up to date input-output methods, which will be introduced in detail in the following section.

4. Methods

The following methodology is based on the environmentally extended input-output methods developed originally by Leontief (1970) and practically summarised in the handbook by Miller and Blair (2009). In a first step the standard methods when working with EE-MRIOT are introduced, using this methodological foundation the derivation of environmental FPs as well as two advanced assessments, namely production layer decomposition (PLD) and scenario-based modelling will be introduced. The three methods are structured in the same sequence as the hypothesis introduced previously in section 2.3.

4.1. Input-Output Analysis using EE-MRIOT

Input-Output Analysis (IOA) is a top-down macroeconomic approach, which allows to exploratively analyse economic interindustry dependencies. IOA can therefore be used to calculate ripple effects throughout the economy induced by external shocks. The basic IO tables and methods can easily be extended to environmental analysis by including so called environmental satellite accounts (Leontief, 1970; Lenzen et al., 2003). These accounts consist of environmental indicators that can be assigned to the specific sectors and products thus perfectly match the interindustry linkages. The advantage of environmental IOA is that it enables to calculate the total environmental impacts, throughout all economic sectors and all distinct supply chains, of a specific economic activity (Proops et al., 1996). Or as Lenzen et al. (2003, p.264-265) put it “this process of industrial interdependence proceeds infinitely in an upstream direction, through the whole life cycle of all products, like the branches of an infinite tree”.

To visualize this structure, IO tables are based on a set of linear equations summarising the global interdependencies between economic sectors (Joshi, 1999). Equations (1)-(6) describe the linear equations needed to derive the required model to understand the economic and environmental impacts of the power generation sector. In IOA the global economy is represented by (1) in which \mathbf{X} is a matrix of required inputs necessary to produce all intermediate and final goods in matrix \mathbf{Z} and \mathbf{Y} respectively (Matthews et al., 2008), with \mathbf{Z} being represented as the red-squares and \mathbf{Y} as green-squares in *Figure 1*. As basic accounting rules are fulfilled in IOA the vector \mathbf{X} simultaneously represents the total output generated (Miller & Blair, 2009). This split in equation (1) thus allows to model “the interdependencies between industries and within and between countries as well as between intermediate and final goods producers and consumers” (Wiebe et al., 2019, p. 6362). To understand the marginal requirements needed to produce an additional unit of a given product the technological coefficient matrix \mathbf{A} needs to be derived by dividing each element of the \mathbf{Z} matrix by the corresponding total output of that product group or sector. In matrix notation this is equal to multiplying the \mathbf{Z} matrix by the diagonalized inverse of the \mathbf{X} vector $\hat{\mathbf{x}}^{-1}$ (2). The equation can be rewritten in (3) to show the element wise derivation in

which \mathbf{z}_{ij}^{vw} equals to the monetary flow from sector i in country v to sector j in country w and \mathbf{x}_j^w is the total output of sector j in country w , resulting in the technical coefficient \mathbf{a}_{ij}^{vw} per unit of output (Giljum et al., 2015).

$$\mathbf{X} = \mathbf{Z} + \mathbf{Y} \quad (1)$$

$$\mathbf{A} = \frac{\mathbf{Z}}{\mathbf{X}} = \mathbf{Z}\hat{\mathbf{x}}^{-1} \quad (2)$$

$$\mathbf{a}_{ij}^{vw} = \frac{\mathbf{z}_{ij}^{vw}}{\mathbf{x}_j^w} \quad (3)$$

Using (1) and (2), one can break down the global total output in all intermediate requirements and FDs in equation (4). By solving for \mathbf{X} one can derive equation (5), the most commonly used equation in IOA to represent the global interconnection (Miller & Blair, 2009). $(\mathbf{I} - \mathbf{A})^{-1}$ is referred to as the Leontief inverse \mathbf{L} , capturing all direct and indirect inputs required to produce one monetary unit of FD (Leontief, 1970).⁷ The Leontief Inverse \mathbf{L} could thus provide information on the additional total output in all supply chain inputs required for the solar electricity purchased, therefore explaining the direct and indirect requirements for a specific or overall FD (Miller & Blair, 2009).

$$\mathbf{X} = \mathbf{A}\mathbf{X} + \mathbf{Y} \quad (4)$$

$$\mathbf{X} = (\mathbf{I} - \mathbf{A})^{-1} * \mathbf{Y} \quad (5)$$

$$\mathbf{L} = (\mathbf{I} - \mathbf{A})^{-1} \quad (6)$$

The prior derivation of equations builds the backbone for the following three specific applications and provides a sound methodological foundation for the empirical analysis.

4.2. Footprint Calculation

In order to calculate a specific FP one needs to use the environmental satellite accounts, where each element represents the unit of each indicator required to produce all products. Similar to equation (2) and (3), each element needs to be divided by the total output from that sector, resulting in an environmental indicator coefficient vector \mathbf{e} or matrix \mathbf{E} for multiple indicators (Lenzen et al., 2003). Such indicators include the CO₂ emissions or the amount of iron ores in kilotons per year which are being assigned to the consumption of a product, product group or sector (Minx et al., 2009), as an example for GHG emissions and material extracted defined in section 3. The basic FP is calculated by pre-multiplying equation (5) with the environmental stressor vector or matrix \mathbf{E} , represented by the first part of equation (7) ($\mathbf{E} * \mathbf{L} * \mathbf{Y}$), and by summing the direct environmental pressures of consumption, by pre-multiplying the \mathbf{H} matrix with a summation-vector consisting of ones \mathbf{i} (Wiebe et al., 2019). Part one of equation (7) thus allows circumventing the problem of not accounting for the upstream

⁷ \mathbf{I} represents the identity matrix with ones on the diagonal and the other elements being set to zero with the same dimensions as matrix \mathbf{A} (Miller & Blair, 2009).

environmental pressures of traded goods (Giljum et al., 2015), representing the red and purple matrices in *Figure 1*, whereas the second part uses the yellow square. If all environmental coefficients and total FD are included, as in equation (7) the global FP for all available 1112 indicators of *Exiobase 3* would be calculated.

$$FP = E * L * Y + iH \quad (7)$$

As the aim of this research is to capture the FP of the final EU consumption of electricity by focusing only on the environmental stressors defined in section 3.1., both the E and Y matrix need to be manipulated to link the environmental pressures from the beginning of the production to the end for a specific FD Y° (Giljum et al., 2015). In the case of E° only the 23 categories defined are kept in the final matrix. Depending on the desired analysis the matrix can be further simplified into a column vector e° by summing the GHG and material categories into a single category each. The manipulation of the FD matrix is done by setting the demand for all other countries outside of the EU and all other product categories other than the electricity products defined in section 3.1. to zero. By taking the transpose of e° the FPs of each EU country will be summed automatically to a single value. Equation (8) could be applied to tracing the global GHG emissions necessary to directly and indirectly satisfy the German demand for electricity generated by solar, or to trace the metal ores necessary to satisfy the EU's demand for electricity by wind technology. The second part of equation (7) has been left out, as in this empirical analysis, there were no direct GHG emissions emitted or materials extracted in the consumption, with all values equal to zero in matrix H .

$$FP^\circ = e'^\circ * L * Y^\circ \quad (8)$$

The FPs used are thus an absolute consumption-based indicator, resulting in a single value representing the size of the environmental impact in a given year. While the absolute values are interesting on their own from an economical perspective it is interesting to additionally see the FP intensities to facilitate the comparison across products, sectors and especially for technological improvements over time (Giljum et al., 2016). The FP intensity can be calculated in various ways by dividing by population, GDP or as in the case of this thesis the monetary value of the sum, pre-multiplying with a one-vector i , of the manipulated total FD Y° , see equation (9).

$$Intensity^\circ = \frac{FP^\circ}{iY^\circ} \quad (9)$$

By calculating both absolute FPs and their intensities a rough overview of difference between the final EU consumption of overall electricity, a differentiation between renewable and non-renewable electricity generation, as well as all 12 electricity generation sources can be drawn. While the method would allow to answer the first hypothesis, it does not allow to make any inference into the complexity and more detailed structure of where and how much environmental pressures are exerted along the GSC.

The next section will introduce a method that allows to decompose the production chain into distinct layers.

4.3. Production Layer Decomposition

The Leontief Inverse (6) can be rewritten algebraically in equation (10) by creating an infinite series to the power of the A matrix (Matthews et al., 2008, Miller and Blair, 2009).

$$L = (I - A)^{-1} = I + A + A^2 + A^3 + A^4 \dots A^\infty \quad (10)$$

By using the infinite series, a production layer decomposition (PLD) method can be applied to assess the structure and complexity of the underlying supply chains (Giljum et al., 2016). PLD is recognized as a simplified version of a structural path analysis as it does not weigh the paths but rather aggregates the environmental pressures to discrete layers of a GSC (Giljum et al., 2016). Thus, the equations (11) represent the applied PLD, where the supply chain is decomposed into so called layers, with each layer describing the distance to the production of the final goods, or in other words the “number of intermediate deliveries before a product or service ends up in FD” (Giljum et al., 2016, p.15). Such that layer 0 represents the environmental burden in the production of the final output, thus the last downstream step, with layer 1 including the environmental burden in the necessary direct inputs, also known as first level suppliers. Layer 2 would then include the environmental pressure embodied in the intermediate deliveries to the first level suppliers in layer 1. Thus, layers higher or equal than two represent the indirect environmental burdens via indirect higher level suppliers. The Leontief inverse is used concluding the infinite series of all layers, otherwise known as all the environmental coefficients in the total supply chain (Matthews et al., 2008).

If one takes a simplified version of a solar photovoltaic GSC the layer decomposition could be summarised in the following way by taking a downstream to upstream approach: Layer 0 is equal to the maintenance and production of electricity within the solar plant, in order for the plant to work the final solar panel modules installed need to be provided by layer 1, the module manufacturing centre, which in return depends on the deliveries from layer 2, the cell manufacturing centre, with the cells being a sub-component of the final solar module. In order to produce the cells in layer 2, intermediate deliveries of goods and materials from layer 3, the ingot and wafer manufacturing centre are necessary, which in return depend on deliveries of silicon from layer 4, with the silicon being produced using sand from the raw material extraction in layer 5 (Dehghani, Jabalameli & Jabbarzadeh, 2018). The overall environmental burden of solar photovoltaic electricity could be calculated in the last step of equation (11).

$$\begin{aligned} \mathbf{Layer}_0 &= \mathbf{e} * \mathbf{I} * \mathbf{Y} \\ \mathbf{Layer}_1 &= \mathbf{e} * \mathbf{A} * \mathbf{Y} \\ \mathbf{Layer}_2 &= \mathbf{e} * \mathbf{A}^2 * \mathbf{Y} \end{aligned}$$

$$\begin{aligned}
& \dots & (11) \\
\mathbf{Layer}_5 &= \mathbf{e} * \mathbf{A}^5 * \mathbf{Y} \\
\mathbf{Layer}_\infty &= \mathbf{e} * \mathbf{L} * \mathbf{Y}
\end{aligned}$$

In order to improve readability and reduce complexity, following Giljum et al. (2016), the layers higher than 5, thus relatively far upstream will be summed together by subtracting the sum of layer 0 to 5 from the infinite series of all layers, as stated in equation (12).

$$\mathbf{Layer}_{>5} = \mathbf{Layer}_\infty - \sum_{i=1}^5 \mathbf{Layer}_i \quad (12)$$

In the empirical analysis the PLD is applied to make a clear distinction between renewable and non-renewable sources of electricity, in order to see whether there are any clear patterns visible. Applying the method will allow to answer the second hypothesis, whereas for the third hypothesis the abstract level of analysis needs to be turned into a feasible policy application. A possible way to combine EE-MRIOT and policy making will be introduced in the next section.

4.4. Policy Simulation

Following a similar approach of Wood et al. (2018b) the FD can be manipulated by applying a potential policy outcome as introduced in equation (13). A policy outcome, for example, could be a 20% reduction in the use of non-renewable sources of electricity, resulting in an alternative electricity mix. The aim of the policy analysis tool is to compare two scenarios in which the first is based on the FD from the base year, here 2019, and the second is based on the manipulated FD representing the achieved policy allowing to do a ceteris-paribus comparison (Jafar et al., 2008). The ceteris-paribus condition translates into keeping the production technologies as well as the trade structures constant, in order to decompose the change of the FPs to the change in the consumption of the electricity mix. The baseline can also be considered the “business-as-usual” scenario that allows a so called “what-if” scenario comparison (Wiebe et al, 2019) with IO methods being judged an appropriate tool to analyse marginal, or even larger, consumption changes (Hawkins et al., 2007). Therefore, two scenarios need to be calculated prior to do a comparison. Equations (12) and (13) show how the FD vector \mathbf{y}^{int} after a successful policy application is derived. In order to set some boundaries to the policy scenario analysis, the FD categories are summarised into one single FD for the EU, summing up private household as well as governmental consumption, resulting in a row vector \mathbf{y} representing the baseline scenario (Usubiaga, Butnar & Schepelmann, 2018). In order to derive the policy scenario, the reduced FD is calculated by multiplying the original FD by the reduction rate \mathbf{r}^y for the specific product (or group of products) which is multiplied with the penetration rate \mathbf{p}^y , modelling that a desired policy is not able to reach the total set of countries or population (Wood et al., 2018b). As not the whole FD vector \mathbf{y} is considered an elementwise multiplication is done symbolized by \times . Both \mathbf{r}^y and \mathbf{p}^y range between 0 and 1, translating into the final penetration rate of an intervention, also ranging between 0 and 1. A potential policy could

be to reduce electricity by coal by 50% but given that due to political protests a number of coal plants cannot be closed, the actual penetration rate might only be 80%, thus decreasing the desired policy effect and making the scenario more realistic. In order to guarantee electricity security the reduction in electricity generation needs to be compensated by substituting the reduced output by other sources, the amount to be substituted is calculated by subtracting the reduced FD vector from the baseline vector ($\mathbf{y} - \mathbf{y}^{red}$), in order to redistribute to other sources the difference is divided by a distribution key s , generally also set by the desired policy, and added to the baseline vector to derive the \mathbf{y}^{int} policy intervention FD vector in (13).

$$\mathbf{y}^{red} = \mathbf{y} \times (1 - \mathbf{r}^y \times \mathbf{p}^y) \quad (12)$$

$$\mathbf{y}^{int} = \mathbf{y} + (\mathbf{y} - \mathbf{y}^{red})/s \quad (13)$$

By pre-multiplying with the environmental stressor vector, as shown in equation (8) the policy intervention FP is calculated in equation (14). Following Wood et al. (2018b) the electricity FP for a whole year is taken into account, thus the intervention represents the annual impact of a policy. In a final step the baseline FP, calculated based on the original electricity FD \mathbf{y} , and the intervention FP are compared and the percentage difference can be taken via equation (15).

$$\mathbf{FP}^{int} = \mathbf{e} * \mathbf{L} * \mathbf{Y}^{int} \quad (14)$$

$$\Delta \mathbf{FP} = \frac{\mathbf{FP}^{int} - \mathbf{FP}^{ref}}{\mathbf{FP}^{ref}} \quad (15)$$

The proposed analysis thus allows to compare the absolute as well as the relative gains from switching the electricity mix, based on the technology constraint of the base year. The two material FPs will be aggregated into a single material FP for improved readability. These methods will thus allow to hypothesis three by offering a multitude of potential policy interventions.

4.5. Limitations

The three methods introduced are subject to several shortcomings. As introduced in section 2.1.2. FPs have been criticized for preferring CBA to PBA and thus ignoring technological differences or improved emission intensities in the exporting sector (Kander et al., 2015). In order to account for the sectoral efficiency of GHG and materials, the respective intensities will be further considered based on equation (9). Nevertheless, it should be stated here that only the consumer driven environmental consequences are analysed, thus moving away from a territorial limited environmental impact study. While this implies that environmental pressures are taking place abroad, the geographical scope of the PLD method is missing, as it indicates the environmental pressure per layer, but is not disaggregated into environmental degradation within Europe or outside. This limits the outcome of the analysis to simply understanding the complexity of the GSCs for the GHG emissions and material extraction for the renewable and non-renewable sources of electricity, ignoring in which countries or regions the respective layers are situated. Yet, the analysis will allow to include a discussion about the development

of these trends over time. Finally, the method for the policy analysis conducted here is on the simpler side of scenario-based modelling, due to the ignorance of consumer elasticities, technological adjustment and changes to the economic structure. Thus, it renders the modelling to a static yearly comparison, ignoring that concomitant technological advances to changes in FD could result in broader life-style changes to more sustainable consumption (Wilting et al, 2008). Finally, the parameter setting for the policy modelling are set rather arbitrary and follow indications made from the literature but do not represent official electricity policies, the results are therefore to be considered as suggestive. Keeping the limitation in mind the following section will present the results from the empirical analysis.

5. Empirical Analysis

In a first step the results will be described in detail to subsequently discuss their relevance, the coherence with prior research and the potential importance for policy making. All results presented are based on the author's own calculation using the data and methods introduced in section 3 and 4, resulting in three types of FPs: Greenhouse gases, metal ores and minerals, for overall, renewable and non-renewable, or specific sources of electricity finally consumed in the EU.⁸

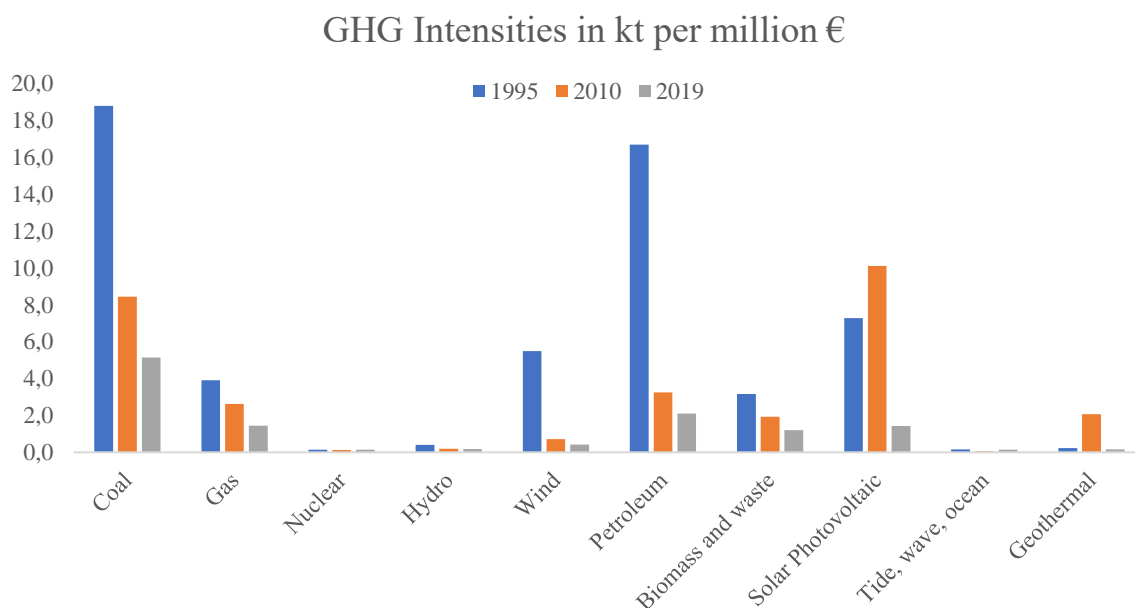
5.1. Footprints 1995-2019

Starting with an overview of the three FPs for the total electricity consumption of the EU between 1995 and 2019 indicates that from an initial absolute emission equivalent of 220,798 kt in 1995 the level of emissions dropped to 157,139 kt in 2019, following a decreasing trend with a short uprise after the recovery from the financial crisis in 2008. Contrary, the metal ores FP has seen an increasing trend from 2,749 kt to 3,601 kt of material extracted to fulfil the EU electricity demand, with the mineral FP following the GHG FPs trend from 23,986 kt to a reduced 14,548 kt in 2019.⁹ The share of the renewable and non-renewable sources FPs of the total electricity FP can be computed and compared overtime. Between 1995 and 2019, the non-renewable electricity sources consistently make up more than 95% of the total electricity GHG FP and around 85% of the two material FPs. Especially, in the case of GHG, the three sources of coal, gas and petroleum are responsible for the majority of emissions related to power generation, while in the case of metal ores and minerals, the four major contributors are coal, gas, nuclear and hydro. While the overall FPs allow to see a clear improvement in two of the three absolute FPs over the 25 years analysed, *Figures 2 to 4* given an overview of the different FP intensities, in unit per million € demanded, per source of electricity for three representative years of the total sample 1995, 2010 and 2019. Due to the limited use of solar thermal, the results were omitted from the

⁸ The computation of the result was conducted using Matlab in order to process the large amount of data in Exiobase 3. For further information on the do-file please contact the author. After the calculation the results were exported to Excel for additional analysis and graphical illustrations.

⁹ For a more in depth overview of the FPs between 1995-2019 please refer to *Table A1* in the appendix.

illustrations.¹⁰ *Figure 2* shows that in the year 1995 the electricity produced by coal, petroleum and other oil derivatives as well as by solar photovoltaic had the highest GHG intensity. Meaning for each unit, in million €, of electricity consumed they emit the most GHG directly and indirectly along the complete supply chains. Interestingly, if the same comparison is conducted for the year 2010 it shows that the production of electricity based on solar photovoltaic became the most emission intensive way of production, with more than 10 kt of GHG emitted per million €, whereas both coal and petroleum have seen massive improvements of their emission intensity. The two findings could be indicative that larger emissions regulations and improved carbon capture technologies made the electricity production of both coal and petroleum more efficient. The technology of solar photovoltaic saw a similar massive improvement of emission intensity in the second period analysed between 2010-2019, where the emission intensity dropped to below 2 kt per million € demanded. This strong reduction might be the result of improved technology requiring less emissions along the complete supply chain, or of an improvement of the efficiency of power generation. *Figure 2*, on the contrary, clearly shows that electricity by nuclear, hydro, and tide, wave and ocean are the cleanest sources of electricity based on the GHG FP intensities, with wind as a source rapidly catching up by improving the intensities from 5 kt to below 1 kt per million € over the 25 years.

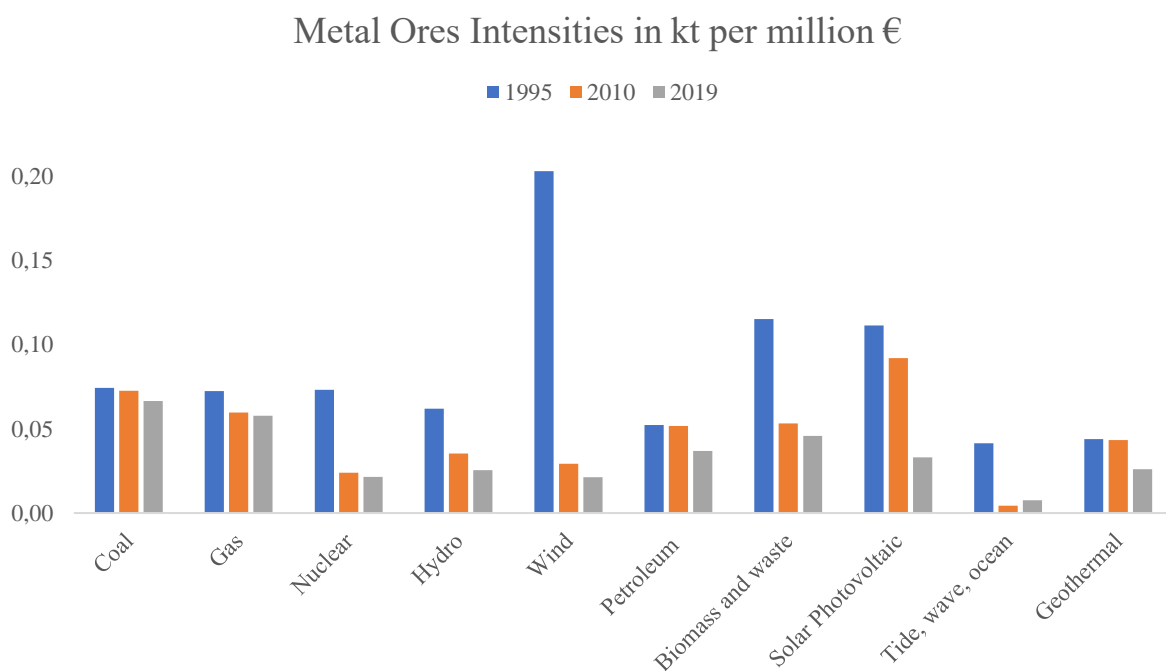


Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)

Figure 2 - Greenhouse Gas FP Intensities

¹⁰ For the detailed absolute FPs, final demands, intensities and growth rates please refer to *Tables A2-A4* in the appendix.

If the metal ores intensities are looked at in *Figure 3* a different picture arises. In 1995 the most metal ore intensive sources of electricity production were wind, biomass as well as solar photovoltaic, with more than 0.10 kt per million €. Yet, especially in the case of wind there is a rapid drop in the intensity when the year 2010 and 2019 are taken into account, with a similar drop more than halving the metal ore intensity from solar photovoltaic between 2010 and 2019. This is in stark contrast with the relatively stable intensities of coal, gas and petroleum sources, indicating that whereas the former sources depend on initial investment and infrastructure, the latter continuously have a high level of metal extraction over the years. Contrary to the GHG intensity results, nuclear and hydro are no clear outliers anymore, with similar intensities in 1995 and improved levels in 2019.



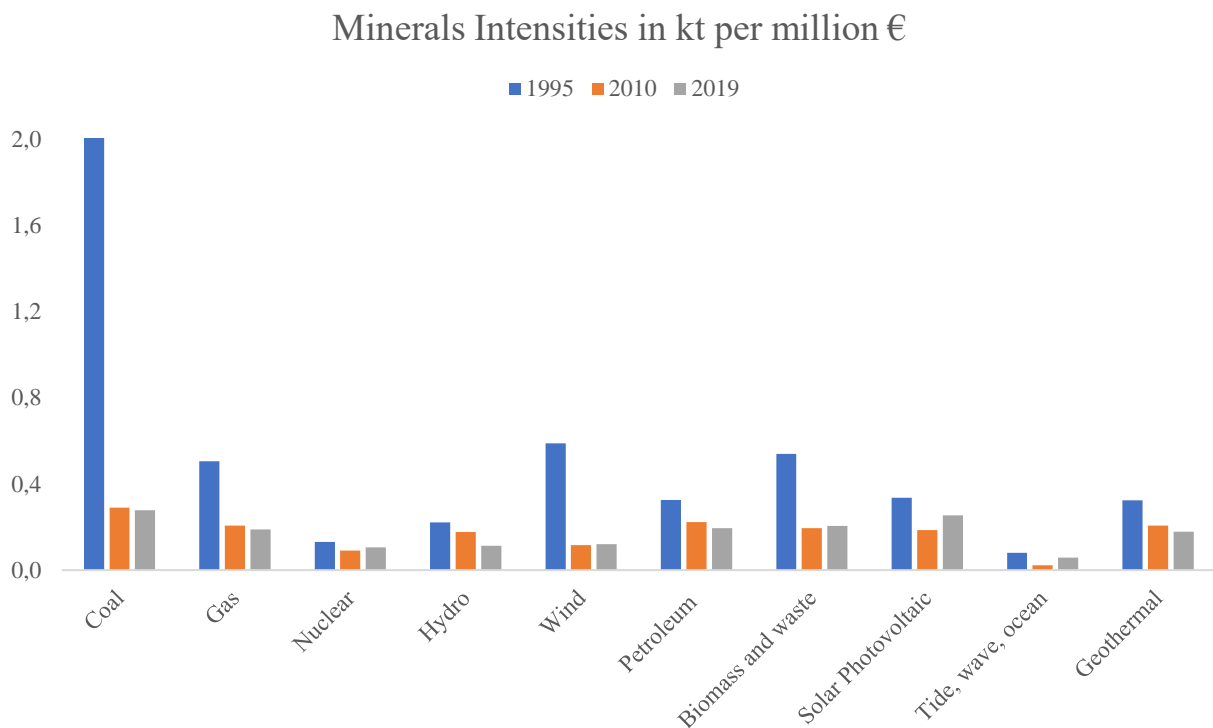
Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)

Figure 3 - Metal Ores FP Intensities

In the case of the mineral intensity in *Figure 4* the picture gets more homogenous with the only anomaly existing for the year 1995, which saw high mineral intensities for the production of electricity by coal petroleum, wind, as well as biomass and waste. Apart from the outliers, the impression is that overall the mineral intensities per million € are in a similar range for renewable and non-renewable sources alike. Interestingly, compared to the development of metal ores intensities over time, solar photovoltaic as well as biomass have seen an increase in intensities between 2010 and 2019, indicating at a change in the underlying production technologies requiring different forms of minerals.

The FP intensities were based on three aggregated environmental accounts to facilitate the analysis and interpretation. However, the GHG, metal ores and minerals can easily be broken down into their sub-categories, which has been done for the years 1995, 2010, and 2019 to indicate the growth rates of the

GHG and materials in renewable and non-renewable sources of electricity in *Figure 5* and *A8*. *Figure 5* shows the 23 environmental stressors exerted by the FD for electricity from renewable sources. The blue and orange bars indicate the growth rates between 1995-2010 and 2010-2019 respectively. As can be seen the only stressors with a positive FP growth in the second time period are iron, lead, nickel, uranium and thorium ores, as well as clays and kaolin, and limestone, gypsum chalk and dolomite, all other indicators have seen a negative growth in between 2010-2019. Once again, the absolute values are compared, and thus from an absolute FP perspective, the increasing trends in the overall metal ores FP seem to have been especially driven by the increased demand for uranium and thorium ores. If the same analysis is applied to the 23 stressors in the case of non-renewable sources of electricity a somewhat similar trend is visible, see *Figure A8* in the appendix, with significantly higher growth rates of uranium and thorium ores, as well as lead, nickel and copper ores. Simultaneously, it is encouraging to see that in the case of GHG the growth rates are negative between 2010-2019 for both renewable and non-renewable sources, and that overall, the positive growth rates in the period 1995-2010 have been reversed into negative ones in most of the 23 categories studied. The specific environmental stressors further show that a disaggregation is helpful to disentangle the overall environmental impacts to address specific technological improvements or a substitution of materials used in production along the GSC. In the next section the results for the production layer decomposition are presented, allowing to draw insights into the complexity of GSCs.



Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)

Figure 4 - Minerals FP Intensities

Environmental FP growth rates for electricity from renewable sources

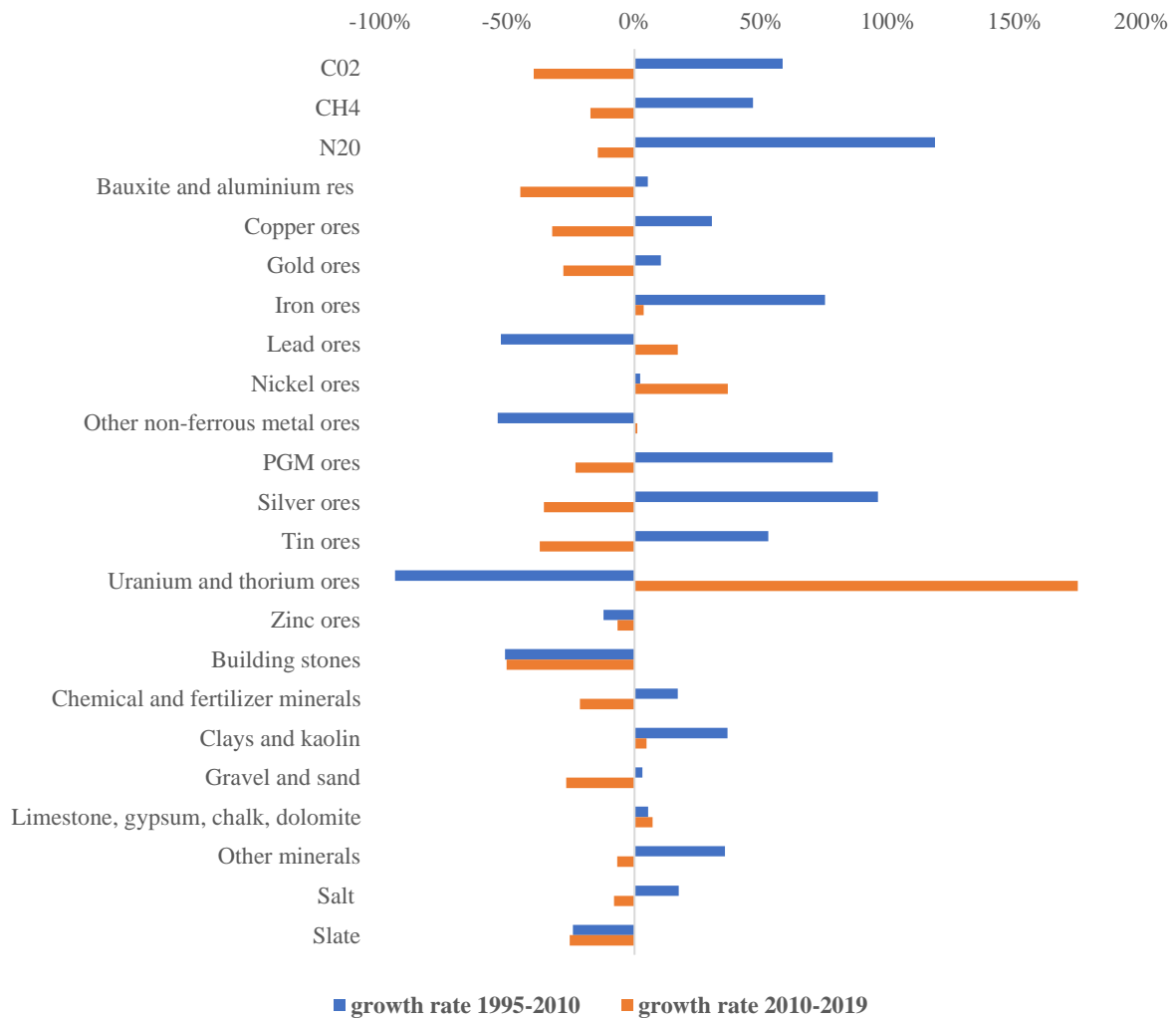


Figure 5 - Renewable Electricity - Environmental FPs Growth Rates

5.2. Production Layer Decomposition

The insights from the previous section require to further disaggregate the supply chains of the electricity sources. The applied PLD analysis is summarised in *Table 1* and *Figure 6* and allows to gain insights into the distribution of the environmental impacts for renewable and non-renewable sources of electricity. For both categories and for the years 1995, 2010, and 2019 the share of each environmental stressor per layer, as well as the total FPs are indicated. As introduced in section 4, Layer 0 shows the direct consequences of electricity generation, emissions and material extracted while electricity is produced. Therefore, unsurprisingly it is found that in case of non-renewable sources, most of the GHG emissions are emitted during the generation of electricity, with the higher layer rapidly losing in importance. Layer 0-2 combined make up roughly 99% of all the total emissions FP for non-renewables.

Contrary, renewable sources emit between 50-60% in layer 0 with layer 1 and 2 around 20% and 10% each over the 25 years analysed. Contrary to non-renewables roughly 10% of all emissions take place in activities further upstream in layer 3 and higher. This is a clear indication for longer and more complex supply chains, which are more dependent on a multitude of emission heavy activities. If the time trends are considered, there is a clear shift of emissions towards higher layers in the case of non-renewables, clearly visible in *Figure 6* with the blue bar of layer 0 decreasing and the other layers increasing from 1995, to 2010 and to 2019.

Metal ores and minerals both follow a similar trend with layer 0 equalling 0%, as no additional resources are extracted for the actual electricity generation for both renewable and non-renewables sources. But there are clear differences when it comes to the distribution along the layers. Metal ores are more evenly distributed. Layers higher than 5 still contain 15%, for renewables, and 14%, for non-renewables, of metal ores in 2019. Similar to metal ores, the main burden of mineral extraction takes place in layer 2 and 3 for renewables and non-renewables alike. In the case of renewable electricity sources a clear redistribution from downstream to upstream mineral extraction pressures is visible induced by a large and radical change from 1995 to 2010, and the distribution among layers held constant for the period 2010 to 2019.

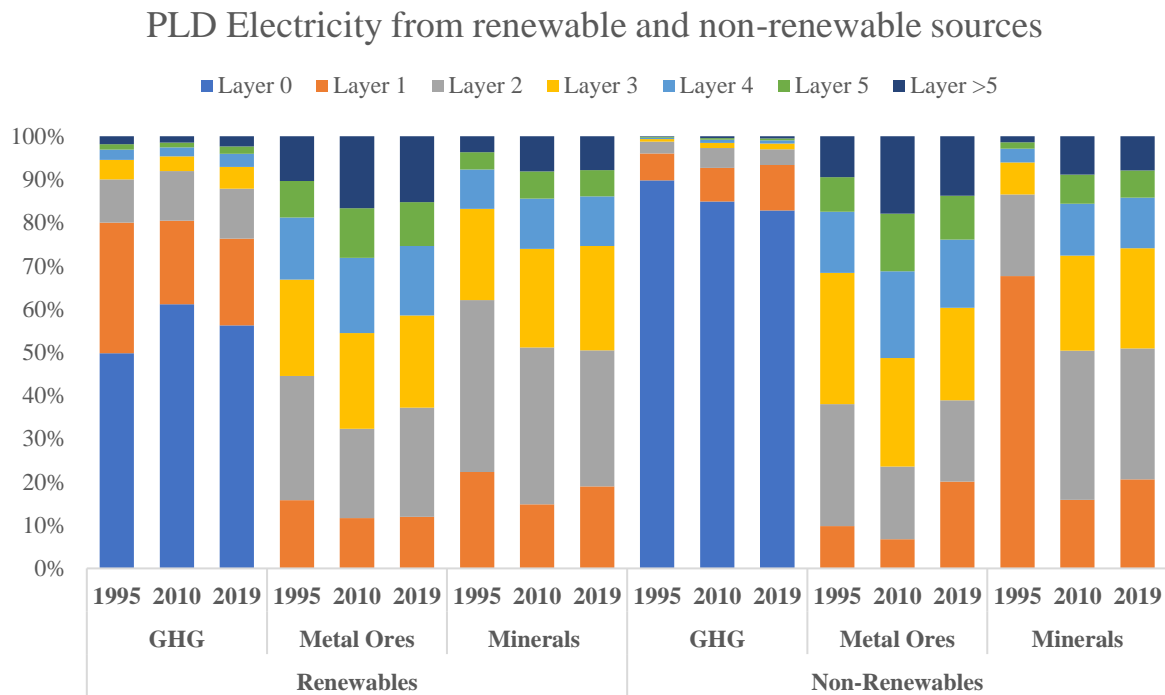


Figure 6 - PLD renewable and non-renewable sources of electricity

Table 1 - Production Layer Decomposition 1995 - 2010 - 2019

Production Layer Decomposition								
	Layer 0	Layer 1	Layer 2	Layer 3	Layer 4	Layer 5	Layer >5	FP in kt
Electricity from renewable sources								
GHG								
1995	50%	30%	10%	5%	2%	1%	2%	4767.1
2010	61%	19%	11%	3%	2%	1%	1%	7550.2
2019	56%	20%	12%	5%	3%	2%	2%	4552.3
Metal Ores								
1995	0%	16%	29%	22%	14%	8%	10%	381.0
2010	0%	12%	21%	22%	17%	12%	17%	397.6
2019	0%	12%	25%	21%	16%	10%	15%	328.6
Minerals								
1995	0%	22%	40%	21%	9%	4%	4%	1423.5
2010	0%	15%	36%	23%	12%	6%	8%	1748.9
2019	0%	19%	31%	24%	11%	6%	8%	1537.7
Electricity from non-renewable sources								
GHG								
1995	90%	6%	3%	1%	0%	0%	0%	216030.6
2010	85%	8%	5%	1%	1%	0%	0%	207438.4
2019	83%	11%	4%	1%	1%	0%	1%	152586.4
Metal Ores								
1995	0%	10%	28%	30%	14%	8%	9%	2368.3
2010	0%	7%	17%	25%	20%	13%	18%	3174.1
2019	0%	20%	19%	21%	16%	10%	14%	3272.8
Minerals								
1995	0%	68%	19%	7%	3%	1%	1%	22562.6
2010	0%	16%	35%	22%	12%	7%	9%	11981.8
2019	0%	21%	30%	23%	12%	6%	8%	13010.2
Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)								

Based on the PLD, similar to the FP analysis in section 5.1., the largest difference between renewables and non-renewables lies in the GHG emissions. Whereas metals and minerals show a more homogenous picture, this becomes more evident if the left and right side of *Figure 6* is considered. Renewable electricity sources see a clear redistribution of GHG towards upstream activities, whereas non-renewables exert most pressure in the downstream activities. Minerals and metal ores are needed in all production steps with the exception of the final production of electricity with very similar trends between renewables and non-renewables.

The PLD analysis shows the difficulties of simply comparing renewable and non-renewables sources of electricity, making a prediction based on FP intensities and PLD relatively difficult. Thus, in the last

section of the empirical analysis the scenario based what-if modelling of an electricity transition towards more renewable and less non-renewable sources is presented.

5.3. Policy Analysis

Given the fact that the largest increase between 2009-2019 in renewable sources used for electricity are coming from wind, solar PV, as well as biomass and waste (European Commission, 2020), the policy analysis will focus on a re-distribution away from non-renewables towards the former three sources. Hydro is further included to account for the large power generating capacities of this technology. The method introduced in 4.4. is applied for three distinct scenarios. The first two are focusing on proposed policies trying to close down coal power generation stations in the EU, and more specifically in the UK (Allan et al., 2007; Foxon, 2013) and the second scenario is based on a generalised policy to the EU following Germany's goal to completely stop nuclear electricity production (Quitow et al., 2016). The results from the analysis are summarised in *Table 2* with scenario 1 applying a policy reduction rate of 50%, estimated to penetrate with 75% effectiveness in the EU, with the lost electricity being substituted by 50% wind and 50% solar photovoltaic. Scenario 2 is based on scenario 1 but redistributed the electricity to hydro and biomass as well, with an equal share of 25% each. Finally, scenario 3 hopes to reduce nuclear electricity by 90%, applying the same penetration rate of 75% and the same redistribution key as in scenario 2.

Table 2 - Policy Analysis Scenario Assessment

Policy Analysis			
Electricity Policies	Scenario 1	Scenario 2	Scenario 3
	Coal	Coal	Nuclear
Possible reduction rate 1	0.5	0.5	0.9
Penetration rate t	0.75	0.75	0.75
Shifting Key / Distribution	0.5 Wind, 0.5 Solar	0.25 Wind, 0.25 Solar, 0.25 Hydro, 0.25 Biomass/Waste	0.25 Wind, 0.25 Solar, 0.25 Hydro, 0.25 Biomass/Waste
GHG FP after intervention	164404	143791	155036
GHG change	7266	-13348	-2103
GHG change percentage	4.6%	-8.5%	-1.3%
Material FP after intervention	16883	17396	16261
Material change	-1266	-754	-1889
Material change percentage	-7.0%	-4.2%	-10.4%
Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)			

Scenario 1 is remarkable and interesting given that the GHG FP based on the policy intervention would result in an increase of 4.6% compared to the baseline scenario, or an increase of a yearly estimate of 7,266 kt more of GHG emission. Yet, based on this scenario the material FP could be reduced by 1,266

kt or by -7%. While any shift to renewable electricity has generally been seen as more sustainable and less environmentally harming the contrary could result due to the massive scale effect, thus investment effect, needed to generate the large quantity of electricity to compensate the reduction in coal electricity. Therefore, large scale production of wind and solar components are necessary and as has been seen in the PLD analysis, the production of renewable is more emission heavy in upstream production activities. Scenario 2, using a different redistribution key would reverse the findings from scenario 1, as the GHG FP would be reduced by -8.5%, with the material FP decreasing by -4.2% as well. The two what-if scenarios thus result in two different outcomes by simply changing by which sources the electricity lost should be compensated, and given the low emissions intensities of hydro, the overall impacts of the policy can drastically be influenced. Taking a look at scenario 3 shows that reducing nuclear electricity production and transitioning to the four renewable sources would reduce the GHG FP by only -1.3%, whereas the material FP could be reduced considerably by -10.4%, showing that there is potential to further decrease the environmental pressure from a relatively low emission intensive source of electricity.

In the last part of the empirical analysis the results found will be put into the perspective of prior research, discuss how this affects our hypothesis and specifically discusses the policy implications.

5.4. Discussion and Result Implications

If the intensities of renewable and non-renewables are aggregated from the categorical breakdown in section 5.1., then the first hypothesis, expecting lower GHG FP intensities and higher material intensities of renewable sources of electricity, is supported in the case of GHG intensities. The conclusion is based on the whole period showing a massive improvement of all GHG intensities, most impressively for the period 2010-2019 for renewable electricity sources. In the case of the material FP intensity the first hypothesis cannot be supported, as surprisingly with a few exceptions in the year 1995, the material per unit of electricity demanded was relatively equal among all categories. Similar to hypothesis one, the second hypothesis, expecting a higher dependence on upstream environmental pressures for renewable electricity sources, can be confirmed in the case of GHG emissions. As there is an increasing shift of emissions towards higher layers, thus more upstream activities, over time in the case of renewable sources. While non-renewables sources see a similar pattern, it is less distinctive due to 85% of emission taking place in layer 0. Again, metal ores and mineral ores show a more homogeneous picture, as seen with the FP intensities. Hence, no clear differentiation between renewables and non-renewables can be drawn, and further, there are no prominent patterns over the 25 years analysed. Finally, hypothesis three, expecting a reduced environmental impact after an improved renewably sourced electricity consumption composition, is weakly supported. Whereas in one model specification the GHG FP increased and the material FP decreased, the other scenarios clearly showed a decrease in environmental pressure exerted by an improved FD of the EU electricity mix.

Subsequently, the question of how the conclusions drawn from the hypotheses fit into prior research findings arises.

In contrast to Hertwich et al. (2015), the focus on metal ores and minerals specifically does not confirm that renewable electricity sources in general have higher material intensities, possibly due to the truncation error in the LCA, which does not consider all layers compared to this work. While Frantzeskaki and Gekas (2005) found solar technologies to be more material intensive than alternative sources, this work has shown by extending the time-horizon and using a more inclusive approach that there were clear improvements in the material intensity not just for solar photovoltaic but almost all renewable sources of electricity. Based on the PLD prior results could further be confirmed, showing that the operating phase of power generation stations are accounting for most of the GHG emissions (Proops et al., 2008). Additionally, this analysis was able to show that there are increasing trends towards more emissions in more upward activities, reducing the importance of the operating phase in generating environmental pressures. Allan et al. (2007) hypothesized that due to the high level of heterogeneity in the environmental impact of different sources of electricity an aggregation can result in large differences in their environmental impacts. This has been confirmed with the policy analysis in section 5.3., as depending on which sources will compensate for a reduced non-renewable electricity source, the outcome differs greatly, confirming prior studies considering the impact of changes in the electricity mix (Jafar et al., 2008). The surprising result of an increasing GHG FP when coal is replaced by solar and wind electricity follows the logic of Neij & Astrand (2006) finding a higher up-front investment for green technologies. Given the structure of EE-MRIOT and the yearly scope, the analysis does not allow to dynamically disaggregate the policy intervention into investment and production effects. Nevertheless, given that the production layer decomposition for renewables sources showed that upstream GHG emissions are increasingly important, the increase in GHG emissions can potentially be explained. This is due to the fact that layers higher than 2 de-facto represent the manufacturing of wind and solar components and are thus representative of an increased necessary up-front investment. The what-if scenario thus compares the two scenarios based on the same technology in a given year but does not predict how the environmental impacts dynamically evolve. Similar to Wiebe et al. (2019) material extraction rates can be decreased massively if a transition towards alternative and more renewable electricity mixes are achieved, with the three applied scenarios resulting in a 4% to 10% reduction of the material FP. Conclusively, the applied empirical analysis fits prior insights from the literature and allows to draw new and improved insights into electricity and mitigation policies.

While the discussion on greenhouse gases might be the most prevalent in the public discussion, the empirical results show the need to include the material costs of renewable electricity generation in policy making as well. The PLD analysis shows that the whole chain needs to be analysed as metal ores and minerals would not be included if only the direct consequences of electricity production were analysed. Interestingly it shows that on average a percentage higher than 10% is in the layers above 5,

signalling the complexity and length of electricity supply chains. This finding underlines the importance of finding new and improved indicators for resource efficiency as well as for environmental impacts (Wiedmann et al., 2015). Yet, if the FP intensities are looked at given the relatively more stable variation in the case of metal and mineral ores there is a less significant potential gain through a transition, whereas in the case of GHG intensities a larger potential can still be fulfilled. Thus, the suggestion that by focusing on the GHG efficiency policies can lead to relative quick wins (Liu et al., 2019) by improving the GHG intensities is further confirmed in the case of electricity generation. Nevertheless, it is important to acknowledge that even though material intensities have been improving, the total stock of resources remains finite and thus a “true dematerialization has to mean an absolute decoupling of impacts” (Wiedmann et al., 2015, p. 6275). Such a decoupling in terms of materials can be achieved by focusing on extending the lifetime of the materials (Haas et al., 2015). In the comparison between renewable and non-renewable sources the assumption can be drawn that while the material intensities are on similar levels, the use of non-renewable sources will automatically result in a depletion of resource stocks, while renewables require higher up-front material extraction for the creation of the power plants, these resources can be re-integrated and recycled after use.

Overall, the conducted analysis clearly shows the potential gains from a more renewable electricity mix. So, it becomes evident that in order to achieve future climate neutral electricity generation a clear paradigm shift is necessary to boost clean electricity by heavily investing in technology and innovation to gain a competitive edge in the production of green electricity (European Commission, 2019 & 2020). A new development path-way focusing on the lock-in of renewable sources of electricity generation is therefore of utmost importance (Jiborn, Kulionis & Kander, 2020). However, the use of CBA results in the fact that the implementation and the design of national strategies for reducing the FPs are more difficult to address if foreign upstream GHG and material flows are to be considered (Barrett and Scott, 2012). In terms of policy making, the consumption-based view should be considered as a complement to the direct emission and absolute materials extracted within a territory, as implementing national mitigation strategies are more easily enforced (Giljum et al., 2015). Nevertheless, from an EU perspective the insights from the policy intervention analysis here could have interesting consequences for policy making, as if countries with higher renewable electricity sources availability would focus on producing energy intensive goods, this could lead to an overall improvement of both the global GHG and global material FPs.

6. Conclusion

The conducted study aimed to derive an improved understanding of the environmental consequences from a transition towards a more sustainable electricity generation mix. A broad strain of literature has prior analysed the barriers to energy transitions, the potential economic benefits but also environmental threats of continuing the unsustainable path of energy consumption. Whereas the terms of electricity and energy cannot be used interchangeably, understanding the different sources of electricity generation in more depth is assumed to provide a solid empirical foundation on how to tackle the heating and transport sector as well. So far, the literature has neglected the analysis of environmental impacts caused by the generation of electricity, and the few studies conducted mainly focused on the energy efficiency as well as carbon emissions. Hence, the objective of this study was to extend the environmental pressures exerted from the consumption of electricity, by taking different greenhouse gases as well as metal ores and mineral extractions into account. Further, the study made use of newly available data by applying improved input-output methods in order to account for emissions and material extractions outside of national or regional territories. Using the latest update of *Exiobase 3* (Stadler et al., 2021), three different input-output methods were applied. The first method calculated GHG and material FPs for the European Union's overall FD of electricity for the time period of 1995-2019. The FPs were further disaggregated into the 12 underlying sources of electricity, as well as the 23 environmental stressors, to calculate the different intensities per unit of million € consumed. In a next step a production layer decomposition for renewable and non-renewable sources of electricity was applied to trace the environmental impacts along the whole supply chain. Thus, the importance and pressure of each layer could be categorized from upstream to downstream. The final part of the empirical analysis conducted a policy intervention scenario comparison, in which the FP of the current EU electricity mix was compared with a hypothetical scenario in which non-renewable sources were reduced and substituted by a range of renewable alternatives.

The empirical analysis has shown, that when the GHG, metal ores and minerals intensities of different sources are compared, GHG emissions were changing drastically over time as well as between the source categories. While initial renewable electricity, mainly from wind and solar photovoltaic, was relatively GHG intensive the period 2010-2019 has led to large decreases in the emission intensities due to improved technological advances. At the same time, metal ores and mineral intensities are characterized by less significant differences among the sources, resulting in a lower reduction potential from switching towards renewable sources of electricity. However, if the absolute FPs are taken into account the growth rates for the 2010-2019 period for almost all environmental stressors except uranium, iron ores, lead ores and nickel ores, have seen a decline. This generally represents an improvement in technological efficiency of the electricity generation sector. The applied production layer decomposition resulted in similar insights, clearly showing that there are large differences in the distribution along the complete supply chain of GHG between renewable and non-renewable sources.

Again, metal ores and minerals showed similar patterns and a more evenly distributed extraction rate over all the layers, clearly distinguishing them from the GHG supply chain. Finally, the applied policy intervention showed that transition towards renewables per se must not be environmentally beneficial, as in one of the scenarios reducing coal and compensating it with increased solar photovoltaic and wind electricity would result in an increased GHG FP. Yet, if the same reduction was distributed more evenly among renewable sources, in a second scenario, both the GHG and material FP would be reduced drastically.

In showing that the improved data availability and input-output methods can be used to derive empirical results, this thesis further contributes to the debate of electricity and energy transitions. The proposed policy intervention model can be used for quick-scan assessments by modelling what-if scenarios that go beyond simple analyses of electricity consumption. In order to improve future academic work, a final consideration is to improve the data availability, harmonization and acceptance of input-output tables by official institutions in order to enhance the quality and constancy of environmental assessment analyses. Initiatives such as the OECD's Trade in Value Added database, and projects such as the EU's DESIRE will be key to continue reaching policy makers outside of academia (Yamano & Webb, 2018) and will contribute to the statistical validity (Tukker et al., 2018).

The clear distinction between renewables and non-renewables in the PLD shows that transitioning towards a more sustainable and renewably sourced electricity mix will at least in the case of GHG emissions outsource large parts of the emissions to more upstream activities in the supply chain. This has far reaching policy implications, the main one being that simply switching without considering all environmental consequences within and outside a territory will result in biased policy targets that neglect the outsourcing of emissions and material extractions. The aggregation into renewable and non-renewables further contributes to an improved understanding of the consequences of a change in the electricity consumption mix, whereas more concise policies will need to further disaggregate the electricity groups. Due to the high resolution of the database, the policy intervention analysis can be further improved in future studies taking market responses based on changes in the electricity mix into account. A further disaggregation of the electricity generation products into imports and exports and the corresponding technological adjusted GHG and materials accounting, following Kander et al. (2015), could provide additional insights into the global consequences of a change in the electricity mix towards renewable sources. Whereas the analysis presented here descriptively showed the environmental impact of the final electricity demand from EU it indirectly assigned the responsibility to the consumers. Whether the global responsibility of material extraction and GHG emissions ultimately lies with the consumer or the producer will remain a heated debate and should be further studied in an intersectoral approach. Potential insights into the global distribution of responsibilities could be taken by extending the production layer decomposition analysis to include the geographical scope. This would further allow to take the material import dependency or energy security into account

(Giljum et al., 2016). Additionally, in policy discussions both the flow and stock of resources need to be considered, therefore extending future research to take a circular economy stance. This would allow to infer how the recyclability of renewable sources of electricity could reduce the depletion of earth's stock of material (Tsoutos et al., 2005). Ultimately, achieving a transition towards renewable electricity should be a just transition, thus future analysis should take socio-economic factors such as employment into account to guarantee the "Green Energy for all Europeans" initiative (European Commission, 2020).

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Appendix

A 1 - FP Overview 1995-2019

EU28 FP Overview 1995-2019									
Year	GHG FPs in kt			Metal Ores FPs in kt			Minerals FPs in kt		
	Electricity EU28	Total EU28	Share	Electricity EU28	Total EU28	Share	Electricity EU28	Total EU28	Share
1995	220798	3408969	6,5%	2749	530277	0,5%	23986	3619766	0,7%
1996	217263	3773323	5,8%	2852	726239	0,4%	15179	3713833	0,4%
1997	229357	3581574	6,4%	3273	686143	0,5%	33178	3768072	0,9%
1998	244998	3736830	6,6%	4437	889777	0,5%	11462	3915846	0,3%
1999	213252	3740284	5,7%	3300	847652	0,4%	29750	4321725	0,7%
2000	210137	3811630	5,5%	3237	875070	0,4%	13021	4292847	0,3%
2001	176627	3801326	4,6%	3517	908790	0,4%	12172	4369516	0,3%
2002	211869	3794927	5,6%	3205	862459	0,4%	11866	4344737	0,3%
2003	248719	3999187	6,2%	3794	891564	0,4%	26129	4508764	0,6%
2004	202175	4021218	5,0%	3013	827390	0,4%	11723	4642000	0,3%
2005	204886	4066149	5,0%	2860	804232	0,4%	17001	4939836	0,3%
2006	202500	4176998	4,8%	3567	866880	0,4%	13114	5107498	0,3%
2007	184439	4239867	4,4%	3543	900185	0,4%	14700	5444657	0,3%
2008	160758	4130930	3,9%	2866	851219	0,3%	13452	5046548	0,3%
2009	195767	3677157	5,3%	3507	709671	0,5%	15651	4417470	0,4%
2010	214989	3754391	5,7%	3572	736077	0,5%	13731	4174437	0,3%
2011	228569	3772786	6,1%	4119	709285	0,6%	17476	4464504	0,4%
2012	246839	3538903	7,0%	3738	641024	0,6%	16828	3959452	0,4%
2013	185588	3436029	5,4%	3492	593140	0,6%	16878	4148536	0,4%
2014	198709	3380627	5,9%	3100	584541	0,5%	14936	4015810	0,4%
2015	183450	3364528	5,5%	2979	525405	0,6%	13558	3689747	0,4%
2016	157870	3280093	4,8%	2596	528952	0,5%	12110	3421865	0,4%
2017	124903	3251880	3,8%	2934	592303	0,5%	12444	3610749	0,3%
2018	168800	3366532	5,0%	2792	605681	0,5%	12978	3975828	0,3%
2019	157139	3210995	4,9%	3601	620699	0,6%	14548	3983583	0,4%

Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)

A 2 - GHG FP for each electricity source

EU28 GHG FP per Electricity Group						
		Coal	Gas	Nuclear	Hydro	Wind
GHG FPs in kt	FP 1995	151888,6	25343,7	2205,0	1723,9	1040,1
	FP 2010	139619,6	53818,9	2690,9	1139,4	1684,0
	FP 2019	111508,5	30576,0	3115,1	1397,5	917,6
Final EU28 Demand for Electricity in million €	FD 1995	8082,0	6492,0	15904,8	4321,0	189,2
	FD 2010	16552,8	20565,9	23050,8	5700,4	2367,1
	FD 2019	21663,1	21113,1	21996,8	7665,7	2194,0
Intensity GHG kt per million €	I 1995	18,8	3,9	0,1	0,4	5,5
	I 2010	8,4	2,6	0,1	0,2	0,7
	I 2019	5,1	1,4	0,1	0,2	0,4
Growth Rate FP	1990-2010	-8%	112%	22%	-34%	62%
	2010-2019	-20%	-43%	16%	23%	-46%
		Petroleum and other oil derivatives	Biomass and waste	Solar Photovoltaic	Tide, wave, ocean	Geothermal
GHG FPs in kt	FP 1995	34861,0	1987,2	1,9	2,9	11,1
	FP 2010	9613,9	4419,4	113,1	1,0	193,2
	FP 2019	5781,9	2161,6	51,1	3,4	21,1
Final EU28 Demand for Electricity in million €	FD 1995	2089,2	627,2	0,3	18,1	48,0
	FD 2010	2962,0	2281,6	11,2	17,5	93,6
	FD 2019	2751,1	1798,7	35,7	23,5	127,4
Intensity GHG kt per million €	I 1995	16,7	3,2	7,3	0,2	0,2
	I 2010	3,2	1,9	10,1	0,1	2,1
	I 2019	2,1	1,2	1,4	0,1	0,2
Growth Rate FP	1990-2010	-72%	122%	5726%	-65%	1648%
	2010-2019	-40%	-51%	-55%	229%	-89%
Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)						

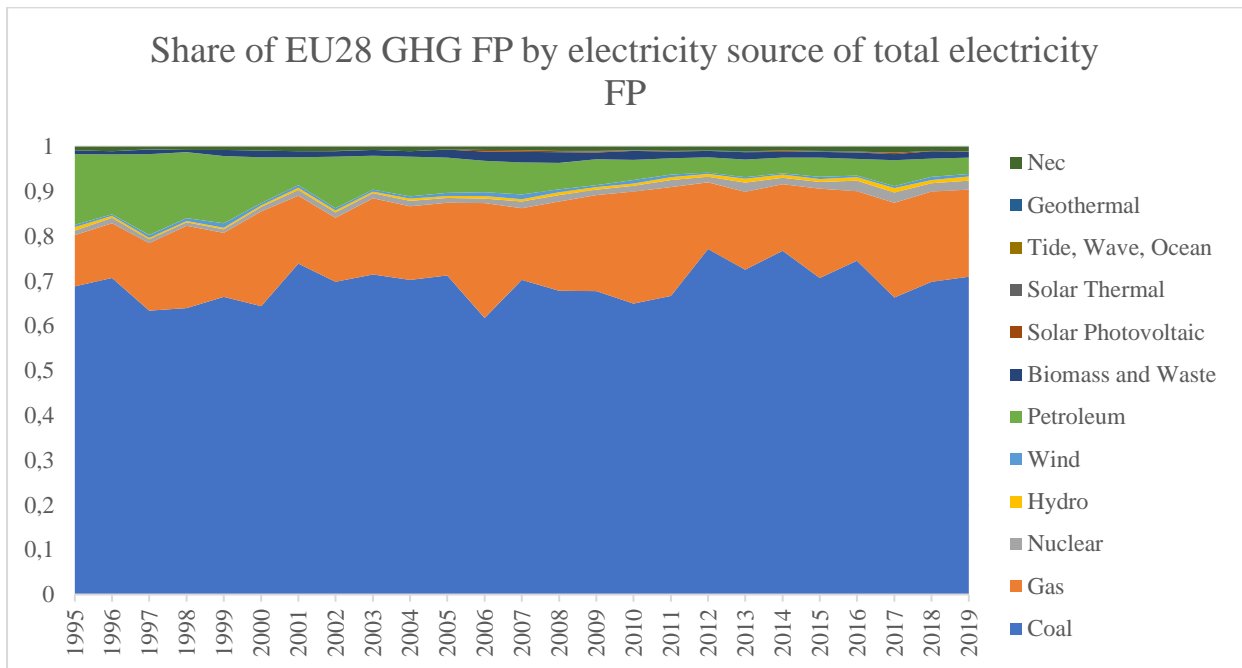
A 3 - Metal Ores FP for each electricity source

EU28 Metal Ores FP per Electricity Group						
		Coal	Gas	Nuclear	Hydro	Wind
Metal Ores FPs in kt	FP 1995	599,9	470,5	1164,6	267,5	38,4
	FP 2010	1200,7	1226,0	549,6	201,5	69,5
	FP 2019	1442,3	1219,7	471,3	195,0	46,7
Final EU28 Demand for Electricity in million €	FD 1995	8082,0	6492,0	15904,8	4321,0	189,2
	FD 2010	16552,8	20565,9	23050,8	5700,4	2367,1
	FD 2019	21663,1	21113,1	21996,8	7665,7	2194,0
Intensity GHG kt per million €	I 1995	0,1	0,1	0,1	0,1	0,2
	I 2010	0,1	0,1	0,0	0,0	0,0
	I 2019	0,1	0,1	0,0	0,0	0,0
Growth Rate FP	1990-2010	100%	161%	-53%	-25%	81%
	2010-2019	20%	-1%	-14%	-3%	-33%
		Petroleum and other oil derivatives	Biomass and waste	Solar Photovoltaic	Tide, wave, ocean	Geothermal
Metal Ores FPs in kt	FP 1995	109,1	72,2	0,0	0,8	2,1
	FP 2010	152,9	121,5	1,0	0,1	4,1
	FP 2019	101,3	82,3	1,2	0,2	3,3
Final EU28 Demand for Electricity in million €	FD 1995	2089,2	627,2	0,3	18,1	48,0
	FD 2010	2962,0	2281,6	11,2	17,5	93,6
	FD 2019	2751,1	1798,7	35,7	23,5	127,4
Intensity GHG kt per million €	I 1995	0,1	0,1	0,1	0,0	0,0
	I 2010	0,1	0,1	0,1	0,0	0,0
	I 2019	0,0	0,0	0,0	0,0	0,0
Growth Rate FP	1990-2010	40%	68%	3364%	-90%	93%
	2010-2019	-34%	-32%	15%	126%	-18%
Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)						

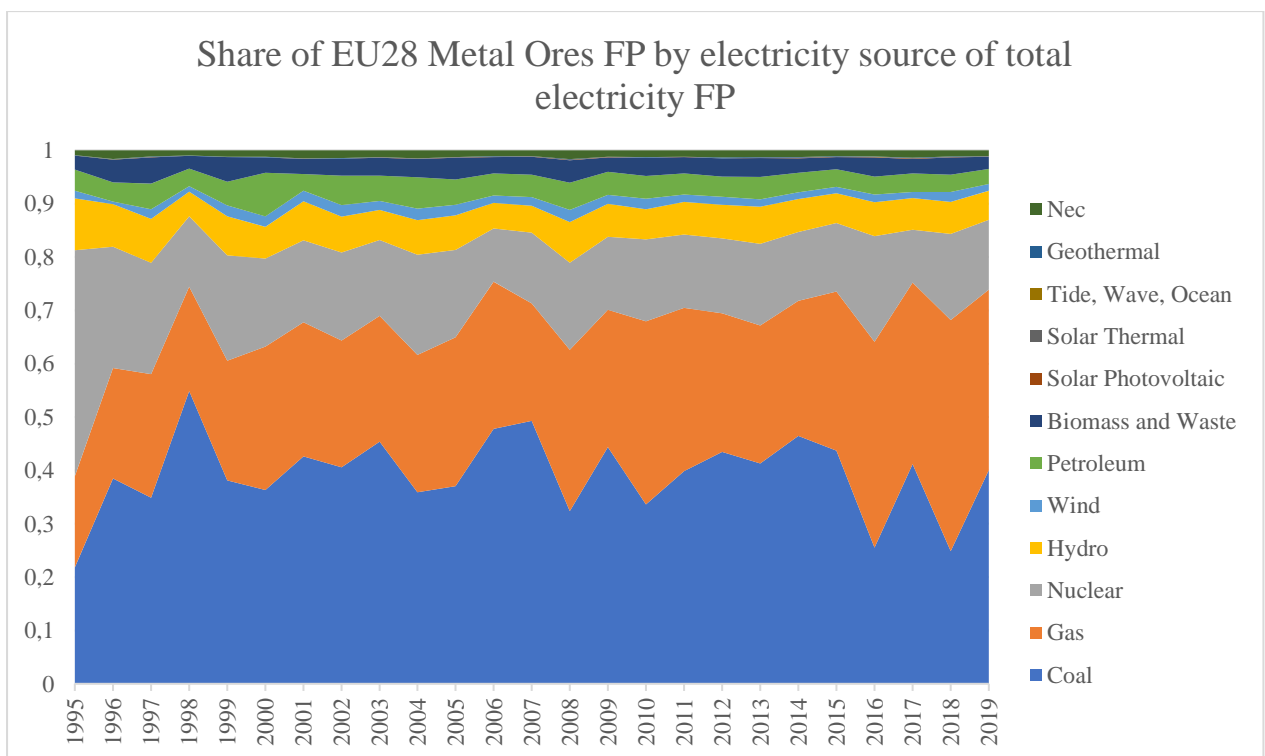
A 4 - Minerals FP for each electricity source

EU28 Minerals FP per Electricity Group						
		Coal	Gas	Nuclear	Hydro	Wind
Minerals FPs in kt	FP 1995	16358,0	3280,6	2089,1	956,8	111,3
	FP 2010	4789,0	4242,1	2105,2	1008,5	274,1
	FP 2019	6007,1	3971,8	2327,0	869,3	265,1
Final EU28 Demand for Electricity in million €	FD 1995	8082,0	6492,0	15904,8	4321,0	189,2
	FD 2010	16552,8	20565,9	23050,8	5700,4	2367,1
	FD 2019	21663,1	21113,1	21996,8	7665,7	2194,0
Intensity Minerals kt per million €	I 1995	2,0	0,5	0,1	0,2	0,6
	I 2010	0,3	0,2	0,1	0,2	0,1
	I 2019	0,3	0,2	0,1	0,1	0,1
Growth Rate FP	1990-2010	-71%	29%	1%	5%	146%
	2010-2019	25%	-6%	11%	-14%	-3%
		Petroleum and other oil derivatives	Biomass and waste	Solar Photovoltaic	Tide, wave, ocean	Geothermal
Minerals FPs in kt	FP 1995	679,9	338,3	0,1	1,5	15,5
	FP 2010	661,2	444,5	2,1	0,4	19,3
	FP 2019	537,5	370,1	9,1	1,4	22,8
Final EU28 Demand for Electricity in million €	FD 1995	2089,2	627,2	0,3	18,1	48,0
	FD 2010	2962,0	2281,6	11,2	17,5	93,6
	FD 2019	2751,1	1798,7	35,7	23,5	127,4
Intensity Minerals kt per million €	I 1995	0,3	0,5	0,3	0,1	0,3
	I 2010	0,2	0,2	0,2	0,0	0,2
	I 2019	0,2	0,2	0,3	0,1	0,2
Growth Rate FP	1990-2010	-3%	31%	2229%	-73%	24%
	2010-2019	-19%	-17%	336%	244%	18%
Author's own calculation based on data from EXIOBASE 3 (Stadler et al., 2021)						

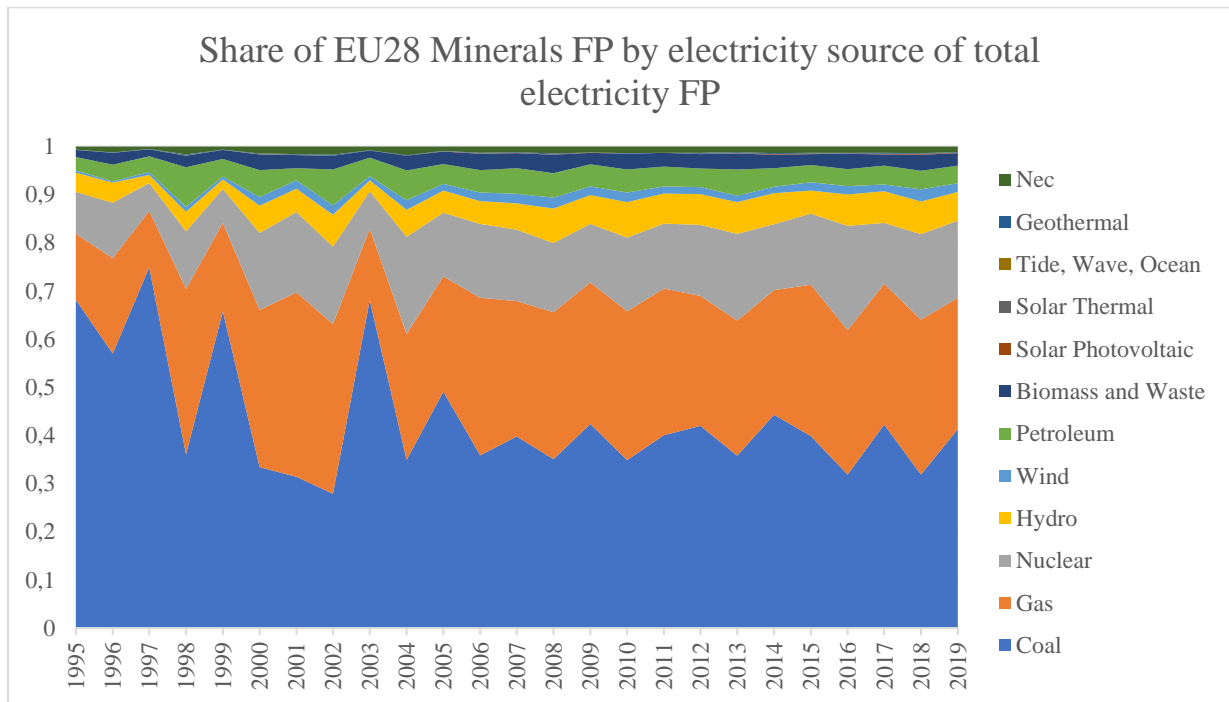
A 5 - Share of total GHG FP for each electricity source



A 6 - Share of total metal ores FP for each electricity source



A 7 - Share of total minerals FP for each electricity source



A 8 - Growth rates of 23 environmental stressors FP for non-renewable sources

