Life cycle assessment of electricity generation options

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Executive summary

Well-informed energy policy design is key to reaching decarbonisation targets, and to keeping global warming under a 2°C threshold. In particular, low-carbon electricity provision for all is an essential characteristic of a 2°C-compatible energy system, as the IPCC shows that the most ambitious climate mitigation scenarios entail the electrification of most of our economy [1]. Therefore, understanding the full scale of potential impacts from current and future electricity generation is required, in order to avoid "impact leakage", i.e. increasing non-climate environmental pressure while reducing greenhouse gas emissions. Life cycle assessment allows the evaluation of a product over its life cycle, and across a wide range of environmental indicators – this method was chosen to report on the environmental profiles of various technologies.

Candidate technologies assessed include coal, natural gas, hydropower, nuclear power, concentrated solar power (CSP), photovoltaics, and wind power. Twelve global regions included in the assessment, allowing to vary load factors, methane leakage rates, or background grid electricity consumption, among other factors.

Results for **greenhouse gas (GHG) emissions** are reported on Figure 1.

Coal power shows the highest scores, with a minimum of 751 g CO_2 eq./kWh (IGCC, USA) and a maximum of 1095 g CO_2 eq./kWh (pulverised coal, China). Equipped with a carbon dioxide capture facility, and accounting for the CO_2 storage, this score can fall to 147–469 g CO_2 eq./kWh (respectively).

A **natural gas combined cycle plant** can emit 403–513 g CO_2 eq./kWh from a life cycle perspective, and anywhere between 49 and 220 g CO_2 eq./kWh with CCS. Both coal and natural gas models include methane leakage at the extraction and transportation (for gas) phases; nonetheless, direct combustion dominates the lifecycle GHG emissions.

Nuclear power shows less variability because of the limited regionalisation of the model, with 5.1–6.4 g CO_2 eq./kWh, the fuel chain ("front-end") contributes most to the overall emissions.

On the renewable side, **hydropower** shows the most variability, as emissions are highly site-specific, ranging from 6 to 147 g CO_2 eq./kWh. As biogenic emissions from sediments accumulating in reservoirs are mostly excluded, it should be noted that they can be very high in tropical areas.

Solar technologies generate GHG emissions ranging from 27 to 122 g CO_2 eq./kWh for **CSP**, and 8.0–83 g CO_2 eq./kWh for **photovoltaics**, for which thin-film technologies are sensibly lower-carbon than silicon-based PV. The higher range of GHG values for CSP is probably never reached in reality as it requires high solar irradiation to be economically viable (a condition that is not satisfied in Japan or Northern Europe, for instance).

Wind power GHG emissions fluctuate between 7.8 and 16 g CO_2 eq./kWh for onshore, and 12 and 23 g CO_2 eq./kWh for offshore turbines. Interestingly, these results show that the technologies conventionally called "low-carbon" may not always yield low lifecycle GHG emissions, namely coal and gas equipped with CCS or hydropower.

Most of **renewable** technologies' GHG emissions are **embodied in infrastructure** (up to 99% for photovoltaics), which suggests high variations in lifecycle impacts due to raw material origin, energy mix used for production, transportation modes at various stages of manufacturing and installation, etc. As impacts are embodied in capital, load factor and expected equipment lifetime are naturally highly influential parameters on the final LCA score, which may significantly decrease if infrastructure is more durable than expected.

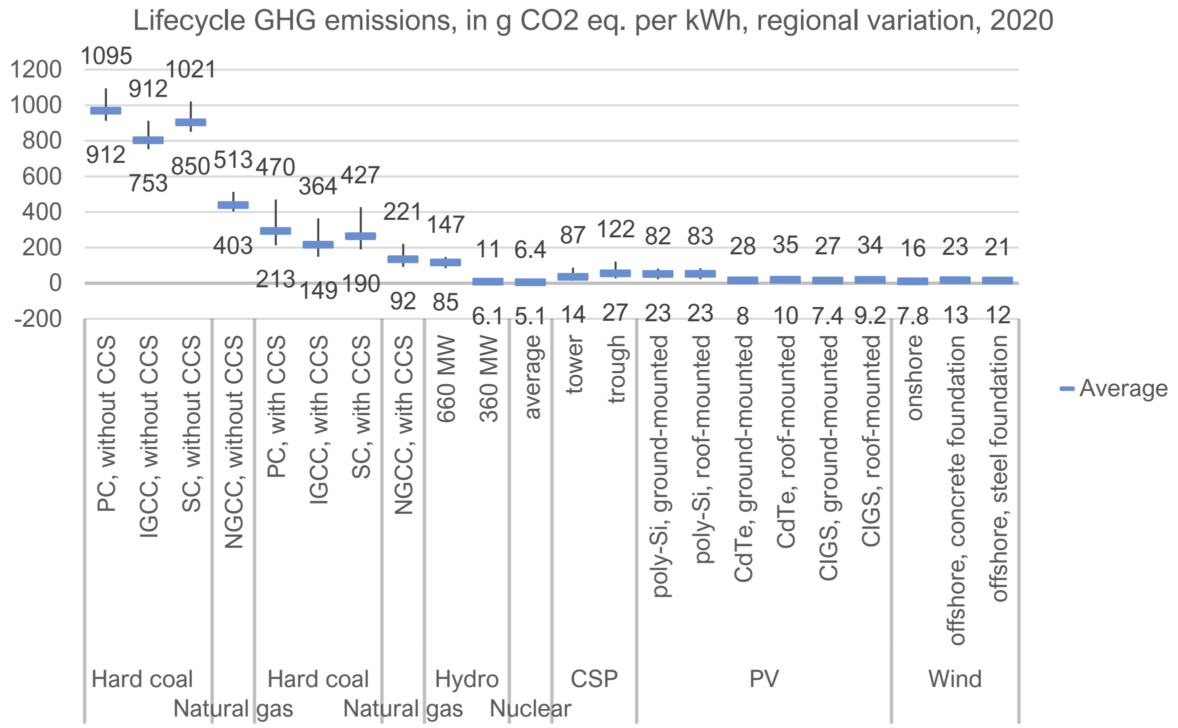

Figure 1. Lifecycle greenhouse gas emission ranges for the assessed technologies.

All technologies display very low **freshwater eutrophication** over their life cycles, with the exception of coal, the extraction of which generates tailings that leach phosphate to rivers and groundwater. CCS does not influence these emissions as they occur at the mining phase. Average P emissions from coal range from 600 to 800 g P eq./MWh, which means that coal phase-out would virtually cut eutrophying emissions by a factor 10 (if replaced by PV) or 100 (if replaced by wind, hydro, or nuclear).

Ionising radiation occurs mainly due to radioactive emissions from radon 222, a radionuclide present in tailings from uranium mining and milling, or coal extraction. Coal power is a potentially significant source of radioactivity, as coal combustion may also spread radionuclides such as radon 222 or thorium 230 (highly variable across regions). Growing evidence that other energy technologies emit ionising radiation over their life cycle has been published, but data was not collected for these (see Box 5 and [2]).

Human toxicity, non-carcinogenic, has been found to be highly correlated with the emissions of arsenic ion linked with the landfilling of mining tailings (of coal, copper), which explains the high score of coal power on this indicator. Carcinogenic effects are found to be high because of emissions of chromium VI linked with the production of chromium-containing stainless steel – resulting in moderately high score for CSP plants, which require significant quantities of steel in solar field infrastructure relatively to electricity generated.

Land occupation is found to be highest for concentrated solar power plants, followed by coal power and ground-mounted photovoltaics. Variation in land use is high for climate-dependent technologies as it is mostly direct and proportional to load factors: 1-to-5 for CSP, 1-to-3.5 for PV, and 1-to-2 for wind power. The same variations can be found for water and material requirements. Lifecycle land occupation is minimal for fossil gas, nuclear and wind power. The land occupation indicator is originally in “points”, a score reflecting the quality of soil occupied, but values in m2-annum (m2a) are also provided in section 7.2.2.

Water use (as dissipated water) was found high for thermal plants (coal, natural gas, nuclear), in the 0.90–5.9 litres/kWh range, and relatively low otherwise, except for silicon-based photovoltaics, as moderate water inputs are required in PV cell manufacturing.

Material resources are high for PV technologies (5–10 g Sb eq. for scarcity, and 300–600 g of non-ferrous metals per MWh), while wind power immobilises about 300 g of non-ferrous metals per MWh. Thermal technologies are within the 100–200 g range, with a surplus when equipped with carbon capture. Finally, **fossil resource** depletion is naturally linked with fossil technologies, with 10–15 MJ/kWh for coal and 8.5–10 MJ/kWh for natural gas.

Uncertainties have not been precisely characterised in this exercise, which only takes into account regional variations (and time variations: all technologies' GHG emissions will decrease as the grid decarbonises). Additionally, storage and grid reinforcement will become vital elements of the decarbonisation strategies across the world, as we do not explicitly assess the impacts of grid & storage, we provide elements showing that the additional environmental impact of such infrastructure may be non-negligible relative to the impact of the technologies that they support.

With no exception, *every* electricity generation technology generates environmental impacts over its life cycle; and these impacts may vary widely with implementation site and other design choices. Proper energy policy should consider site-specificity by conducting lifecycle assessments that consider local conditions and potential prospective changes.

1 Introduction

The substantial change in global electricity generation modes, driven by the double constraint of depleting fossil resources and upcoming climate emergency, is pressing nations to devise low-carbon energy policies. Electrification of the global economy combined with the rapid decarbonization of the grid has been identified by the Intergovernmental Panel on Climate Change (IPCC) as a key measure to reduce greenhouse gas (GHG) emissions and keep global warming under 1.5°C or 2°C (see Figure 2.14 in Rogelj, Shindell [1]). Global energy sector activities, from extraction, conversion, intermediate and final use, accounts for roughly three quarters of greenhouse gas emissions [3], mainly due to the combustion of coal, natural gas, and oil products; most of this combustion is used today to produce electricity. In 2019, 17 PWh electricity was produced from fossil fuels, 2.8 from nuclear power, and 7.2 from renewable power (Figure 2).

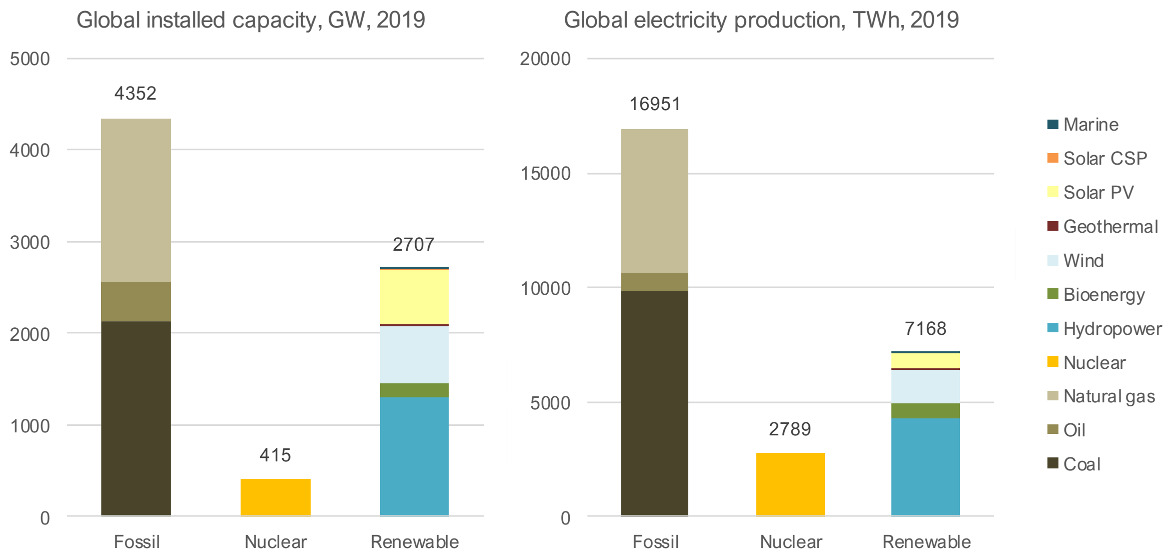

Figure 2. Global installed capacity, and production, of electricity-generating plants in 2019. Source: International Energy Agency [4].

This report presents an assessment of various utility-scale technologies for electricity generation, regarding their potential environmental impacts on human health, ecosystems, and their resource requirements. The objectives of this report are: *first*, to offer an update to the existing data of [5], by using the latest values in renewable efficiencies, electricity mixes as well as the value chain for nuclear power; *second*, to explore in details where environmental impacts (chiefly greenhouse gas emissions, and a few select others) occur within each technology's scope, and *third*, to identify the reasons for variations in impact. A cross-comparison of technologies is proposed in the penultimate section, then a discussion concludes the report.

Cradle-to-grave analyses of electricity systems are critical to identify potential problem-shifting along supply chains and technology lifecycles (e.g. reducing operation impacts while increasing those of construction), or across types of environmental burden (e.g. reducing greenhouse gas emissions while increasing material requirements or land use). Life cycle assessment (LCA) is a transparent and rigorous method that can provide insight into the potential environmental impacts of differing low carbon technologies and the contribution of these technologies to global sustainable development. The method is comprehensive and appropriate for a comparative analysis of technologies because it considers potential environmental impacts using a cradle-to-grave analysis. As shown in Hertwich, de Larderel [5], considering all environmental dimensions of electricity technologies may lead to environmental co-benefits and/or increased impacts, whereby adopting climate change mitigation strategies can also decrease or increase particulate matter emissions, human or ecotoxicity, eutrophication, mineral or fossil resource depletion, or land and water use. Depending on a country's or region's configuration, options may differ.

Recognising the urgency in designing efficient energy policies to comply with a climate neutrality pathway, the UNECE has initiated this work to identify and quantify the environmental impacts for various technologies in the context of UNECE regions. In particular, material requirements (although not "environmental impacts" *sensu stricto*) have been analysed through the LCA lens. Furthermore, the life cycle inventory update for nuclear power has been performed with the support of the World Nuclear Association (WNA), and consultations with their expert network. The work on conventional nuclear technologies provides a much needed update upon data currently available in LCA databases (reflecting the higher share of in-situ leaching and the phasing out of enrichment through diffusion) and also explains the imbalance between the nuclear-specific data (section 7.3 in Annex) and the rest of the technologies studied. Finally, biopower has been left out of the scope due the complex modelling required to assess the various [feedstock type–agricultural techniques–conversion technology] combinations. We note that a consensus is yet to be reached among scientists regarding the actual climate neutrality of biomass as an energy carrier [6-8].

2 Method

2.1 Description

The environmental evaluation of technologies is carried out using life cycle assessment (LCA). LCA it both a method and a tool that relies on the exhaustive accounting of environmental flows that are directly or indirectly linked with a well-defined product system. A first principal property of LCA is the completeness of its approach, sometimes qualified as “cradle-to-grave”. This guarantees that all flows of materials and energy, waste and emissions, are accounted for from extraction to end-of-life treatment. The second main characteristic of LCA is its multicriteria nature: as many elementary flows as realistically possible are accounted for, including natural resources, or emissions to air, water, or soil.

LCA is ISO-standardized, and used in increasingly many international initiatives and regulations to define the environmental performance of a product or a service, among others: the GHG Protocol (organizational carbon footprinting) [9], the “EU taxonomy for sustainable activities” (guidelines for sustainable investment) [10], or the EN 15804 standard (rules for environmental product declarations). The ISO 14040 standard series offers a minimum of harmonization in LCA; without guaranteeing direct comparability between ISO-compliant LCA studies, it ensures that LCA studies be reproducible, and transparent. LCA is defined as a four-step technique, including namely: (i) the goal and scope definition, (ii) the life cycle inventory modelling, (iii) the life cycle impact assessment, and (iv) the interpretation phase.

2.2 Goal and scope definition

The objective of this study is to assess the environmental impacts of the functional unit, namely **the delivery of 1 kWh of electricity to a grid**, on a global average (unless otherwise specified), for the year 2020. The study therefore excludes load balancing systems such as storage elements and additional grid connections. The study aims at comparing the following electricity-generating technologies:

- Coal and natural gas, with and without carbon dioxide capture and storage,
- Wind power, onshore and offshore,
- Solar power, photovoltaics, polycrystalline and thin-film,
- Concentrated solar power,
- Hydropower,
- Nuclear power, conventional.

We choose to exclude biomass in this exercise due to the complexity of modelling the various feedstock-agricultural practices-conversion-technology combinations. Two “extreme” cases can be found in Gibon, Hertwich [11] for lignocellulosic feedstocks, namely forest residues and purpose-grown energy crops. The variation in impact is wide and impacts highly dependent on parameters such as irrigation or agricultural practices – which would require a detailed modelling at the regional level.

2.3 Life cycle inventory modelling

Basic data sources include the UNEP Green Energy Choices study, Gibon et al. (2017) as well as the ecoinvent 3.7 database. These inventories are then adapted with more recent data, collected through expert consultation, with the support of the UNECE and the World Nuclear Association (WNA). The data collected is presented in this report. Sources for adapting the life cycle inventories (LCIs) include scientific literature, technical reports, and best estimates from expert elicitation.

Regionalization is performed, namely through the adaptation of background electricity mixes, as well as the technological description of a few processes (e.g. cement production) as well as local conditions dictating load factors, namely irradiance for solar technologies, wind regimes for wind power (based on average regional data from existing wind farms), as well as average regional load for hydropower plants. In practice, it means that the technology description is identical in each region but the origin of electricity or fuel inputs, and performance factors, have been adapted. Only the nuclear fuel cycle is modelled with global data, and is only representative of the average conventional power plant as of 2020.

Table 1. Summary of life cycle inventories' scopes, per type of technology.

Technology		Included	Excluded
Coal power	without CCS	Energy carrier supply chain, from extraction to combustion, including methane leakage Infrastructure construction, operation, and dismantling (energy inputs and waste production) Connection to grid	Potential recycling of dismantled equipment
	with CCS	Same as above, plus Capture equipment and chemicals, transportation of captured CO2 and storage infrastructure (well)	Same as above, plus Potential emissions (leakage) from captured CO2 transportation or from the storage site
Natural gas power	without CCS	Energy carrier supply chain, from extraction to combustion, including methane leakage Infrastructure construction, operation, and dismantling (energy inputs and waste production) Connection to grid	Potential recycling of dismantled equipment
	with CCS	Same as above, plus Capture equipment and chemicals, transportation of captured CO2 and storage infrastructure (well)	Same as above, plus Potential emissions (leakage) from captured CO2 transportation or from the storage site
Hydropower		Construction, site preparation, transportation Connection to grid	Potential recycling of dismantled equipment Site-specific biogenic emissions of CO_2 and CH_4
Nuclear power		Fuel element supply chain (from extraction to fuel fabrication) Core processes (construction and decommissioning of power plant, as well as operation) Back-end processes: spent fuel management, storage, and final repository Connection to grid	Potential recycling of dismantled equipment Reprocessing of spent fuel (conservative assumption that all fuel is primary)
Concentrated solar power		Infrastructure, site preparation and occupation, operation and maintenance (including 6-hour storage) Decommissioning (energy inputs and waste production) Connection to grid	Potential recycling of dismantled equipment
Photovoltaics		Infrastructure, site preparation and occupation, operation and maintenance Decommissioning (energy inputs and waste production) Connection to grid	Potential recycling of dismantled equipment
Wind power		Infrastructure, site preparation and occupation, operation and maintenance	Potential recycling of dismantled equipment

	Decommissioning (energy inputs and waste production) Connection to grid

Inventories are regionalised according to the classification used in the MAgPIE-REMIND integrated assessment model (IAM). This list of regions (Table 2) is used to match electricity mixes for electricity inputs, the adaptation of load factors for concentrated solar power, photovoltaics, wind power and hydropower, as well as the region-specific sourcing of coal and natural gas for fossil fuel technologies.

Table 2. Region classification. UNECE regions in bold, used for detailed assessment in Section 4.

REMIND regions	**Code**
Canada, Australia & New Zealand	**CAZ**
China	CHA
European Union	**EUR**
India	IND
Japan	JPN
Latin America	LAM
Middle East and NorthAfrica	MEA
Non-EU member states	**NEU**
Other Asia	OAS
Reforming countries	**REF**
Sub Saharan Africa	SSA
United States	**USA**

2.4 Life cycle impact assessment

Life cycle impact assessment involves the characterization of potential impacts and selection of impact assessment categories based on their contribution to the normalized and weighted results of the analysis. Two approaches can be used to characterize environmental impacts, either a *midpoint* approach and midpoint indicators, which is recommended by the EC Environment Footprint Guidelines [12, 13] or an *endpoint* approach and endpoint indicators. These approaches differ in terms of objectives and robustness; a comprehensive LCA may display results using both approaches to ensure that the conclusions remain the same. This study characterizes results using both a midpoint and endpoint approach.

Note: we use the term “impact” as shorthand for “potential impact”, as defined in ISO standards. In LCA, the word “impact” (and associated terms such as “impact assessment” or “impact category”) is therefore primarily associated with the **potential** detrimental effects that a substance or a stress may have on the environment, human health or resources. Specifically, “*Only potential environmental impacts can be regarded, as real impacts are influenced by factors that usually are not included in the study.*” [14] [15] adds that “***The LCIA does not necessarily attempt to quantify any actual, specific impacts associated with a product, process, or activity***. *Instead, it seeks to establish a linkage between a system and potential impacts.”*

2.4.1 Midpoint characterisation

Midpoint characterization focuses on the potential environmental impacts associated with **actual biophysical phenomena occurring through the emissions of substances**. The International Life Cycle Data (ILCD) System proposes 19 categories commonly used in LCA to describe and model potential environmental impacts of technologies using a midpoint approach (see full list in Appendix 7.2, Table 13, which presents the whole set of results). An analysis was completed to determine the potential environmental impacts associated with each technology and the contribution of each impact category to overall environmental impacts (Figure 54). The impact assessment categories that contributed to greater than 80% of the total environmental impact of each technology were selected for presentation and comparison in Section 4. These selected impact assessment categories and their key assumptions are shown in Table 3. The “Reference” column contains sources to the underlying models of each category.

Table 3: Selected environmental indicators for Life Cycle Impact Assessment

Category	Unit	Reference	Description
Climate change	kg CO_2 eq.	IPCC (2013)	Radiative forcing as global warming potential, integrated over 100 years (GWP100), based on IPCC baseline model.
Freshwater eutrophication	kg P eq.	EUTREND, Struijs, Beusen [16]	Expression of the degree to which the emitted nutrients reach the freshwater end compartment. As the limiting nutrient in freshwater aquatic ecosystems, a surplus of phosphorus will lead to eutrophication.
Ionising radiation	kBq ^{235}U eq	Frischknecht, Braunschweig [17]	Human exposure efficiency relative to ^{235}U radiation. The original model is Dreicer, Tort [18] and follows the linear no-threshold paradigm to account for low dose radiation (details in Box 5).
Human toxicity	CTUh (comparative toxic units)	USEtox 2.1. model Rosenbaum, Bachmann [19]	The characterization factor for human toxicity impacts (human toxicity potential) is expressed in comparative toxic units (CTUh), the estimated increase in morbidity in the total human population, per unit mass of a chemical emitted, assuming equal weighting between cancer and non-cancer due to a lack of more precise insights into this issue. Unit: [CTUh per kg emitted] = [disease cases per kg emitted][1]
Land use	points	LANCA model, Bos, Horn [20]	The LANCA model provides five indicators for assessing the impacts due to the use of soil: 1. erosion resistance, 2. mechanical filtration, 3. physicochemical filtration, 4. groundwater regeneration and 5. biotic production
Water resource depletion	m^3	Swiss Ecoscarcity Frischknecht, Steiner [21]	Water use related to local consumption of water. Note: only air emissions are accounted for. **In this method, all flows have an identical characterisation factor of 42.95 m^3/m^3 – we therefore choose to account for these flows uncharacterised, i.e. 1 m^3/m^3.**
Mineral, fossil and renewable resource depletion	kg Sb eq.	Van Oers, De Koning [22]	Scarcity of resource in relation to that of antimony. Scarcity is calculated as « reserve base ».

2.4.2 Material requirements

The last indicator in Table 3 characterises the depletion of mineral resources via modelling the scarcity of each resource elementary flow compared to a reference flow (antimony). As the scarcity model is limited in scope and needs a regular update to match annual fluctuations for the production of each metal [23], we also propose to display the raw inventory of select materials. The list of these materials is adapted from [24] and includes: aluminium, chromium, cobalt, copper, manganese, molybdenum, nickel, silicon, and zinc.

2.4.3 Endpoint characterisation

Endpoint indicators aim at conveying the **effects that these phenomena cause on ecosystems**, human health, or natural resource depletion (coined “areas of protection”). Damage on ecosystems and human health is shown in Section 4.9.1. The “resources” category consists in an aggregation of fossil and metal depletion indicators, they are already fully shown via midpoint characterisation and not replicated. The LCIA methodology used for this calculation is ReCiPe version 1.13. As a reminder, the UNEP IRP report “Green Energy Choices” uses a former version of ReCiPe, version 1.08.

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[1] From USEtox FAQ, available at https://usetox.org/faq

In this version of the ReCiPe methodology, impacts are directly converted into “points”, based on the global average impacts (in disability-adjusted life years, DALY, for human health, and species-year, for ecosystem services) of 1 person over one year. If a given technology has an impact of 3 points per MWh, it means that it has the same effect as the impacts of 3 persons over 1 year, or 1 person over 3 years, through the various midpoint-to-endpoint pathways. DALY-to-point and species-year-to-point coefficients can be found at https://www.rivm.nl/en/documenten/normalization-scores-recipe-2016.

2.4.4 Normalisation and weighting

Normalised and weighted results are also calculated in this exercise. Normalised results are obtained by multiplying each “midpoint” indicator by a coefficient based on a single individual’s share of the corresponding environmental impact. In other words, the normalised impact is the sum of all indicator scores divided by the footprint of a single individual. This footprint may change depending on the scope, for example, if an average European has a GHG footprint of about 10 tonnes CO_2 eq./year, then a 1 ton CO_2 eq. emission will be normalised to 1/10 = 0.1, whereas a global scope will yield a higher number as the global average per-capita carbon footprint is lower. Weighting denotes the more subjective ranking of impact categories, and a step through which normalised results are multiplied with variable coefficients (weights) to yield a single score.

According to LCA software developers and consultants “PRé”, “*Weighting is the optional fourth and final step in Life Cycle Impact Assessment (LCIA), after classification, characterization and normalization.* ***This final step is perhaps the most debated****. Weighting entails multiplying the normalized results of each of the impact categories with a weighting factor that expresses the relative importance of the impact category.*”[2]

Normalisation and weighting are also applied directly to the endpoint indicators, which are aggregated into DALYs (for damage to human health) or species-year (damage to ecosystems) in a first step, then normalised and weighted, resulting in scores expressed in “points” instead of absolute units.

2.5 Software implementation

The python package brightway2 [25] was used to compute the impact assessment results. The ecoinvent 3.7 database [26] has been used as background data for life cycle inventories. This marks a clear difference with the “Green Energy Choices” report, where data relied both on ecoinvent 2.2 [27], as well as EXIOBASE 2 [28], to complement life cycle inventories where physical flows were unavailable. Using a matrix-based hybrid LCA approach is significantly more data-intensive with ecoinvent 3.7, as in matrix form, ecoinvent 3.7 is about 19000 × 19000 elements, whereas ecoinvent 2.2 was 4000 × 4000. An alternative was therefore chosen.

Life cycle inventories from the “Green Energy Choices” report were imported in their MATLAB format, and parsed into the *brightway* inventory format [25] through an ad-hoc conversion script. The relinking from ecoinvent 2.2 to 3.7 has been performed, both for technosphere and biosphere elementary flows. Unlike the original inventory format, the brightway format ensures shareability and reproducibility, with an open source mindset (conversely, MATLAB is proprietary). Further modifications were then brought upon the datasets as described in the technology-specific sections.

The prospective LCA module *premise* (Sacchi et al., in preparation) was used to model the evolution of electricity mixes and industry efficiency, in a similar fashion as in THEMIS [29], but with a much higher degree of flexibility. Using premise guarantees that background scenarios align with various socio-economic pathways by using REMIND and IMAGE, two integrated assessment models (IAMs) including a detailed energy system model developed respectively by the Potsdam Institute for Climate Research (PIK) and the Netherlands Environmental Assessment Agency (PBL).

Calculations were therefore made in a pure process-LCA fashion, with a changing background, depending on the outputs of the various IAM scenarios. In the present work, this *does not mean* that the

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[2] A longer discussion on the relevance and interpretation of normalisation and weighting is available at https://pre-sustainability.com/articles/weighting-applying-a-value-judgement-to-lca-results/

new technologies modelled become part of the background electricity mixes (as was done in the THEMIS model). On the other hand, multiple prospective scenarios are testable to assess the per-kWh impact of electricity technologies.

2.6 Caveats

Life cycle assessment is a powerful tool *within its domain of application*, and as long as uncertainties, variabilities, and incompleteness are well-understood. This report is focused on potential impacts from the expected routine and non-routine circumstances that either have occurred or are predicted to occur during the life cycle of the low carbon electricity generation technologies modelled. The potential environmental impacts of catastrophic failures that could occur in the future are not modelled. Only impacts due to the *expected* emissions of substances and waste, or the consumption of energy and materials are therefore considered in this report. Likewise, potential impacts not assessed by the LCIA (e.g. specific biodiversity-related impacts, noise or aesthetic disturbance) are not assessed.

By nature, LCA relies on data compiled from many different sources, from existing databases, to technical reports, expert consultation, or academic literature. LCA guidelines recommends the characterisation of the uncertainty linked with each data point, to be able to estimate the degree of uncertainty of final impact assessment results. By default, we do not characterise the uncertainty of all the flows in the models.

As nuclear power datasets have been refined, attention is brought on the ionising radiation indicator, with a “Box” describing how radioactivity is characterised in LCA. On the data side, radionuclide emissions have not been fully updated, namely regarding the emissions of radon 222 from uranium milling tailings, which end up dominating the emissions over the nuclear fuel cycle – the modelling behind these emissions extends beyond the scope of this work.

Finally, natural regional and temporal variability of systems implies that the collected data cannot be accurately representative of specific, real cases. Parameterised and dynamic models exist to take into account these potential variabilities on a site-specific basis.

3 Technologies

This section presents the list of technologies assessed in the LCA model. Each section contains a short technology description (status of the technology, available designs, potential current issues and challenges), a subsection on life cycle inventory data, and a presentation of baseline (2020) results for the EU28 region (a comparison of region-specific impacts is proposed in the next section).

3.1 Coal

Coal-fired electricity, with an annual production of 9 PWh (34% of the global total), remains a substantial source of energy around the world [30]. As a result of this high reliance on hard coal and lignite, coal power plants emit about 20% of global greenhouse gas emissions [31]. Coal, especially lignite, is the second highest carbon-emitting electricity source per kWh, after oil (which accounts for less than 5% of global electricity production). Despite international and national pledges to phase out unabated coal power, it is estimated that current commitments to coal energy infrastructure represent the majority of energy-related future emissions, eating up a significant share of the remaining global carbon budget – see Figure 3 [32]. A few causes explain why coal continues to dominate the global energy portfolio. First, institutional lock-in is slowing down phase-out processes, even in industrialised countries [33]. Second, cheap feedstock remains a principal reason for coal popularity around the world; it is therefore a strategic energy carrier for countries with enough resources. Carbon dioxide capture and storage (CCS) retrofit of existing plants could secure a safer transition to a low-carbon electricity grid globally, hence a sensible share of the most ambitious climate mitigation scenarios includes CCS [1]. This technology could cut per-kWh GHG emissions of coal power plants by 60%, all the while increasing feedstock consumption (termed “energy penalty”, see Singh [34]) and other environmental impacts, depending on the capture technology [35].

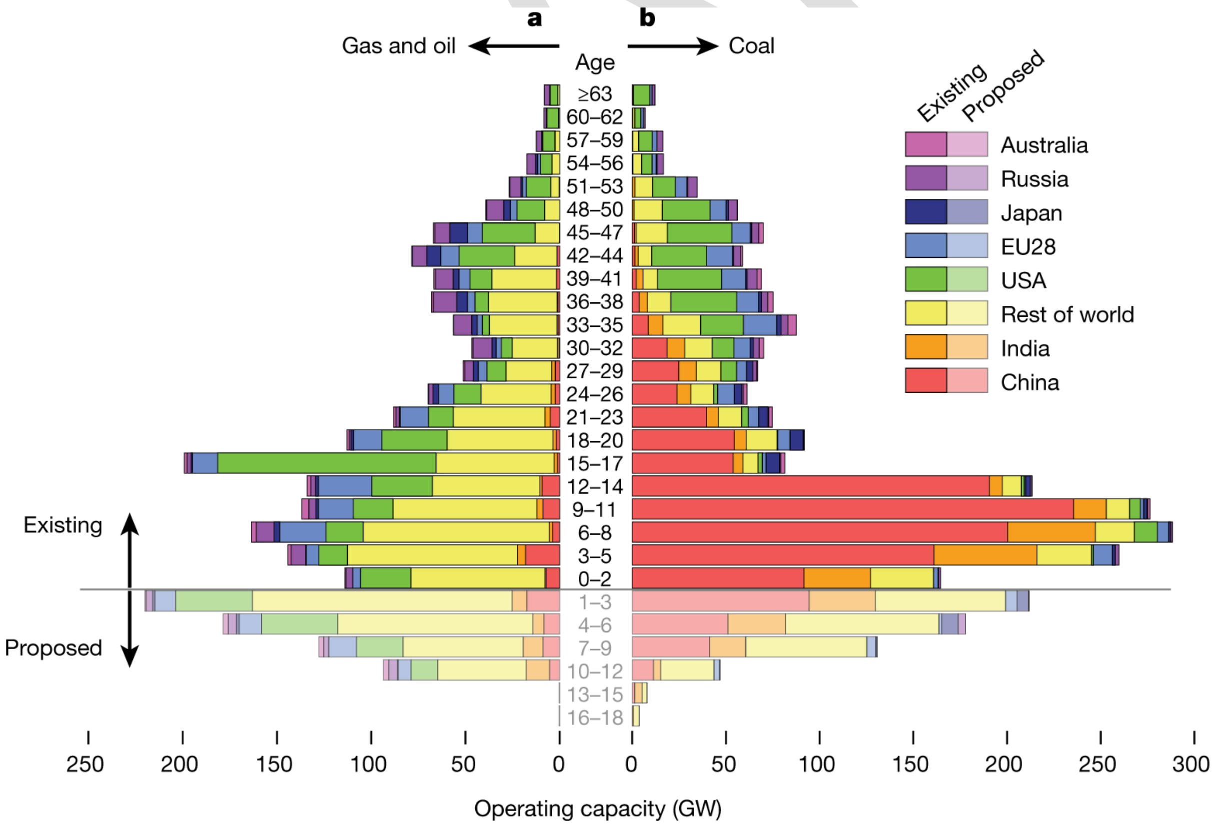

Figure 3. Operating capacity of existing and future fossil fuel power plants, oil and gas on the left, coal on the right. Source: Tong, Zhang [32].

3.1.1 Technology description

Coal power plants are commercially available in various designs. The overwhelming majority of power plants today use the “pulverized coal” (PC) technology, which consists in preparing coal for combustion by finely grinding it, and operating a steam turbine. The average overall plant efficiency of subcritical technologies (the most common version of PC plants) is 35%. Supercritical power plants are also based on the PC technologies, but they achieve much higher internal pressures and temperatures than their subcritical variants. The high pressure forces water to remain liquid instead of turning into vapour, which allows higher efficiencies, typically up to 40%. These two PC variants, subcritical and supercritical, are modelled in the present exercise. A third technology is added to the list, namely integrated gasification combined cycle (IGCC). The IGCC technology relies on turning coal into a synthetic gas (instead of powder) before combustion. The process allows overall efficiencies typically in the 40-45% range, with claims reaching 48% [36]. These three technologies are assessed with and without CCS equipment. See Box 1 for a discussion on coal power plant efficiencies and how it may have led to a potential issue in emission reporting for coal power plants.

3.1.2 Life cycle inventory

Data for the modelling of fossil-fuelled plants have been collected from Hertwich, de Larderel [5]. Inventories are all originally built from technical reports published by the National Energy Technology Laboratory (NETL) of the United States. Main parameters are shown in Table 4. Only **hard coal** is assessed as a feedstock, lignite or peat are not included in this analysis.

Table 4. Coal power plants characteristics, from [5], original source: [37].

Parameter	Pulverised	Supercritical	IGCC
Nameplate capacity (MW) (with CCS)	550		629 (497)
Capacity factor	85%		80%
Net efficiency (with CCS)	36.8% (26.2%)	39.3% (28.4%)	42.1% (31.2%)
CO2 capture efficiency	90%		
Flue gas desulphurisation efficiency	98%		Sulphur captured in Selexol process
Selective catalytic reduction efficiency	86%		-
Particulate matter removal efficiency	99.8%		Cyclone and barrier filter
Mercury reduction efficiency	90%		95%

Changes to original inventories

As this study does not use inputs from an IO database, IO inputs have been substituted with their process LCA equivalents when possible. In the case of coal power, this encompasses infrastructure investments, namely for power plants, which have been replaced by a global “market for hard coal power plant” input from ecoinvent 3.7, each scaled to their nameplate capacity relatively to the original plant of 500 MW.

Radioactive emissions at mining and combustion phases have also been included in this model, based on data for China reported in [2]. The Chinese inventory is therefore updated to account for these changes, namely: the emission of ^{222}Rn in the mining phase (from 0.012 to 0.93 kBq/kg coal), and ^{222}Rn (0.008 kBq/kWh), ^{210}Po, ^{210}Pb, ^{226}Ra, ^{234}U, ^{238}U and ^{230}Th (all in the 4.3–8.5 kBq/kWh range) in the combustion phase.

Coal extraction fugitive emissions have been updated in 2018 in the ecoinvent database, based on UNFCCC-declared values in 2017[3].

Regionalisation has been applied to the supply chains, in order to account for the variations in methane leakage rates and efficiencies in different world areas, as shown in Table 5. Electricity inputs are also regionalised to match the REMIND region mix in 2020 (and 2050 in section 4.1.2).

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[3] National inventories are accessible at https://unfccc.int/process-and-meetings/transparency-and-reporting/reporting-and-review-under-the-convention/greenhouse-gas-inventories-annex-i-parties/submissions/national-inventory-submissions-2017

Table 5. Correspondence between technology regions and assumed fossil fuel region of origin.

REMIND region		**Origin of coal, ecoinvent 3.7**		**Origin of natural gas, ecoinvent 3.7**	
CHA	China	CN	China	RoW	Rest of the world
IND	India	IN	India	RoW	Rest of the world
EUR	European Union	Europe, without Russia and Turkey		Europe without Switzerland	
NEU	Non-EU Europe	Europe, without Russia and Turkey		Europe without Switzerland	
USA	United States	RNA	North America	US	United States
CAZ	Canada, Australia, New Zealand	AU	Australia	CA	Canada
JPN	Japan	AU	Australia	JP	Japan
OAS	Other Asia	ID	Indonesia	RoW	Rest of the world
REF	Reforming countries	RU	Russia	RU	Russia
LAM	Latin America	RLA	Latin America	RoW	Rest of the world
MEA	Middle East and Northern Africa	ZA	South Africa	RoW	Rest of the world
SSA	Sub-Saharan Africa	ZA	South Africa	RoW	Rest of the world

3.1.3 Environmental impact assessment

Two life cycle phases dominate the environmental impact of coal power: extraction, and electricity generation (combustion). Resource use, land use, ionising radiation and freshwater eutrophication are caused by hard coal extraction, whereas water use and greenhouse gas emissions are mostly due to the plant operation. These results are shown on Figure 4, grouped by simplified lifecycle phase, “Electricity” (on-site combustion and operation), “Coal extraction” (hard coal supply chain from extraction to delivery at plant), and “Other”, which represents infrastructure (coal power plant and connection to grid).

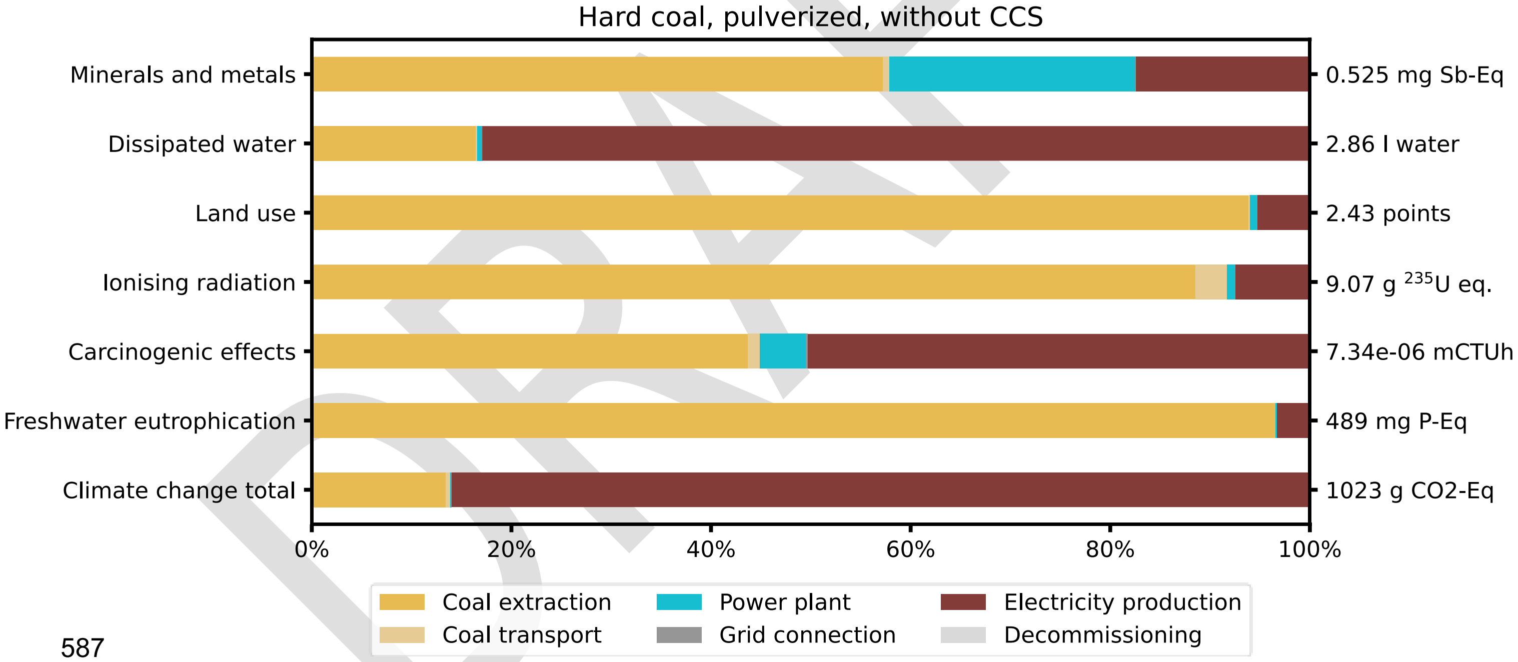

Figure 4. Life cycle impacts from 1 kWh of coal power production, pulverised coal, Europe, 2020.

When equipped with CCS Figure 5, a coal power plant can reduce its direct emissions significantly, which translates into a cut in lifecycle GHG emissions from 1020 to 367 g CO2 eq./kWh, i.e. -64%. On the other hand, other environmental impacts rebound, from +41% (eutrophication) to 78% (water use) – due to an increase in hard coal consumption and use of chemicals for the capture process, as well as the downstream processes of transportation and storage of CO_2 storage in deep geological well.

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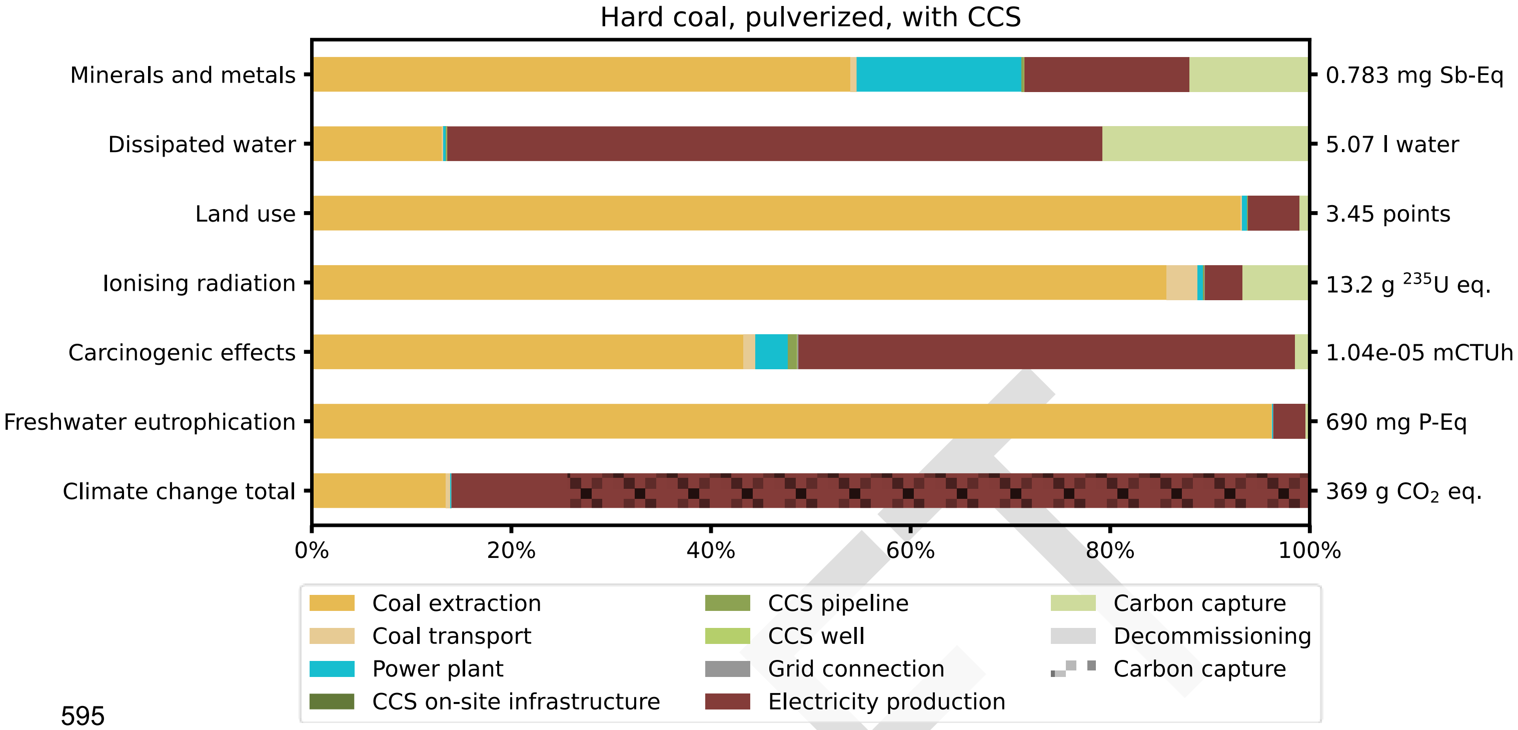
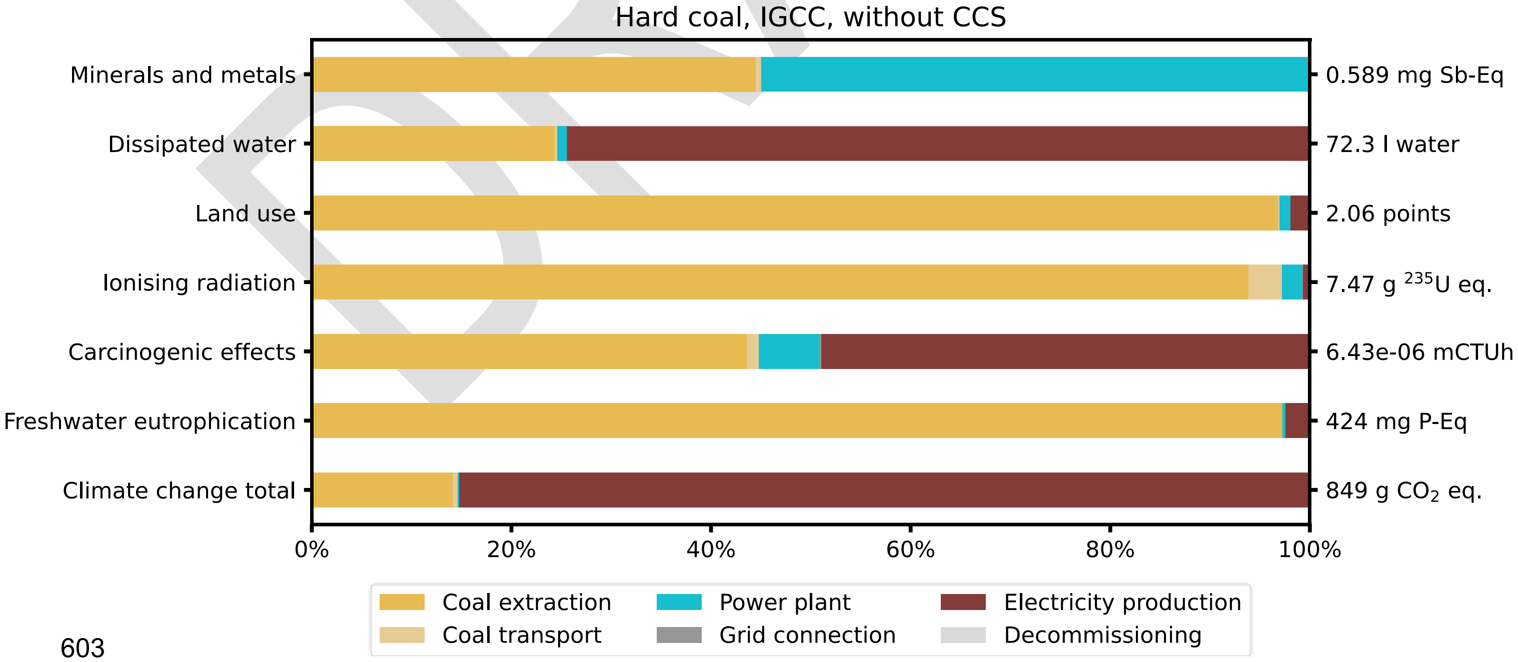


595

596 *Figure 5. Life cycle impacts from 1 kWh of coal power production, pulverised coal with CCS , Europe, 2020. Carbon*
597 *dioxide capture and storage processes are shown in red when positive, in hatched lines when negative.*

598 IGCC plants are more efficient than pulverised coal designs, which explains the lower GHG emission
599 value of 849 g CO_2 eq. (Figure 7). Scores are also lower on all other indicators. In particular, water
600 requirements are significantly lower, with 72 litres per kWh (123 for the PC power plant), 116 litres with
601 CCS (218 for PC).

602 Results for the **supercritical power plants** are shown in Table 14 in Annex (section 7.2).



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604 *Figure 6. Life cycle impacts from 1 kWh of coal power production, IGCC without CCS , Europe, 2020.*

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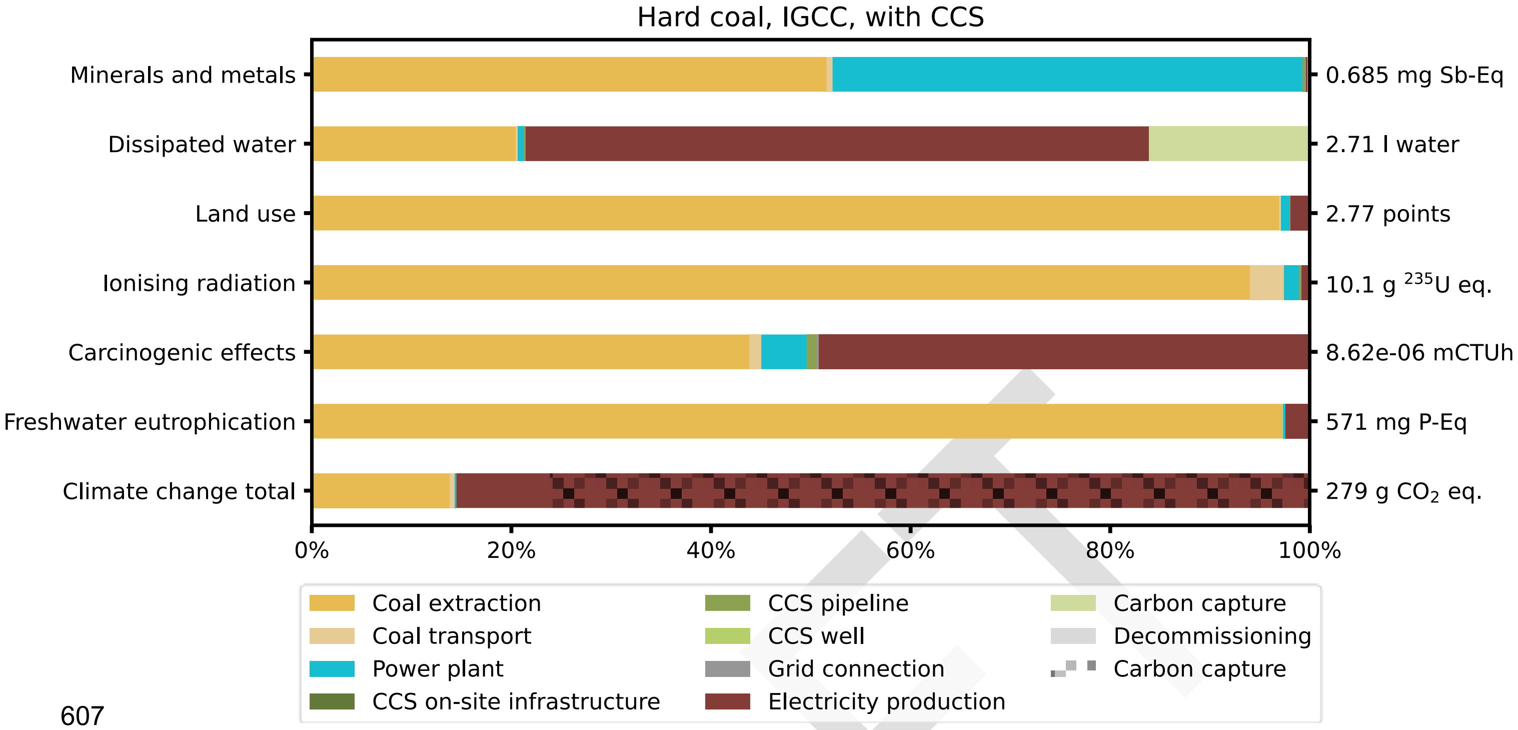

607

608 *Figure 7. Life cycle impacts from 1 kWh of coal power production, IGCC with CCS, Europe, 2020. Carbon dioxide*
609 *capture and storage processes are shown in red when positive, in hatched lines when negative.*

Box 1. Coal in the IPCC AR5

The IPCC Fifth Assessment Report provides a median value of 820 g CO_2 eq./kWh for coal power, over its lifecycle, with a range of 740–910 g CO_2 eq./kWh. Oberschelp, Pfister [38] conducted a plant-by-plant study of virtually all coal-fired power units in the world, and modelled their direct and indirect emissions. They found that the generation-weighted global mean of lifecycle greenhouse gas emissions from coal plants are 1.13 kg CO_2 eq./kWh, with a standard deviation of ± 0.06 kg CO_2 eq./kWh. The difference is considerably high, and deserves a deeper look, namely at the IPCC values.

The IPCC relies on original research as well as a series of reviews, among which the work led by Corsten, Ramírez [39], namely a comparison of LCA studies of coal power with and without CCS, in published literature as of 2012. A major source in this review is a highly-cited study by Viebahn, Nitsch [40], which provides LCA data for certain types of coal power plant designs in Germany, with and without CCS. The authors provide the list of key parameters for each plant type, including nameplate capacity, operating time, efficiency, various costs, fuel CO_2 intensities, as well as the resulting (direct) CO_2 emissions, namely: 676, 662, and 849 g CO_2/kWh for the pulverized coal, IGCC, and pulverized lignite plants respectively, without CCS.

Considering average coal plant thermal efficiencies, below-700 values are virtually impossible to reach without any abatement, in fact, power plant efficiencies in [40] are then-estimates for 2020 and are sensibly above average: 49%, 50%, and 46% respectively for the three plant designs. Whether authors' projections were overly optimistic or turbine-only efficiency (which indeed would fall in the 45-50% range) was used as a proxy to the overall plant efficiency is unknown, but there is a possibility that, from citation to citation, this assumption made its way to the IPCC AR5 report – yielding the 820 lifecycle value. Another major source mentions overoptimistic efficiencies in the 45%-50% range for plants built after 2008, which leads to very low estimates of direct emissions, as low as below the 700 g CO_2/kWh mark [41]. This source explains the lower values of the NREL harmonised LCA for pulverised coal plants (Figure 55).

Last, all these estimates are valid for bituminous coal and anthracite (hard coal) only, the “highest ranks” of coal [42]. Lignite (brown coal) power plants generate higher carbon emissions due to a relatively low heating value. At an average net thermal efficiency of 38% (and older–modern range of 34%–43%), a lignite-fired power plants emits about 1093 (1221–966) g CO_2/kWh, compared to 1001 (849–1084) g CO_2/kWh for a hard coal power plant of a 39% (36%–46%) efficiency [43].

3.2 Natural gas

Natural gas is the second source of global electricity, with an annual production of about 6 PWh, or 23% of all electricity produced in 2020. Per kWh, electricity produced from gas power plants emit less than half the GHG emitted by coal-fired electricity. Additionally, it also emits fewer particles and other pollutants than coal (REF), a characteristic that has made gas power plants interesting candidates to decarbonize coal-based grids globally. While coal electricity generation has decreased from 40% in 2013, to 34% today, natural gas has remained stable in the 20-23% range of global production since 2004.

3.2.1 Technology description

The main technology of power plants used today is the natural gas combined cycle (NGCC), in which heat is recovered from the main gas turbine to run a steam turbine, maximising the overall efficiency by using heat that would otherwise be lost (as it is e.g. in gas “peaker” plants, which only use a gas turbine). NGCC efficiency can range from 50% to 60%. This is the design modelled in this exercise, with and without carbon dioxide capture and storage.

Methane leakage at fossil fuel extraction has been under increased scrutiny as fossil CH_4 emissions have been shown to be systematically underestimated by the extractive industry [44]. As methane is literally natural gas, fugitive emissions from the oil and gas industry are expected; when they occur, they significantly influence the overall greenhouse gas emission profile of gas-fired electricity. However, it has been recently suggested that global (fossil) methane emissions may be driven by the coal mining industry, even after coal is extracted, and mines abandoned [45]. For natural gas, fugitive emissions can also occur after extraction, namely in pipelines. A high enough leakage rate can actually push natural gas-fired electricity to the same level as coal power in terms of GHG emissions per kWh, all the more so when a short time horizon is used to compute the global warming potential. Figure 8 shows how high amounts of leakage along the extraction and distribution process may influence the lifecycle GHG of fossil-fuel technologies.

Regarding this life cycle assessment, **leakage values have been updated in the latest version of ecoinvent** for European natural gas supply. Among other things, a methane leakage rate of 0.5% is assumed for extraction in Russia, of 0.28% for transmission from Russia, and of 0.019% for transmission in Europe [46]. This study therefore updates the THEMIS inventories [47] at least for the UNECE regions [48]. Potential leakage *downstream* the CCS-equipped plants is not taken into account, neither from transportation of the captured CO_2 nor for its permanent storage. This latter assumption can be argued, as further research is required to guarantee the proper monitoring of CO_2 in deep geological formations, lest GHG emissions from seepage could increase to gigaton-levels over the course of this century [49, 50].

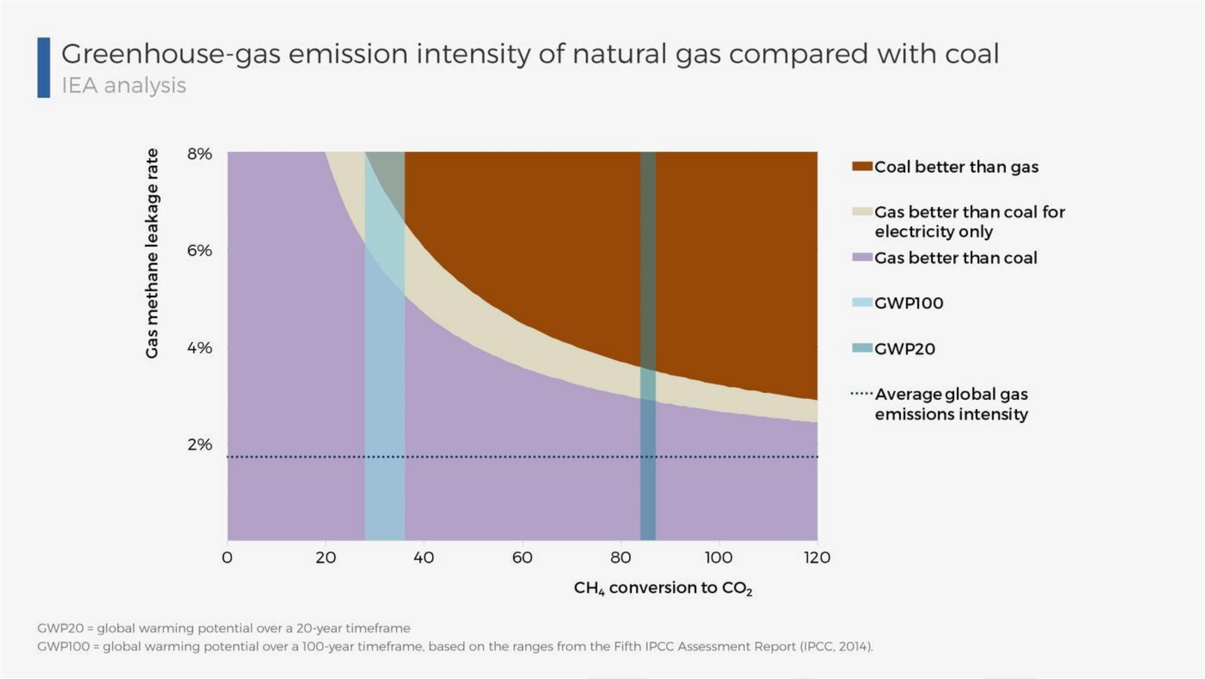

Figure 8. Coal- and gas-fired electricity GHG emissions depending on methane leakage rate in the natural gas supply chain and the time horizon chosen for the GWP calculation. At a 100-year time horizon (light blue), methane has a GWP of about 25–35 kg/kg CO_2 eq. depending on sources and assumptions, while its 20-year GWP is about 85-90 kg/kg CO_2 eq. (in dark blue), in which case a leakage rate of a few percents would be enough to make gas worse than coal except for electricity production (because of the relatively better efficiency of NGCC plants, beige area) or for all uses (per MJ, brown area). Source: Gould and McGlade [51].

Vinca, Emmerling [50] suggest that CO_2 storage may also lead to potential leakages. Leakage rates of 0.01% to 0.1% are tested on several energy scenarios, including scenarios with high CCS penetration, to show that leakage may affect climate targets (with cumulative emissions up to 25 Gt CO_2 eq. until 2100) if not properly addressed with appropriate monitoring of wells. Most pessimistic estimates lead to emissions of 10% of total CO_2 stored over a period of 30 years, authors conclude that there is too little hindsight to conclude on longer time periods [50].

3.2.2 Life cycle inventory

Data for the modelling of fossil-fuelled plants have been collected from Hertwich, de Larderel [5]. Inventories are all originally built from technical reports published by the National Energy Technology Laboratory (NETL) of the United States. Main parameters are shown in Table 6. Only combined cycle power plants are modelled, turbine designs (for peaking plants) are excluded from the scope of this study.

Table 6. Natural gas power plant characteristics, from [5], original source: [37].

Parameter	NGCC without CCS	NGCC with CCS
Nameplate capacity (MW)	497	474
Capacity factor	85%	
Net efficiency	50.2%	42.8%
CO_2 capture efficiency	90%	
Flue gas desulphurisation efficiency	Low-sulphur fuel	
Selective catalytic reduction efficiency	90%	

3.2.3 Environmental impact assessment

Regarding natural gas-fired power plants, a pattern similar to coal power plants emerges: direct combustion is the main contributor to water consumption and greenhouse gas emissions, whereas the natural gas production (the whole upstream chain from extraction to delivery at plant) is principally responsible for resource use, land use, ionising radiation and eutrophication (Figure 9). Overall values are however significantly lower than for coal – especially regarding eutrophication, land use (high values for coal because of mining activities, both open pit and underground) and water use (plant operation).

Adding carbon capture to an existing plant will increase feedstock requirements, for coal as for gas alike, this “energy penalty” explains the increase in non-GHG impacts, while GHG reductions achieved range from -64% for hard coal, to -70% for natural gas (Figure 10).

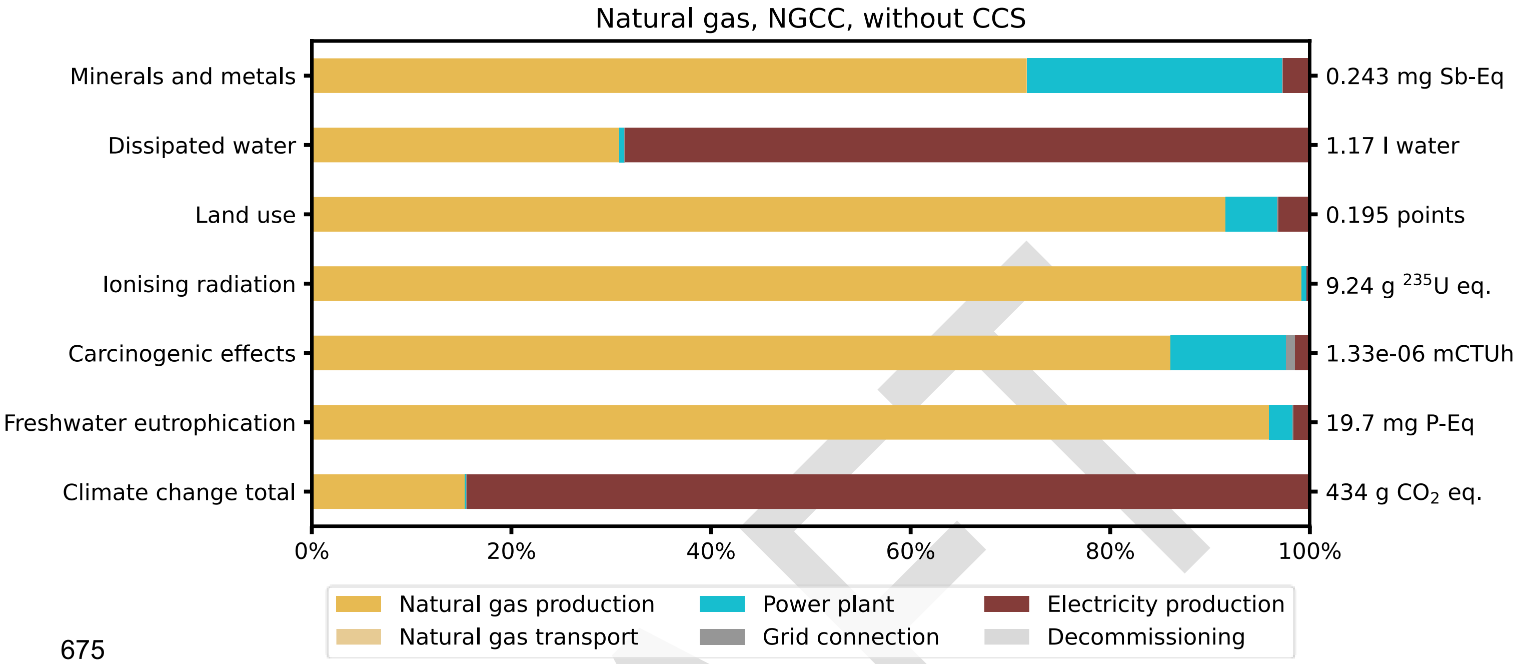

Figure 9. Life cycle impacts from 1 kWh of natural gas power production, NGCC without carbon dioxide capture and storage, Europe, 2020.

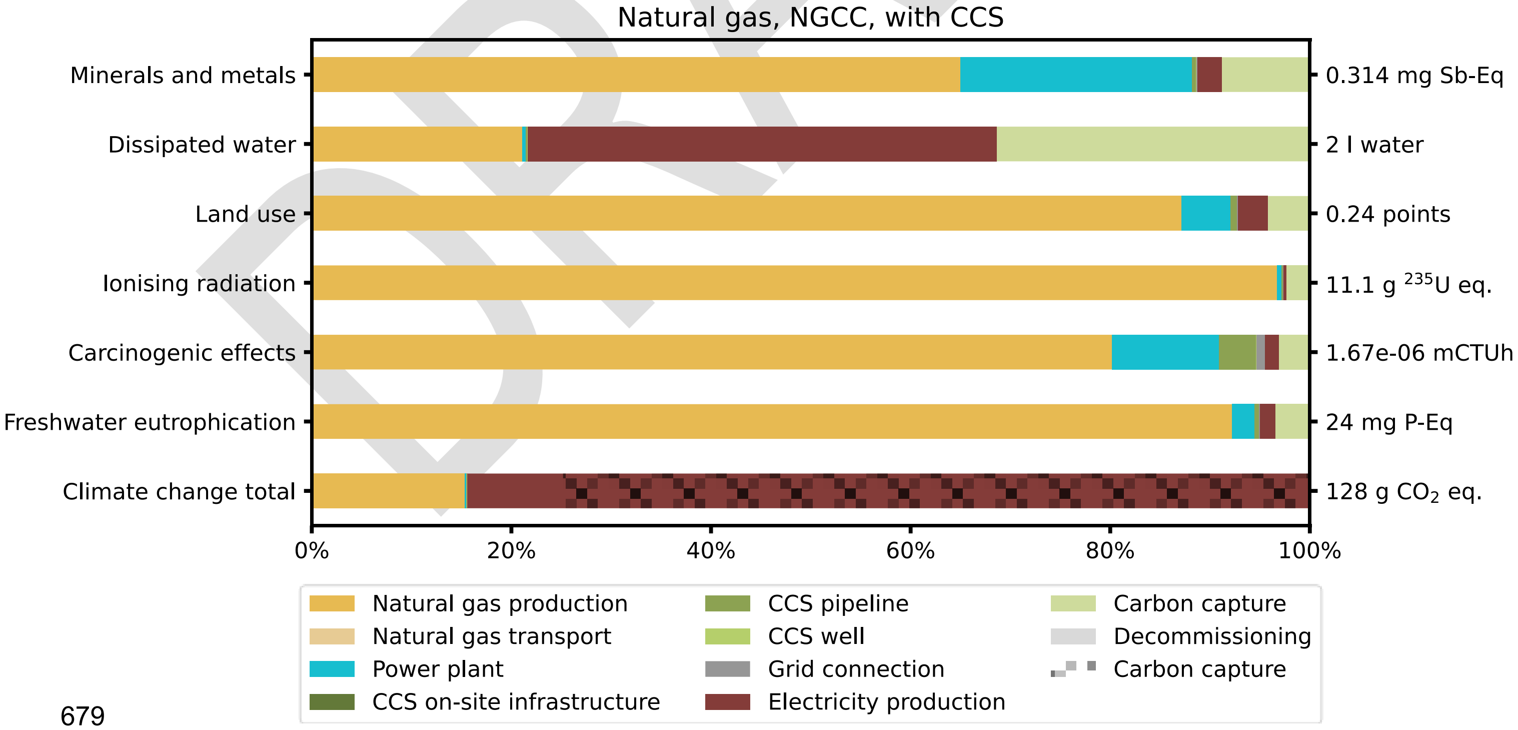

Figure 10. Life cycle impacts from 1 kWh of natural gas power production, NGCC with CCS , Europe, 2020. Carbon dioxide capture and storage processes are shown in red when positive, in hatched lines when negative.

3.3 Wind power

With a grand total of 622 GW installed globally in 2019, onshore wind is the second largest source of renewable electricity after hydropower. Onshore wind power dominates the wind market (594 GW), while offshore wind power represented 28 GW of capacity globally [52].

3.3.1 Technology description

In terms of electricity production, load factors reached 25% and 33% (in 2018) for installed onshore and offshore wind turbines respectively. Global wind power electricity generation was estimated at 1590 TWh in 2020 [30]. Load factors of installed wind power vary significantly across the globe have been adapted to follow the latest estimates per region, Table 7 shows the regional variations that have been assumed in this study.

At the device scale, wind turbines have become increasingly efficient due to their larger size. This increase in turbine size has also led to a reduced environmental impact per kWh of production, as shown in [53] and in Figure 11. The two main factors leading to a decreased environmental impact per unit of electricity generated are scale and technology learning. The former factor, scale, relates to the pure size of the turbine, in particular its height and diameter. Height matters as more wind energy can be captured at higher wind shear factors and hub heights [54]. Diameter relates the area swept by the blade and the amount of kinetic energy harnessed by the turbine. The latter factor, learning, includes experience acquired over time (proportional to cumulated installed capacity) leading to an increased design and manufacturing efficiency, and improvements to the technology itself such as the use of more efficient materials for the blades. Overall, these two factors have been estimated to reducing the lifecycle environmental impacts of wind power by 14% for every doubling in capacity [53].

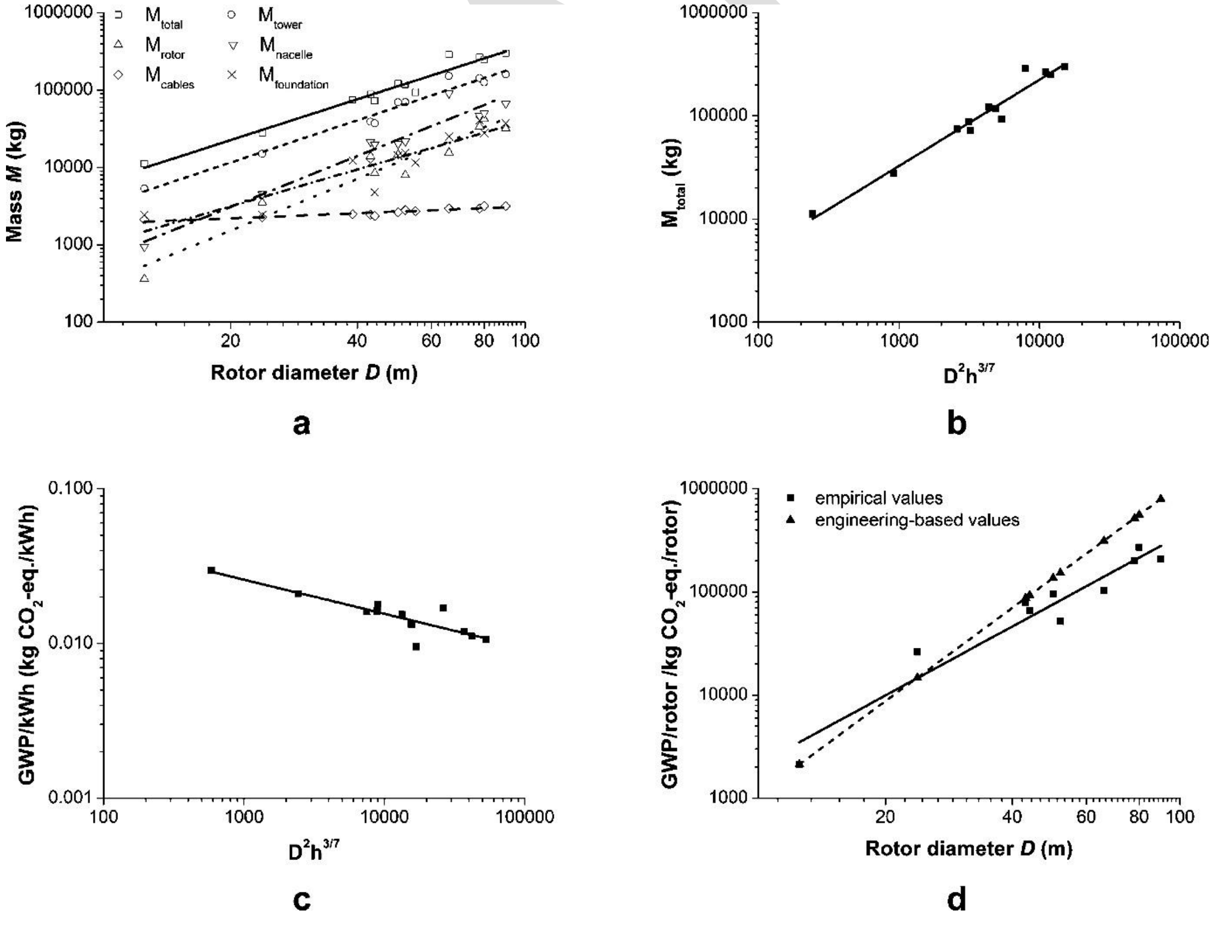

Figure 11. Correlation plots between wind turbines' characteristics, a: mass vs. rotor diameter, b: mass vs. a function of diameter and height, c: lifecycle GHG emissions per kWh vs. a function of diameter and height, d: lifecycle GHG emissions per rotor vs. rotor diameter. Source: Caduff, Huijbregts [53].

3.3.2 Life cycle inventory

Wind power life cycle data has been extracted from various sources, using the same general dataset [55-57]. These sources all rely on a detailed system description of wind power turbines, both onshore and offshore. The latter includes a representative model of offshore maintenance, recognized to be a significant contributor to life cycle impacts. Basic assumptions in the original data have been reused, namely regarding capacity and lifetime, respectively **2.5 MW and 20 years for the onshore** wind turbine, and **5 MW and 25 years for the offshore** wind turbine.

*Table 7. Capacity factors assumed for wind power in each region. *Data not available, global average used. **Data not available, China average used. Source: [52].*

Region	Capacity factor, onshore	Capacity factor, offshore
CAZ	29.2%	30.5%*
CHA	22.7%	22.7%
EUR	22.8%	36.2%
IND	17.8%	30.5%*
JPN	25.0%	30.0%
LAM	36.1%	30.5%*
MEA	29.6 %	30.5%*
NEU	26.2%	31.4%
OAS	22.7%	22.7%**
REF	26.2%	30.5%*
SSA	29.2%	30.5%*
USA	33.4%	40.0%

The “Wind LCA Harmonization” project [58], relying on 49 pre-2012 LCA publications, providing 126 estimates of lifecycle GHG emissions of wind power, showed a full range of 1.7–81 g CO_2 eq./kWh, with a median of 12 g CO_2 eq./kWh. The meta-analysis showed that key parameters for the environmental impact assessment of wind power are lifetime, capacity factor, system boundaries, turbine size, and whether the turbine is onshore or offshore. The IPCC AR5 values indicate similar ranges, with medians and interquartile ranges of 11 [7.0–56] and 12 [8.0–35] g CO_2 eq./kWh for onshore and offshore wind turbines respectively. Relatively high amounts of bulk material are required, specifically steel and concrete needed to deliver 1 kWh to the grid. Beyond GHG emissions and materials, broader LCA studies indicate that wind power offers a wide spectrum of co-benefits: little particulate matter emissions, low acidification, low eutrophication, toxic emissions or low land use.

On that latter aspect, defining the land use of a wind farm is ambiguous due to the sparse nature of a group of wind turbines. Denholm, Hand [59] suggest **the distinction between “total project area” and “direct impact area”**. The former includes all land associated with a wind farm as a whole, whereas the latter only considers the “disturbed land”, at a finer resolution, accounting for the potential use of the land for other purposes. **The “direct impact area” approach is used in this study**. Site selection for wind farms is driven by the following factors, among others: wind speed (most important) and density, distance to roads, power lines, and urban areas, slope, and current land occupation [60]. This suggests that land can be used for other purposes (e.g. agriculture) not requiring tall construction, which would be susceptible to obstruct wind.

Changes to original inventories

Regional load factors have been updated for the various regions, and electricity inputs linked with the REMIND region classification. Inputs from the IO database have not been replaced by process LCA inputs (but they were set to 0 in [5]).

3.3.3 Environmental impact assessment

While the tower and foundations contribute to most impact categories (50%–70%), the generator is notably responsible for half the “minerals and metals” impact category due to copper needs. Blades, made of glass fibre reinforced plastic, contribute only to climate change (16%), ionising radiation (7%)

and dissipated water (27%), due to the use of electricity for their production. Other activities, mainly maintenance, contribute to 12%–20% of all impacts. It is to be noted that other materials may be needed for other wind turbine designs, but are not accounted for in the life cycle inventories, this explains the absence of several processes/parts in the “minerals and metals” indicator, and is addressed in Box 2.

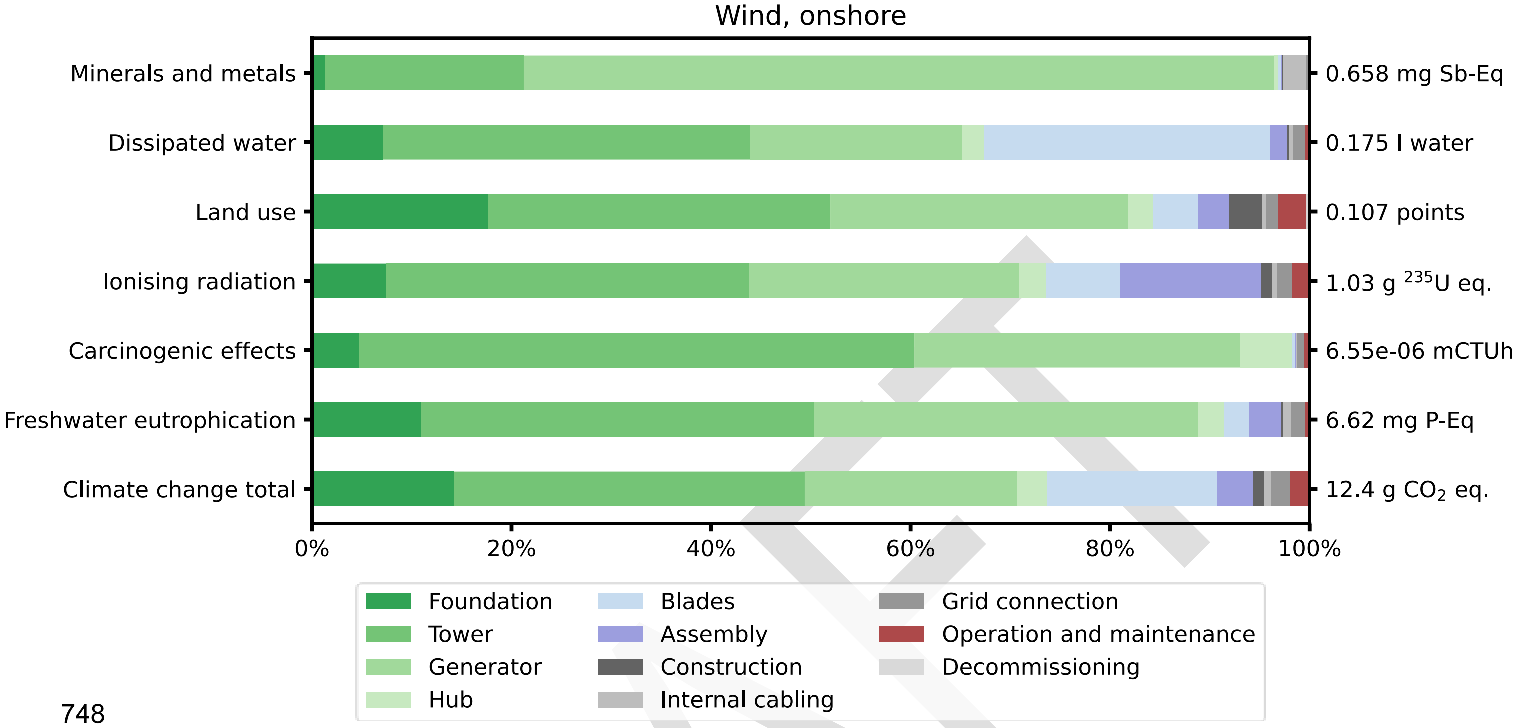

Figure 12. Life cycle impacts from 1 kWh of onshore wind power production, Europe, 2020.

The contribution of ship operations for construction of offshore wind turbines is a clear difference with onshore designs, as ships (under “Construction”) constitute roughly 20% (about 3 g CO_2 eq./kWh) of the lifecycle GHG emissions. Land use of offshore wind turbines is found to be equivalent to that of their onshore counterpart as very little direct land use is taken into account, combined with the absence of any water body use in the impact assessment method. Only indirect land use from mining the various elements is therefore represented here.

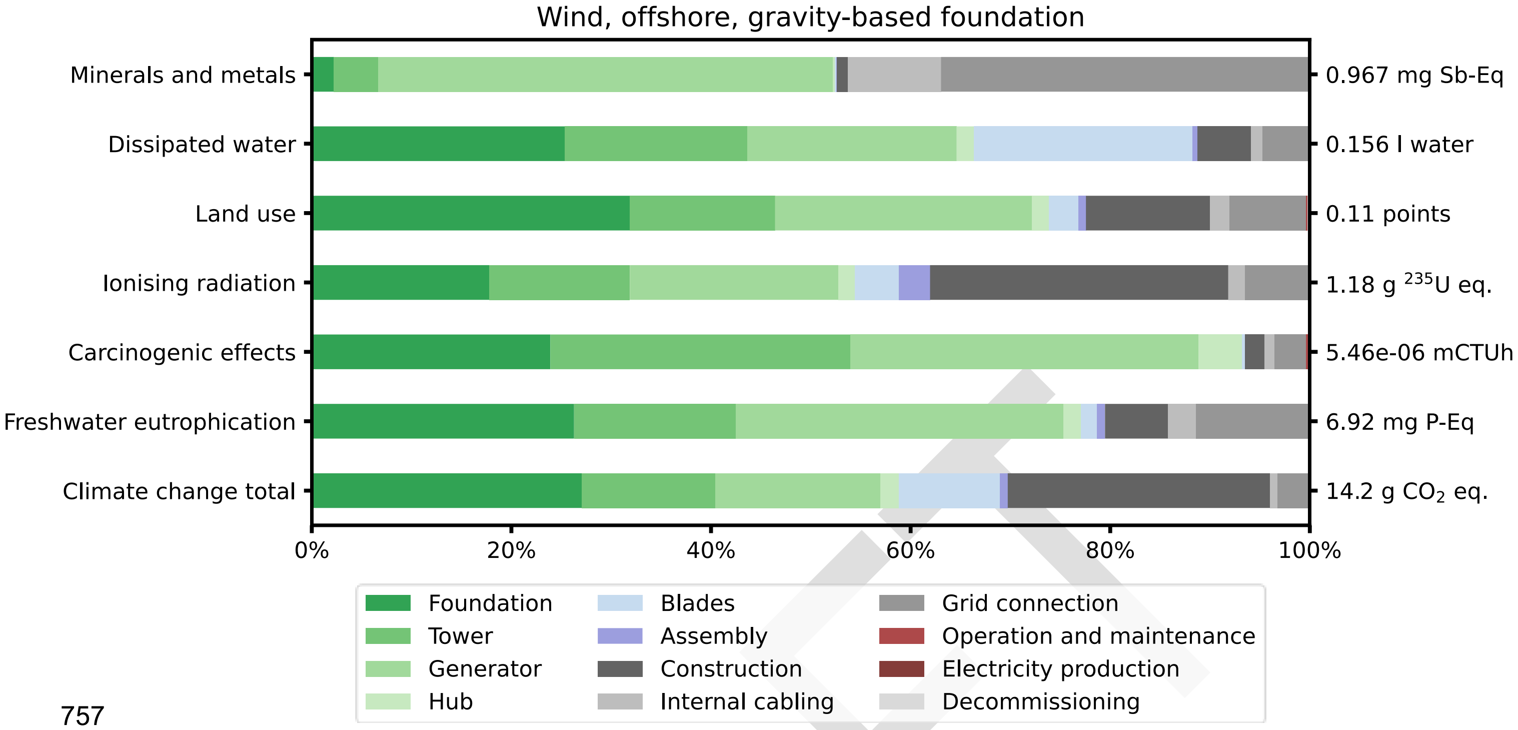

Figure 13. Life cycle impacts from 1 kWh of offshore wind power production, Europe, 2020.

It must be noted that neither aesthetic or noise aspects, or avian mortality issues are assessed in the scope of this LCA. The alteration of natural landscape could be seen as a subjective issue, noise effects on human health (through annoyance and sleep disturbance) have been studied, and shown to be correlated with potential damage [61, 62], and are potentially harmful to the health of workers [63]. On the other hand, the potential threats of wind power to birdlife are well-documented [64, 65], current research suggests that, while death rates may be relatively high in certain areas, they are highly variable (Barclay, Baerwald [66] reports a range of 0.00–9.33 birds per year per turbine, and 0.00–42.7 for bats). In context, these values are a small fraction of fatalities caused by other human activities (windows, domestic cats, …) [67]. Finally, low-tech solutions exist to reduce fatality rates substantially in sensible areas, such as painting one of the blades black to increase visibility; a case study shows that such a solution can decrease mortality by 70% [68].

Box 2. Rare earth and specialty metals, and their use in renewable technologies

The phrase “rare earth” has a strict definition: it qualifies one of the 17 chemical *rare-earth* elements (REEs) composed by scandium, yttrium, and the lanthanides. Despite their designation, these elements are not specifically “rare”, at least not as much as precious metals like platinum or gold can be. Their physical characteristics are of particular interest when it comes to improving the performance of electricity-using or -generating technologies, among other applications. For instance, praseodymium, neodymium, and dysprosium (three lanthanides) naturally hold strong magnetic properties, which are of interest in developing powerful yet compact direct-drive generators for wind turbines or synchronous motors in electric vehicles. Figure 14 shows an estimate of the amount of mineral and REEs embodied per MW of wind power. The designs modelled in the present study do not contain REEs.

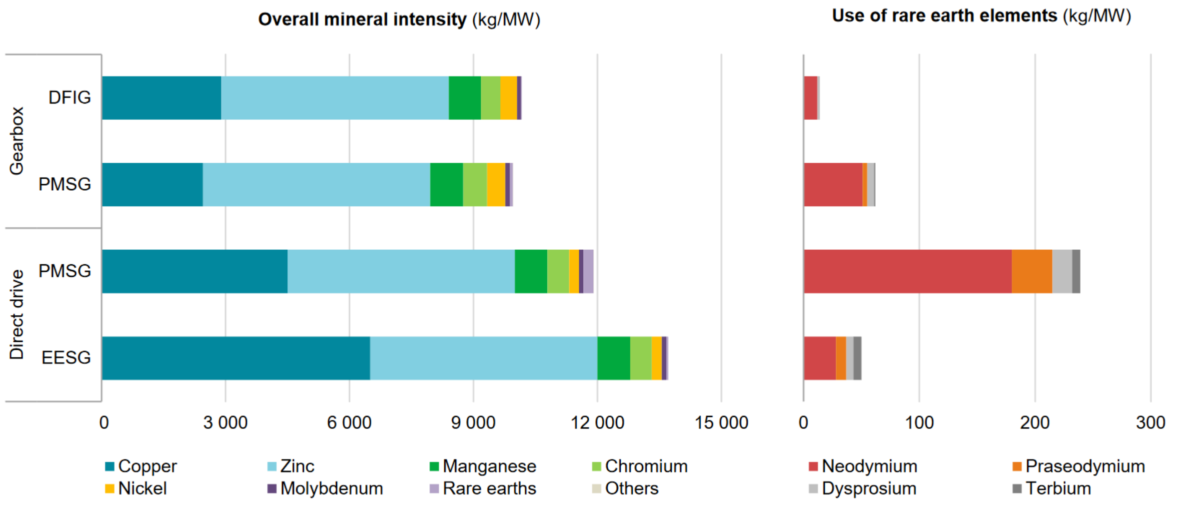

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Figure 14. Mineral intensity for wind power by turbine type. DFIG = double-fed induction generators; PMSG = permanent-magnet synchronous generator; EESG = electrically excited synchronous generator. The intensity numbers are based on the onshore installation environment. More copper is needed in offshore applications due to much longer cabling requirements. Source: International Energy Agency [24], Carrara, Alves Dias [69], Elia, Taylor [70]

The widescale use of REEs is relatively new, and justified concern has grown regarding the viability of a potentially booming demand while supply remains constrained, either because economic sites of extraction are concentrated in only a few countries or because their total reserves are simply unknown. The Herfindahl-Hirschmann Index (HHI) is an economic indicator used by the US Department of Justice to assess the competitiveness of a given market, the EU has also used this index in establishing its list of critical materials [71]. When applied to the current production of REEs and specialty metals, the HHI leads to a similar conclusion: lithium, REEs, and cobalt extraction are highly (geographically) concentrated sectors – from lowest to highest respectively (see Figure 15).

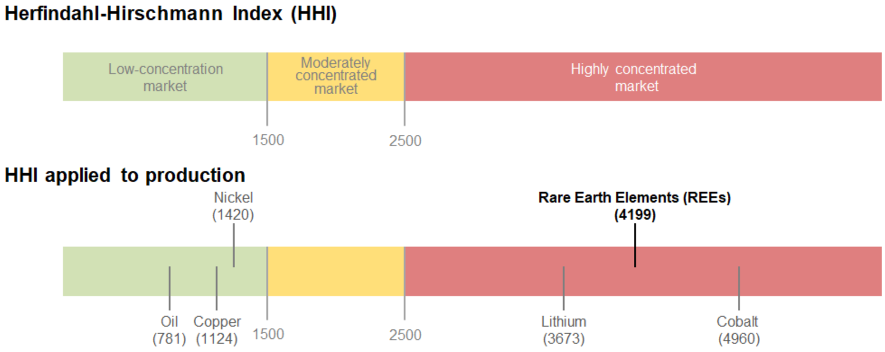


Figure 15. Herfindahl-Hirschmann Index (HHI), indicating the geographic concentration of a market. When applied to the critical material markets, it shows that lithium, REEs, and cobalt are (currently) overconcentrated.

The environmental and social impacts linked with REE extraction are a third concern often raised, as well as social and governance issues. Lèbre, Stringer [72] show that REEs, as well as lithium and cobalt, are the materials with the highest expected production increase, with an estimated median peak production of 2 to 5 times the current global production, indicating potential supply chain pressure. Of these materials, cobalt seems to be the one element whose production entails the highest ESG stress, namely on communities, land use, or social vulnerability. However, global demand in these materials is relatively low, and even dwarfed by the current production of more conventional materials such as copper and iron. All these findings are illustrated on Figure 16.

Unlike fossil fuels, REEs and specialty metals (lithium, cobalt) are however easily substitutable. For instance, gearboxes can replace direct drives in wind turbine generators, REE-free asynchronous motors can replace synchronous ones, and lithium ion-iron-phosphate chemistries can substitute cobalt-based batteries. The IEA is stressing that "reducing material intensity and encouraging material substitution via technology innovation can also play major roles in alleviating strains on supply, while also reducing costs" [24]. Reducing material intensity can be done through economies of scale: a 3.45-MW turbine contains about 15% less concrete and 50% less fibreglass, copper or aluminium than a 2-MW turbine [70].

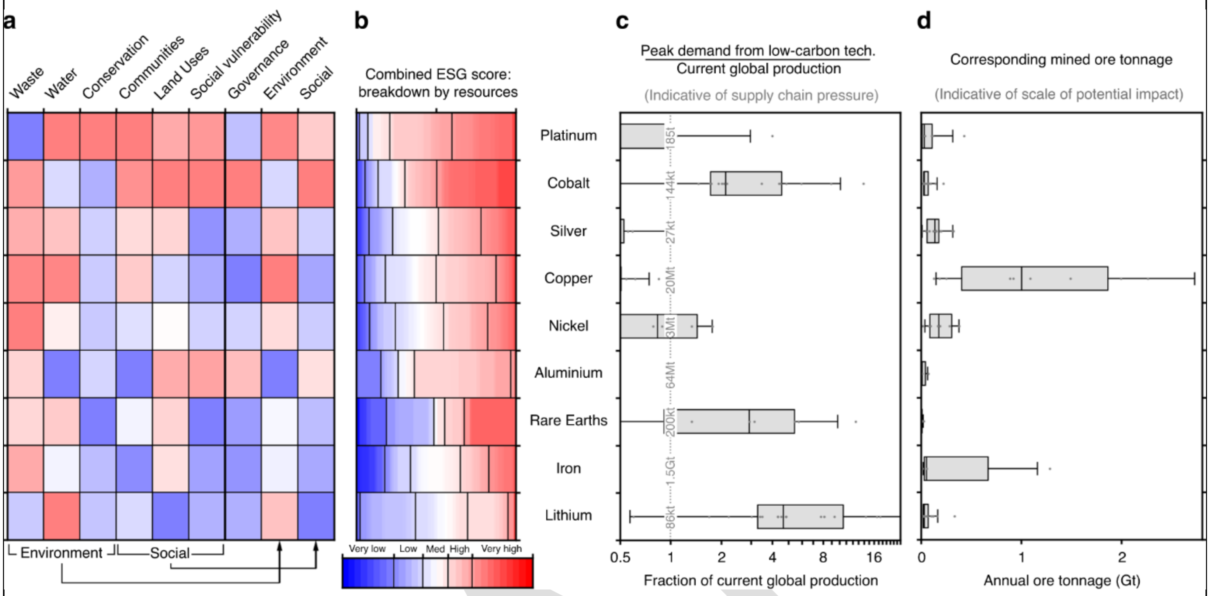


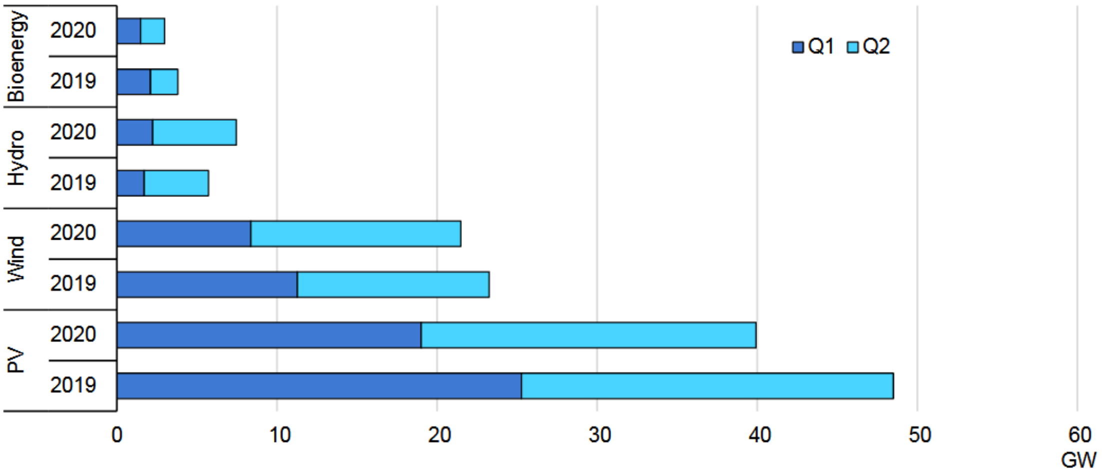
Figure 16. The various dimensions of criticality: a) ESG components, b) combined ESG score, c) multiple of current global production (refined) corresponding to peak demand, d) absolute ore tonnage value globally. Reading guide: the median estimate for peak cobalt demand is about twice the current production of 144 kt per year, 75% of estimates are below a factor 4. From Lèbre, Stringer [72].

Carrara, Alves Dias [69] show that the demand-to-global supply ratio exceeds 100% as soon as 2030 for REEs in wind turbines (as demand increases 14–15 times for Dy, Pr, Tb) and photovoltaic modules (demand increases 86 times for Ge, 40 times for Te) in the cases of high demand scenarios by 2050. In medium demand scenarios, demand increases around 3.5 times for REEs in wind turbines, 3–7 times for specific materials in PV.

3.4 Solar power: photovoltaics

The installation of solar photovoltaics has undergone a steep increase globally. A specificity of this technology is the decentralization potential of the PV infrastructure, whereby individuals or businesses can produce their own low-voltage electricity by installing panels at home or on their property. This installed capacity, of about 164 GW (2018), complements utility-scale installations, which represent 307 GW for the same year, and a grand total of 471 GW installed as of mid-2018 [73]. Net additions have recently surpassed 100 GW per year, which promotes solar PV as the fastest-growing renewable technology in terms of installed capacity.

Figure 1.2 Renewable capacity additions by technology, Q1 and Q2 of 2019 and 2020



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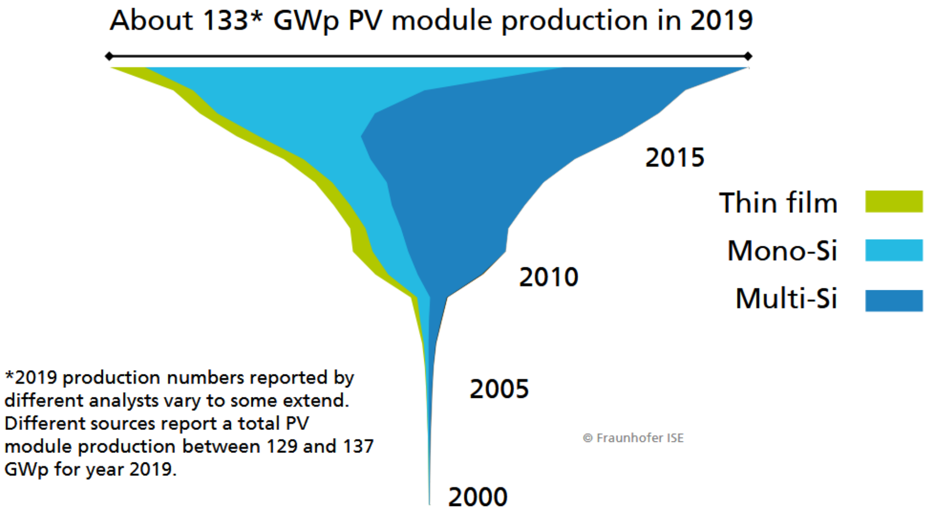
Note: Actual data collected from governments and industry associations cover Argentina, Australia, Brazil, Chile, China, France, Germany, India, Italy, Japan, Korea, the Netherlands, Poland, South Africa, Spain, Sweden, Chinese Taipei, Turkey, the United Kingdom and the United States. These sources represent 75% of total global capacity additions in 2019, with remaining additions estimated based on actual annual data and forecasts.

Figure 17. Renewable capacity additions by technology in 2019 and 2020. Source: International Energy Agency [74], page 18.

3.4.1 Technology description

Photovoltaic systems are diverse. Historically, crystalline silicon PV has been the technology of choice globally, with polycrystalline silicon cells representing the main market share of manufactured PV until 2015. Polycrystalline silicon panels are made of pieces of crystallized silicon melted together, which makes them relatively inexpensive to manufacture, but also less efficient, than their single-crystal counterpart, or *monocrystalline* silicon panels. The latter has tended to dominate the recent market.

The overwhelming majority of panels are therefore silicon-based, but since the early 2010s, the global production market has diversified with thin-film technologies becoming commercially available. Thin-film technologies have the advantage of being lighter than crystalline silicon PV, and flexible. The main thin-film options are amorphous silicon, cadmium-telluride (CdTe), and copper-indium-gallium-selenide (CIGS) modules. They offer an efficiency significantly lower than crystalline PV. Furthermore, thin-film technologies require more specialty materials than silicon-based modules, which may hamper their development depending on the supply of these metals (indium, tellurium, cadmium in particular may be of concern [75], this topic is explored in Box 2) Technologies assessed in this exercise are: polycrystalline-Si, CdTe, and CIGS; each in two variants, ground-mounted (utility-scale) and roof-mounted.



*2019 production numbers reported by different analysts vary to some extend. Different sources report a total PV module production between 129 and 137 GWp for year 2019.

Data: from 2000 to 2009: Navigant; from 2010: IHS Markit. Graph: PSE Projects GmbH 2020

Figure 18. Global photovoltaic module production by main technology. Source: Fraunhofer Institute for Solar Energy Systems and PSE Projects GmbH [76], page 20.

Box 3. Waste management from renewable infrastructure

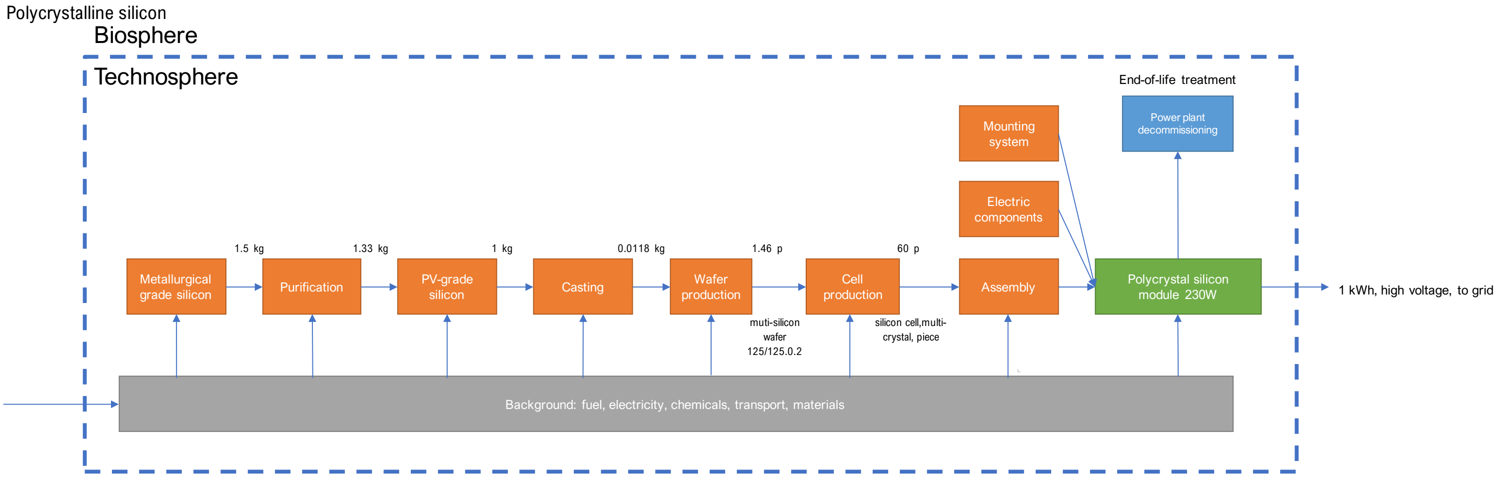
As the first renewable plants are reaching the end of their planned lifetimes, proper end-of- life management needs to be ensured to guarantee their overall sustainability. A high share of the installed infrastructure in wind and solar is bulk material, which (in regions with mature recycling infrastructure) can be readily recycled after disassembly and sorting: steel and concrete in wind turbines' components, as well as glass and metal parts of photovoltaic panels [77].

While somewhat challenging, photovoltaic panels can undergo recycling, as described in Ratner, Gomonov [77]. The modern protocol consists a first separation of the aluminium frame from the panels' glass, both of which can be readily introduced into conventional recycling schemes. The remaining materials are then heat-treated, allowing the silicon to be processed further. This is valid for polycrystalline panels – the recycling process for thin-film modules is more complicated as it involves both liquid and solid parts after a first crushing, semi-conductor materials are therefore more difficult to recover. For polycrystalline panels, recycling brings environmental benefits in terms of energy use and greenhouse gas emissions.

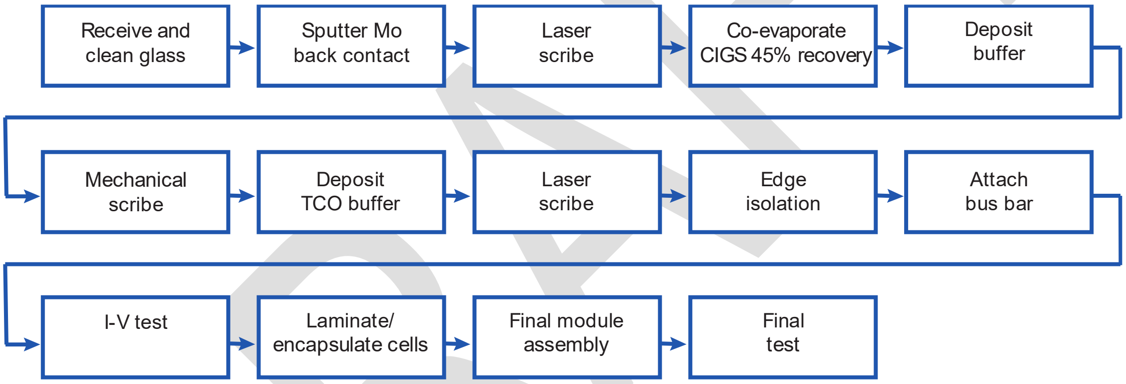
Wind turbines are readily recyclable, from foundation, to tower, gearbox and generator – except for their blades. Jensen and Skelton [78] describe the challenge regarding the incoming inflow of glass-fibre reinforced plastics from decommissioned wind turbines. They highlight that, despite commercially available recycling techniques, the bottleneck is the lack of practical experience in *reusing* secondary materials.

800 ### 3.4.2 Life cycle inventory

801 Data for the three photovoltaic types have been adapted from [5]. System boundaries are shown in
802 Figure 19, Figure 20, and Figure 21.

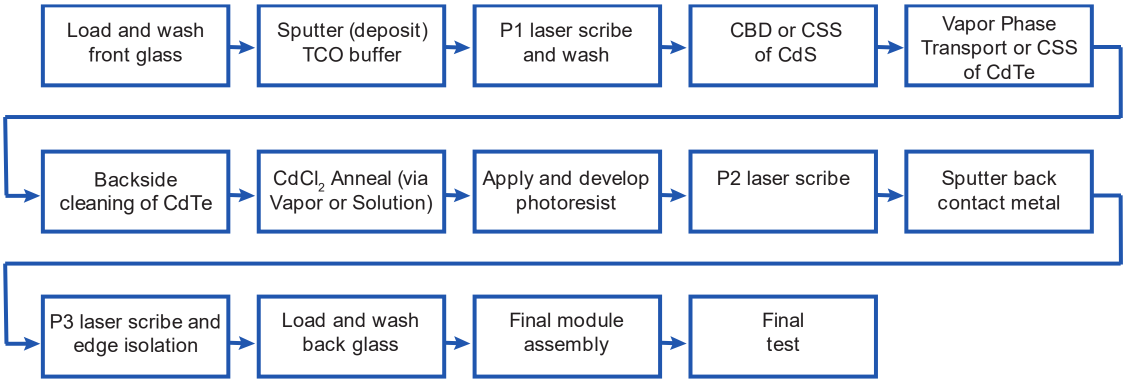


804 *Figure 19. System boundaries for the polycrystalline silicon systems (ground- and roof-mounted, for which only the*
805 *"Mounting system" differs).*



The various scribe stages make electrical connections between the layers.
806 Mo: molybdenum; TCO: transparent conducting oxide; I-V: current versus voltage

807 *Figure 20. CIGS manufacturing flow chart showing discrete process stages as described by NREL manufacturing*
808 *cost model. In the LCI, all processes are direct inputs (first-tier) to the 1.08 m² CIGS module. Source: [5]*



These stages are generalized to protect proprietary information. The process stages build the cell up from the glass; each layer is laid on top of the previous. TCO: transparent conducting oxide; CBD: chemical bath deposition; CSS: close space sublimation. P1, P2, P3 laser scribes make electrical connections between the layers

Figure 21. CdTe manufacturing flow chart showing discrete process stages as described by NREL manufacturing cost model. In the LCI, all processes are direct inputs (first-tier) to the 0.72 m^2 CdTe module. Source: [5]

The average load factors for photovoltaic technologies have been assumed for each region based on average normal irradiation at a reference location, as shown in Table 8.

Table 8. Average efficiencies assumed for photovoltaic technologies. Source: IRENA (2021), NREL (2021)

Region	Capacity factor	kWh/m^2/year	Reference location
CAZ	13.4%	2648	Australia (-32.594,137.856)
CHA	11.6%	2300	China (41.507, 108.588)
EUR	12.4%	2320	Spain (37.442,-6.25)
IND	12.9%	1637	India (27.601,72.224)
JPN	12.9%	1298	Japan(33.22,131.63)
LAM	16.9%	3438	Chile (-22.771,-69.479)
MEA	15.1%	2471	Morocco (30.218,-9.149)
NEU	10.6%	936	Denmark(57.05,9.9)
OAS	15.7%	1412	Thailand (14.334,99.709)
REF	9.58%	1459	Russia(47.21,45.54)
SSA	11.2%	2461	South Africa (31.631,38.874)
USA	18.0%	2817	USA (35.017,-117.333)

3.4.3 Environmental impact assessment

Under European conditions (region "EUR"), photovoltaic technologies show lifecycle GHG emissions of about 37 g CO_2 eq./kWh both for ground- and roof-mounted system – the global average is 52/53 (ground-/roof-mounted). About 40% of this climate change impact is due to the electricity consumption for solar-grade silicon refining. Lifetime assumptions aside, the two main parameters influencing the lifecycle GHG emissions of poly-Si panels are electricity for manufacturing and module efficiency/normal irradiation (see variation in section 4).

Silicon-based PV. As shown on Figure 22, about half of greenhouse gas emissions can be attributed to silicon manufacturing (from primary production to solar-grade refining), while the rest of emissions is split between the rest of the module, site preparation, and electrical equipment (inverters). No maintenance is accounted for in any system, assuming that no cleaning is necessary, which may be slightly optimistic depending on the region of operation. Eutrophication, dissipated water and ionising radiation show the same pattern as they are also linked to energy use for manufacturing. Land use however is mostly due direct occupation by the PV installation itself (60% for the ground-mounted panels) while the rest is linked with energy use and packaging (in containerboard) of the various module elements. Regarding mineral and metal scarcity the use of small amounts of silver in the silicon cells as well as the copper contained in inverters are responsible for most of the impact.

Roof-mounted PV panels (Figure 23) show roughly the same pattern, except for land use, where the impact is drastically reduced. All roof-mounted land use is indirect, embodied in the energy inputs needed for several manufacturing phases. Efficiency has been considered slightly lower, which explains a minor increase in all other impact categories.

Thin-film PV. Thin-film PV technologies, despite lower efficiencies, can offer lower lifecycle GHG emissions as they are completely silicon-free, and avoid the energy-intensive steps of silicon refining. Impacts from the balance of system (mounting frames, …) are more preponderant in thin-film than silicon-based technologies because of the relatively lower impacts of module manufacturing.

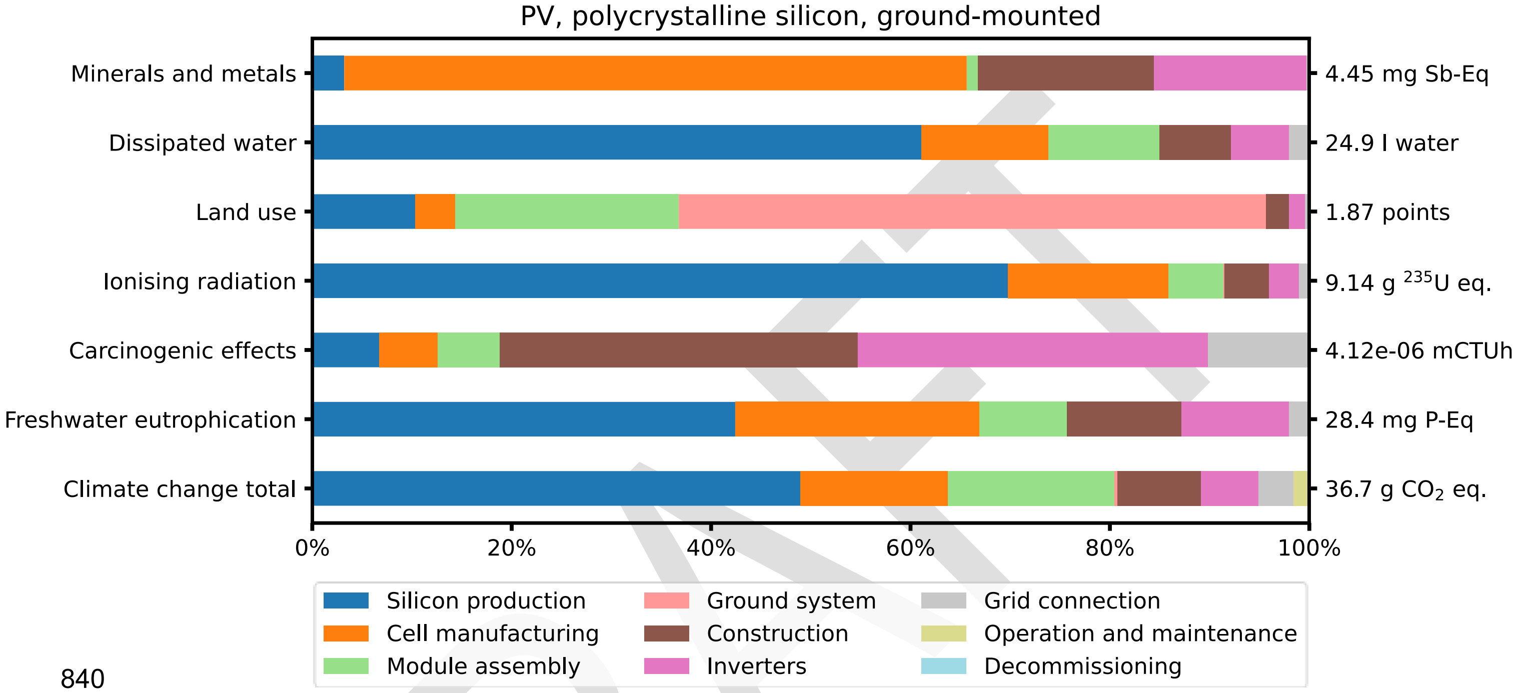
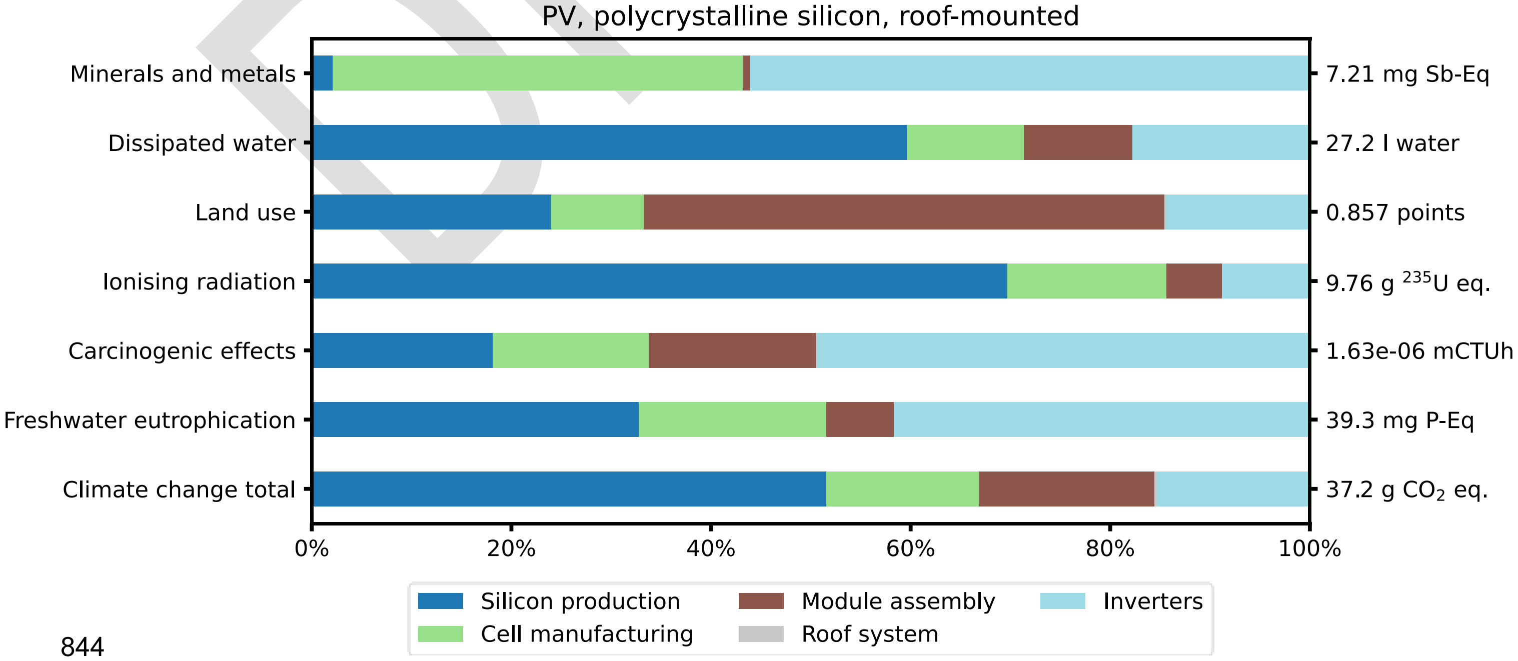


Figure 22. Life cycle impacts from 1 kWh of poly-Si, ground-mounted, photovoltaic power production, Europe, 2020.



845 *Figure 23. Life cycle impacts from 1 kWh of poly-Si, roof-mounted, photovoltaic power production, Europe, 2020.*

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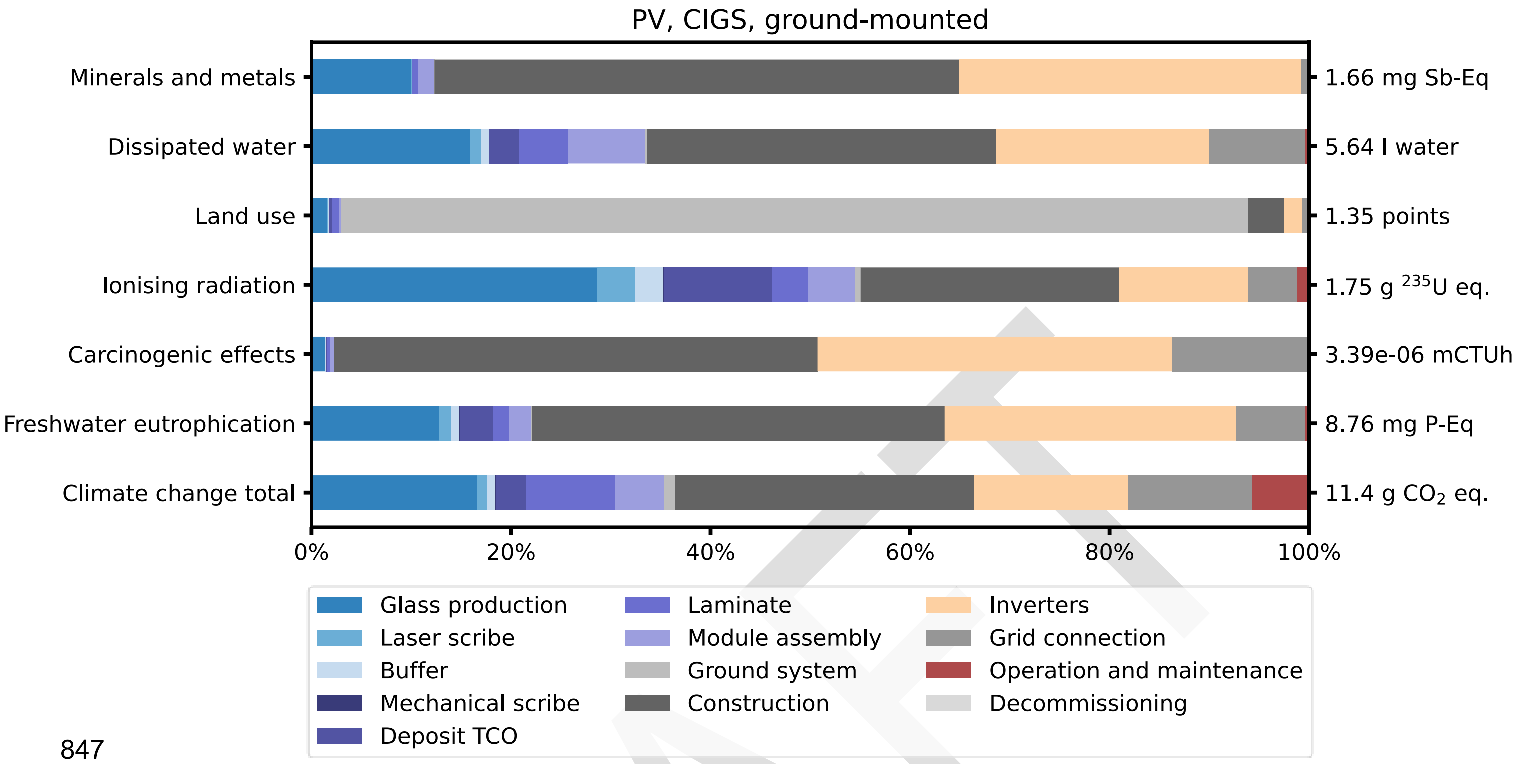

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848 *Figure 24. Life cycle impacts from 1 kWh of CIGS, ground-mounted, photovoltaic power production, Europe, 2020.*

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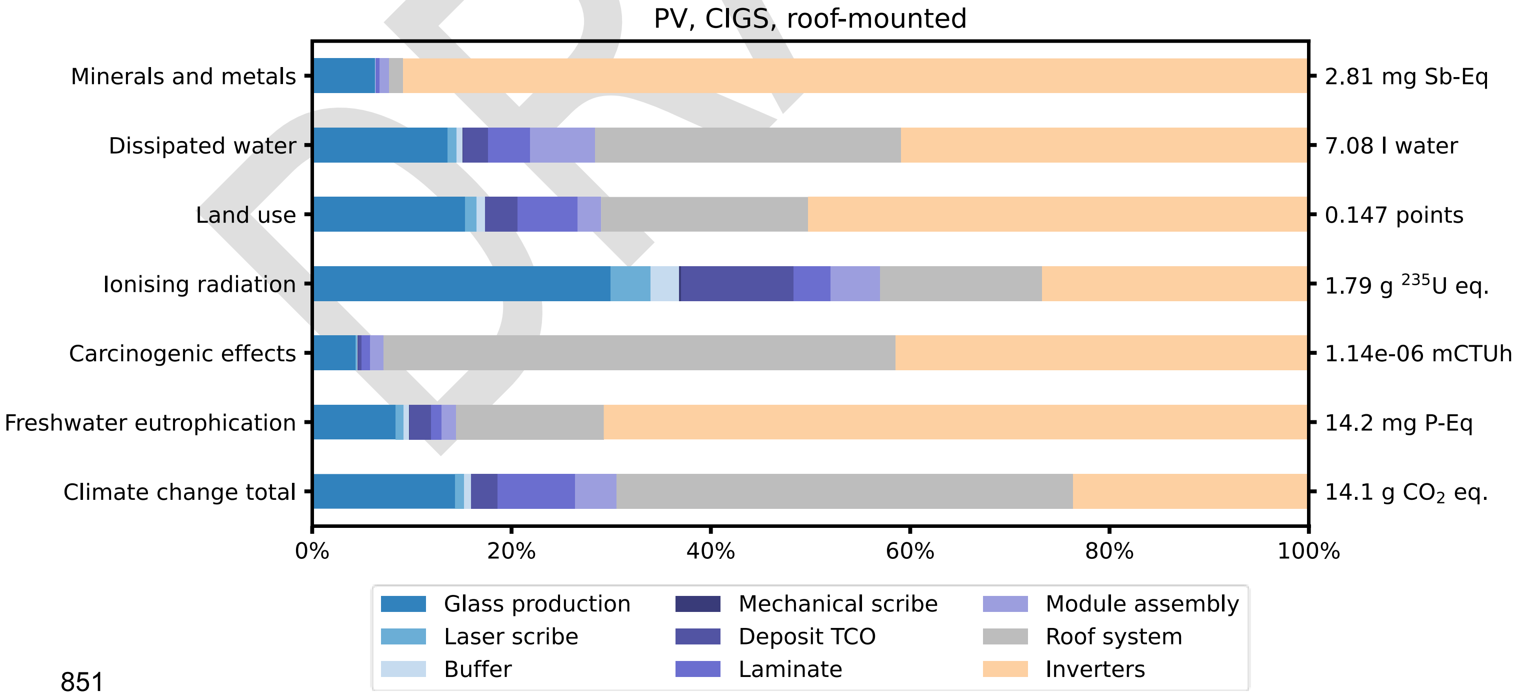

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852 *Figure 25. Life cycle impacts from 1 kWh of CIGS, roof-mounted, photovoltaic power production, Europe, 2020.*

853

854

Box 4. Electricity storage

Grid-scale energy storage is increasingly recognised as crucial to ensure a high degree of renewable electricity capacity on a given network [79]. Numerous options exist to store electricity at various scales of capacity and power, as represented on Figure 26. Larger scale solutions (> 10 MWh) include pumped hydro storage (PHS), compressed air energy storage (CAES), flywheels, and batteries.

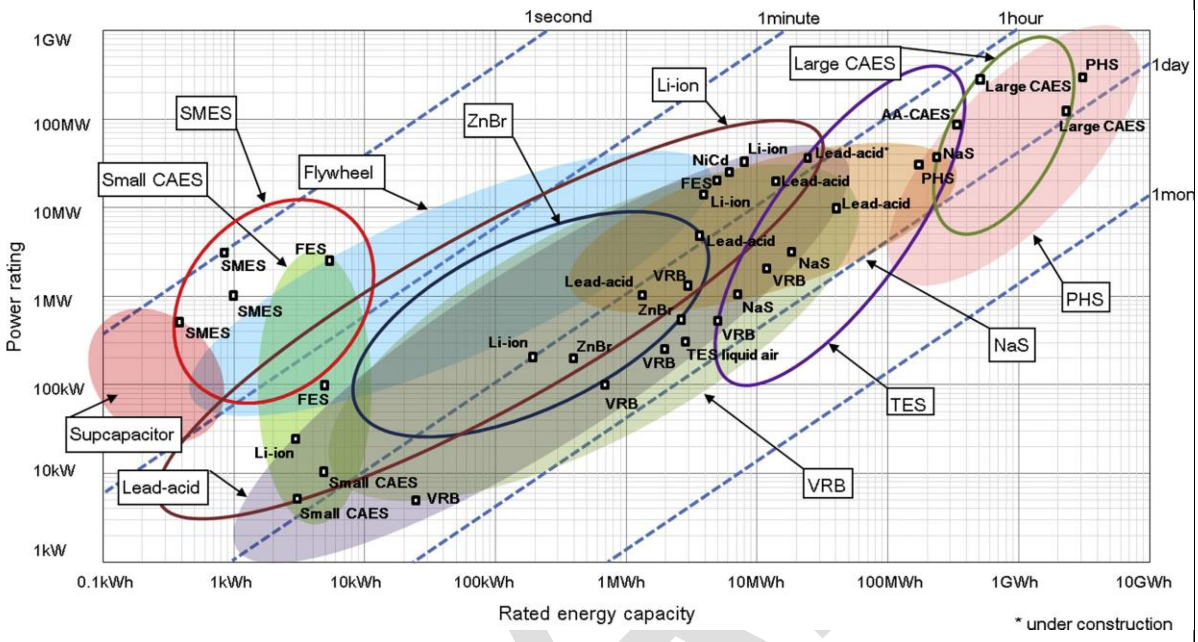

Figure 26. Electricity storage options, ranked by power rating (in MW) and energy capacity (in MWh). Isochrones are drawn to indicate the typical storage time intervals (MWh/MW) adequate to each solution. Adapted from Luo, Wang [80], under Creative Commons licence. PHS = pumped hydro storage, TES = thermal energy storage, VRB = vanadium redox flow battery, SMES = superconducting magnetic energy storage, CAES = compressed air energy storage.

Hottenroth, Peters [81] provide a comparative LCA of utility-scale storage solutions, namely PHS and battery, for the German electricity grid, assuming 2600 GWh of electricity provision per year over 80 years. We present their results in Figure 27, per kWh. For the whole German grid, impacts could range from an additional 30.2 (hydro) 36.3 Mt CO_2 eq. (battery) over 80 years, for comparison, the German electricity sector emitted 249.7 Mt CO_2 eq. directly in 2019[4]. CAES is another viable storage option for reducing intermittency. In particular, two designs exist: conventional CAES stores air to reduce the need for input compression in a fossil gas turbine (i.e. it should be compared to a NGCC or conventional gas turbine); whereas adiabatic CAES (ACAES) does not require any fossil fuel [82]. Conventionally, salt caverns are used for storage in CAES designs – no leakage is modelled.

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[4] Statistics available at https://www.umweltbundesamt.de/daten/umweltindikatoren/indikator-emission-von-treibhausgasen

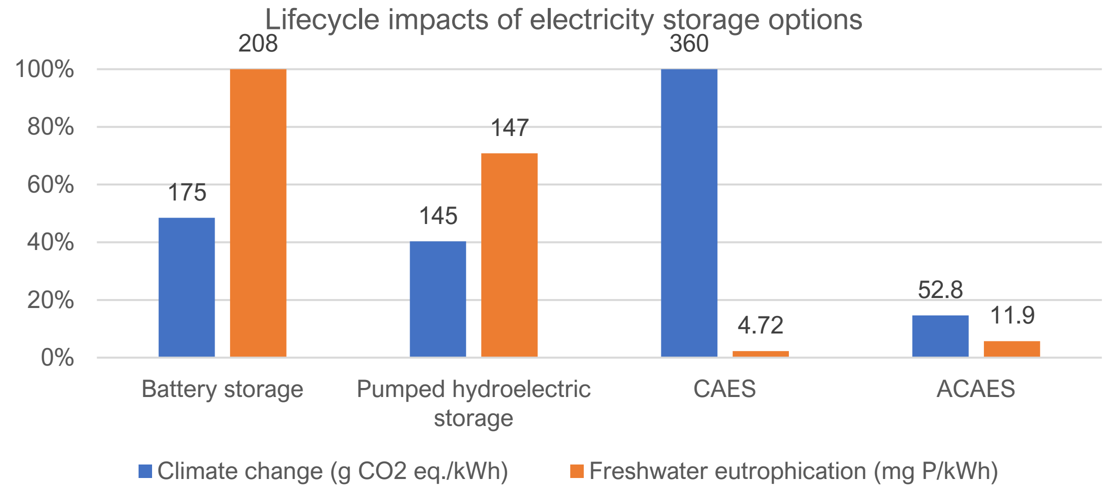


Figure 27. Comparison of lifecycle impacts of select electricity storage options. Source: [81] (for battery and PHS), [82] (for CAES).

The addition of storage capacity and grid reinforcement therefore increases the per-kWh impact of *non-dispatchable* electricity, but this surplus depends highly on local conditions such as the share of intermittent power, load, mix of storage technologies, or interconnection with other grids (exports can absorb a production surplus, imports can palliate limited storage). Raugei, Leccisi [83] find that adding 4 hours of 60-MW storage to a conventional 100-MW PV system would increase GHG emissions from 62 to 71–90 g CO_2 eq./kWh (at the lower end of 1000 kWh/m^2/year of irradiation) or from 27 to 31–39 g CO_2 eq./kWh, depending on battery chemistry. As for the grid extensions necessary to accommodate the variability of intermittent renewable electricity, most of their impacts are land use-related [84].

The potential role of hydrogen production for grid storage

Regarding longer-term storage, such as inter-seasonal capability, a main candidate is hydrogen production from surplus power generation. A study of 35 years of hourly data on the German electricity production shows that storage requirements must be scaled based on periods extending to 9–12 weeks – which translates to more than 50 TWh of hydrogen produced annually [85]. The study is not peer-reviewed and does not provide any data on environmental impacts. Literature shows that the more ambitious the renewable share target, the increasingly more difficult it is to ensure flexibility and grid stability [86]. For example, Ziegler, Mueller [87] find that meeting demand with a dispatchable technology only 5% of the time would *halve* the electricity generation costs compared with a 100% renewable system.

Hydrogen is not a primary energy source, but an energy carrier (much like electricity), which requires conversion from other sources (fossil fuels, or electricity produced from fossils, nuclear or renewables. Hydrogen for long-term grid storage could be produced from surplus production of intermittent sources when load is low, via water electrolysis. Despite significant conversion losses (30 to 40%), electrolysis from renewable electricity sources would confer low-carbon characteristics to the H_2 produced. Converted back to electricity via fuel cells (with losses, again), such a solution could therefore ensure load-following on an annual timeframe, with minimal CO_2 emissions.

Figure 28 shows the ranges of lifecycle GHG emissions for various hydrogen production technologies. For electrolysis, these emissions depend almost entirely on the electricity used as energy input. For comparison, 1 kg H_2 contains about 33 kWh of embodied energy (from about 50 kWh consumed by the electrolysis process), which could deliver about 15 kWh back to the grid, as a PEM cell's average efficiency is about 47% (high-performing cells could reach 70% [88]). The so-called *round-trip efficiency* is about 30%. Roughly said, producing and using H_2 to store electricity at grid level would triple the carbon content of the electricity originally used for production, once losses are accounted for.

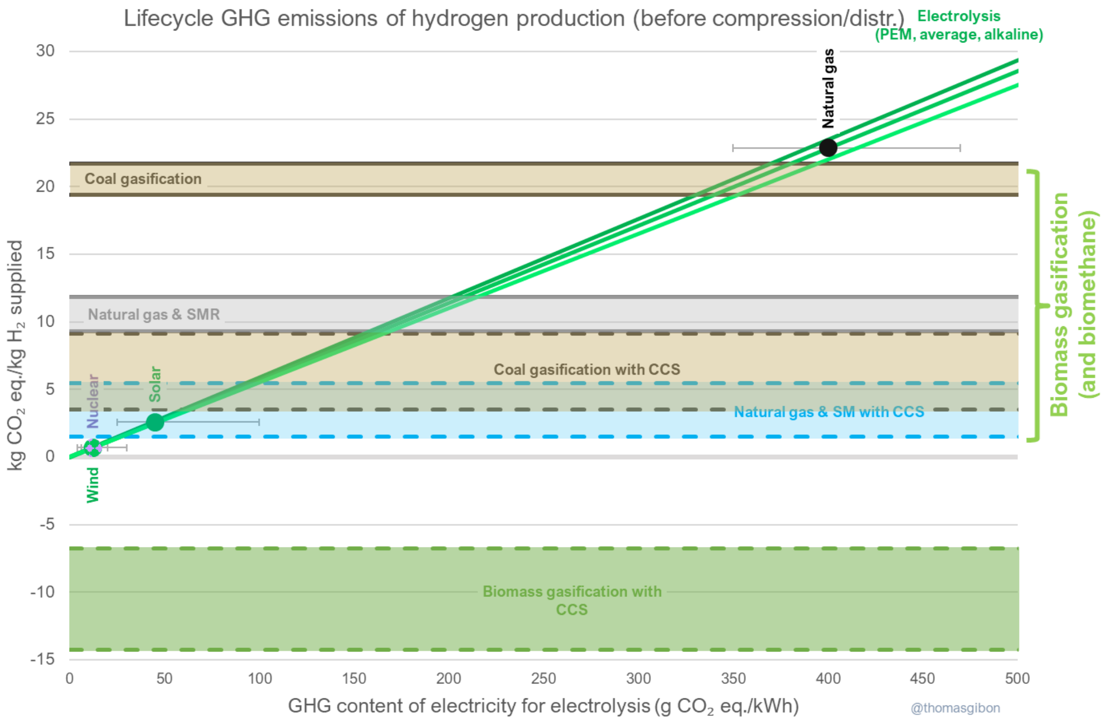


Figure 28. Comparison of hydrogen production methods, depending on the GHG content of the electricity used for electrolysis. Sources: [89-92]

3.5 Solar power: concentrated solar

Compared to photovoltaics, solar thermal, or concentrated solar power (CSP) technologies are a rather niche market, as 6.5 GW of installed capacity was in operation as of 2020 [93]. The common principle to all plants is the harnessing of solar energy, transferred to a heat transfer fluid.

3.5.1 Technology description

CSP encompasses a wide range of designs, generally grouped into "dish", "trough", and "tower" design. The two formers consist in independent systems of mirrors and heat transfer fluid circuits then centralized to run a steam turbine, while the latter relies on a central tower concentrating the light of a vast array of mirrors to a collector. in this current report we focus on the trough and tower designs, as they represent most of the CSP plants in construction today.

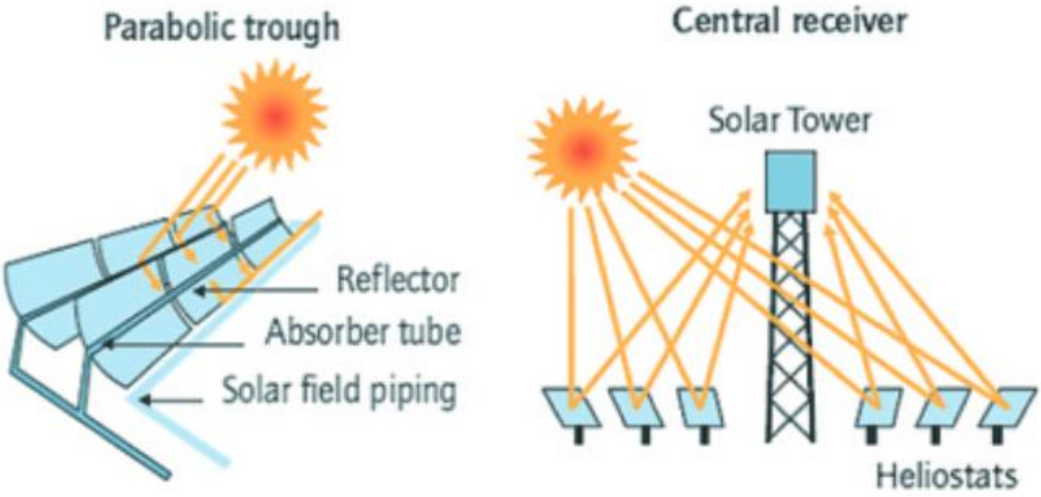


Figure 29. CSP designs: parabolic trough and central tower (receiver). Source: [94]

3.5.2 Life cycle inventory

LCI data is adapted from [5], in turn based on [95] and [96]. Updates include the relinking with the latest ecoinvent database, regionalisation of electricity inputs, and load factors. The trough design has a 103 MW nameplate capacity, and load factors depending on the location (Table 9); while the central tower design is sized to 106 MW of nameplate capacity and is also subject to varying load factors. Both power plants are equipped with thermal energy storage, and are assumed to be operationally viable for 30 years.

The load factor of a CSP technology depends strongly on its location, design, as well as their energy storage capacity (if any). Technically, plant size and year of construction also affect efficiency, but these factors have not been taken into account here. Therefore, the load factors of the technologies modelled have been computed independently – the central tower design offers a higher factor than the parabolic trough due to its 6-hour energy storage facility. Values retained for the model are shown in Table 9.

Table 9. Load factors assumed for the two CSP designs. Sources: [94, 97-99].

Region	Capacity factor, central tower	Capacity factor, parabolic trough	Reference location
CAZ	55.0%	38.9%	Australia (-32.594,137.856)
CHA	49.3%	33.9%	China (41.507, 108.588)
EUR	49.2%	36.9%	Spain (37.442,-6.25)
IND	36.2%	29.3%	India (27.601,72.224)
JPN	14.4%	20.6%	Japan (33.22,131.63)
LAM	70.9%	55.8%	Chile (27.601,72.224)
MEA	55.8%	42.8%	Morocco (30.218,-9.149)
NEU	14.4%	12.3%	Denmark (57.05,9.9)
OAS	29.3%	28.2%	Thailand (14.334,99.709)
REF	29.1%	23.7%	Russia (47.21,45.54)
SSA	55.2%	42.0%	South Africa (31.631,38.874)
USA	60.4%	37.5%	USA (35.017,-117.333)

3.5.3 Environmental impact assessment

For the CSP trough system, the preparation of the solar field, the thermal energy storage, and operation and maintenance contribute to about 75%–80% non-climate impacts (Figure 30). In particular, the solar field itself contributes to the majority (80%) of lifecycle land use. Construction and assembly of the infrastructure, on the other hand, is a minor contributor to non-climate impacts (5–15%) but the first

GHG-emitting process (30%, or 13 g CO_2 eq./kWh, in Europe), due to the use of energy inputs (electricity and diesel) for the fabrication and assembly steps. All in all, the generation of 1 kWh is found to generate about 42 g of CO_2 eq. over the system's life cycle in a European context. Regional variation can be observed in section 4.1.1.

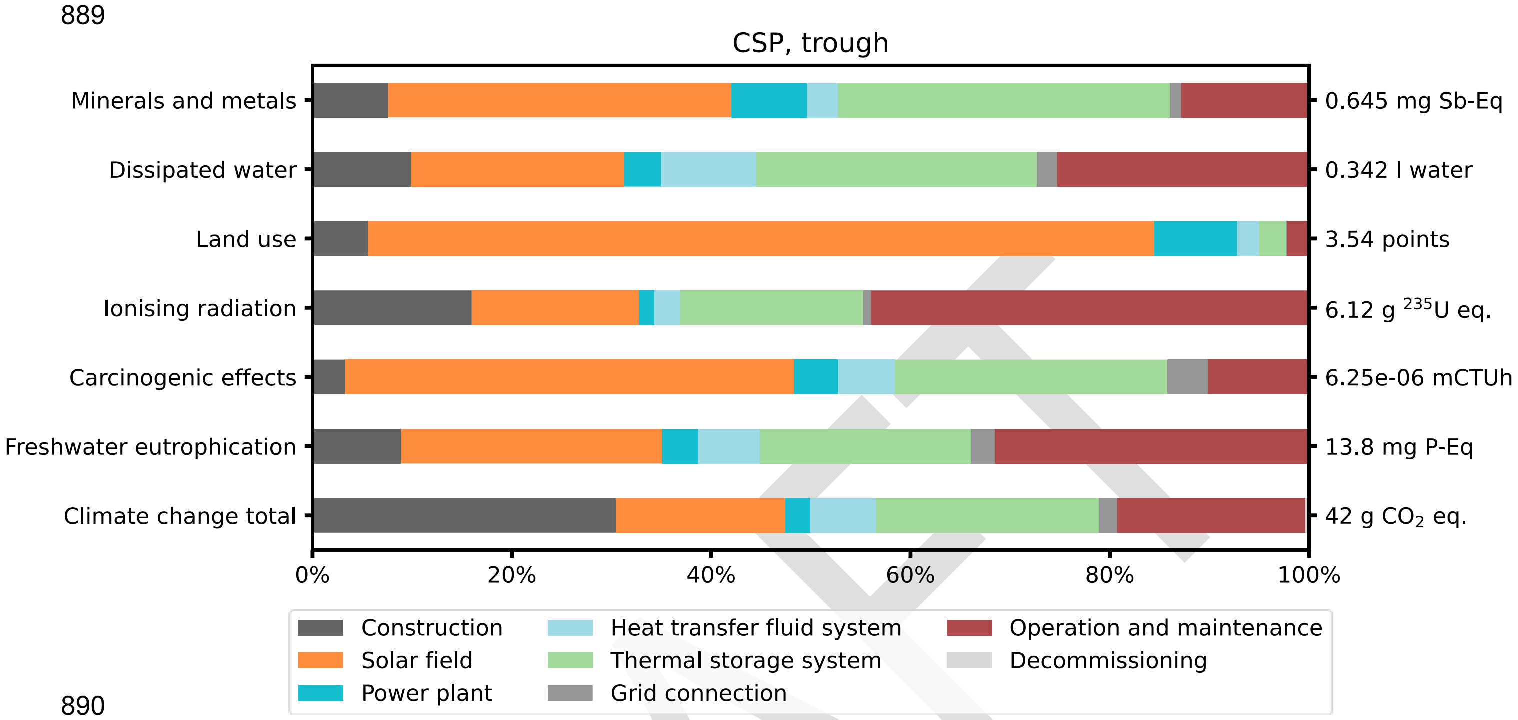


Figure 30. Life cycle impacts from 1 kWh of parabolic trough concentrated solar power production, Europe, 2020.

The central tower design is found to emit significantly less GHG on a life cycle basis, with about 22 g CO_2 eq./kWh, due to a higher estimated efficiency – thus displaying half the emissions of a trough design. Land use is dominated by direct impacts, with the site occupation itself. The CSP plant is backed up by grid electricity for operations when the turbine does not supply power, which explains the contribution of "Operation and maintenance" to climate change, eutrophication, ionising radiation and dissipated water (impacts associated with the use of conventional electricity generation).

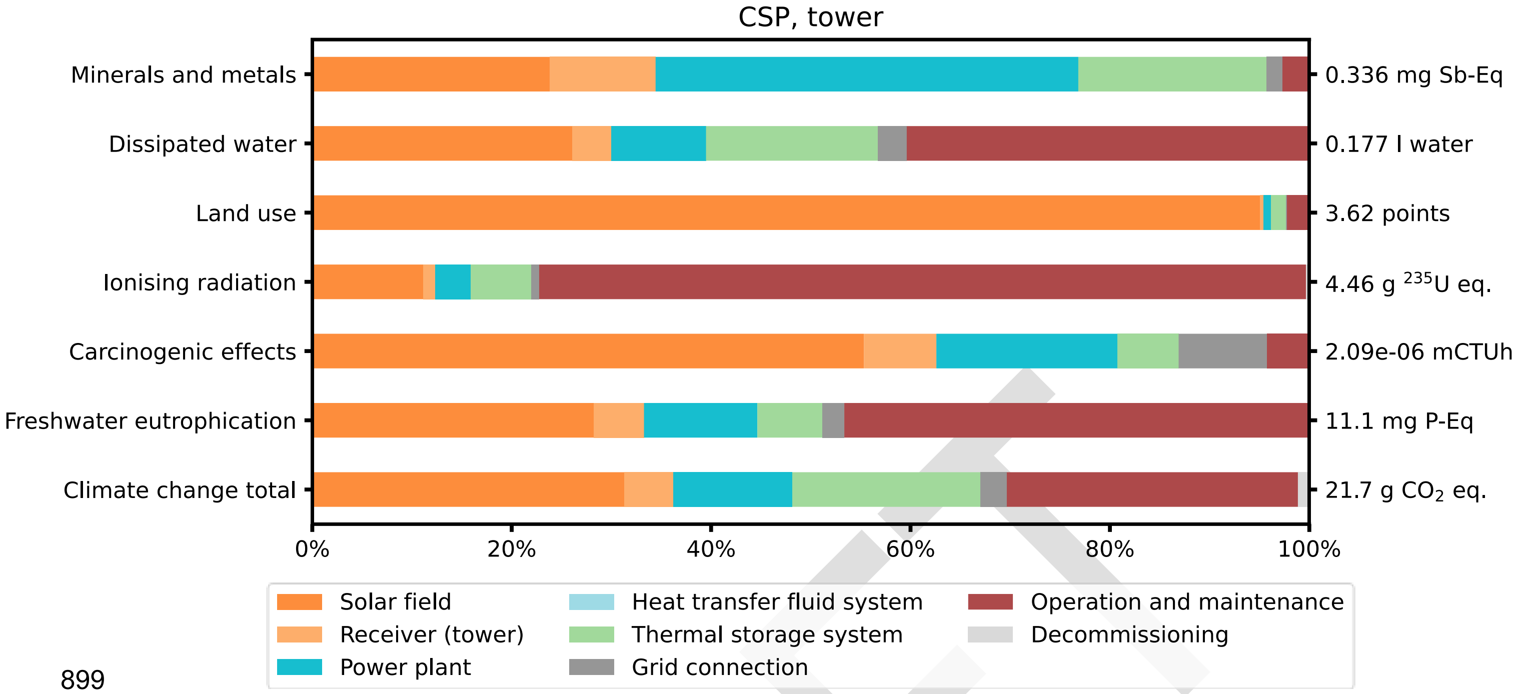


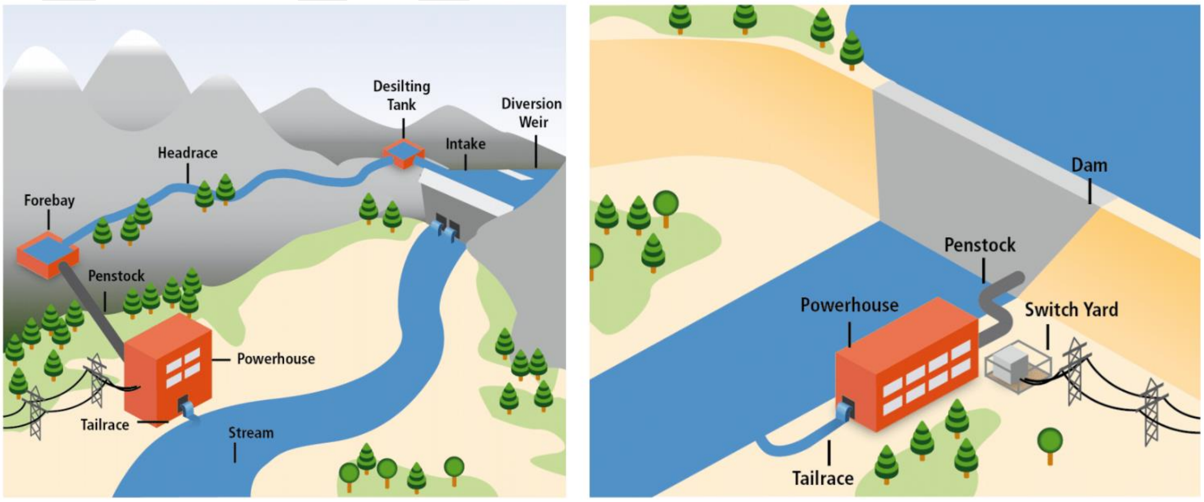
Figure 31. Life cycle impacts from 1 kWh of central tower concentrated solar power production, Europe, 2020.

3.6 Hydropower

Hydropower covers a wide array of technologies harnessing the forces of the natural water cycle. It is globally the dominating renewable resource in terms of electricity production.

3.6.1 Technology description

Designs are conventionally split into two main types: “run-of-the-river” and “reservoir”. The former type is usually smaller in size and capacity, whereas the latter usually delivers more power, and can also store potential energy by pumping water from a lower to an upper reservoir (it becomes a *pumped storage* project). In this study we only include non-storage, reservoir (without pumped storage) dams. Trivially, the impacts of pumped storage electricity depend highly on the impacts associated with the electricity used to pump the water, therefore it is excluded from our analysis – the IPCC clearly states that “pumped storage plants are not energy sources” [100].



3.6.2 Life cycle inventory

The data for the hydropower life cycle inventory was collected from two main projects in Chile [5]. Two power plants are modelled, of 360 MW and 660 MW of capacity respectively. The two projects are actually part of a larger hydroelectric complex in Patagonia – data was gathered from primary sources as reported in [5]. The expected lifetime of these dams is assumed to be 80 years, which corresponds to the average design life of 50–100 years of most global large dams [101].

Changes to original inventories

Regional load factors and electricity mixes have been adapted to match the various REMIND-MAgPIE regions.

Table 10. Load factors assumed for the hydropower designs.

Region	Hydropower, reservoir
CAZ	51%
CHA	50%
EUR	35%
IND	42%
JPN	35%
LAM	61%
MEA	35%
NEU	35%
OAS	47%
REF	55%
SSA	25%
USA	52%

3.6.3 Environmental impact assessment

The performance and environmental impacts of hydropower plants are highly site-specific. The specific topology of valleys flooded, local water regimes, latitude [102], … are as many factors influencing the overall environmental profile of a hydropower plant. Because of their influence on nutrient cycle, dams may be large sources of biogenic greenhouse gas emissions, especially in tropical conditions [103].

For the selected designs, the main contribution to lifecycle GHG emissions are from transportation during construction. This is specific to the modelled dams, as their location is relatively remote. Apart from transportation, the materials of the dam and turbines themselves are the next contributing elements to dissipated water and carcinogenic effects (25%–30%) – the latter is due to the use of stainless steel in the powerhouse. Overall, impacts are generally low in absolute terms, due to the long lifetime assumed for the dam, of 80 years.

A negative value appears for the land use category. The assessment method used, ILCD 2.0, contains characterisation factors that are either negative (when transforming an area from a “lesser quality” land) or positive (when transforming an area to a “higher quality” land). Building a dam will change the local area by transforming *a priori* unknown terrain to a water body. Unfortunately, the underlying model (LANCA) does not have characterisation factors for water bodies yet. As reported in [104]“*The LANCA model already provides CFs associated to a list of elementary flows compatible with the ILCD nomenclature. Therefore, no mapping was needed. The main difference with the original model presented in Bos et al. (2016) is the absence of CFs for elementary flows related to water bodies, hence, the land use indicator recommended for EF has no CFs for water bodies’ occupation/transformation. The reason behind this choice is that at the moment, LANCA addresses only the terrestrial biomes and not the aquatic ones.*”

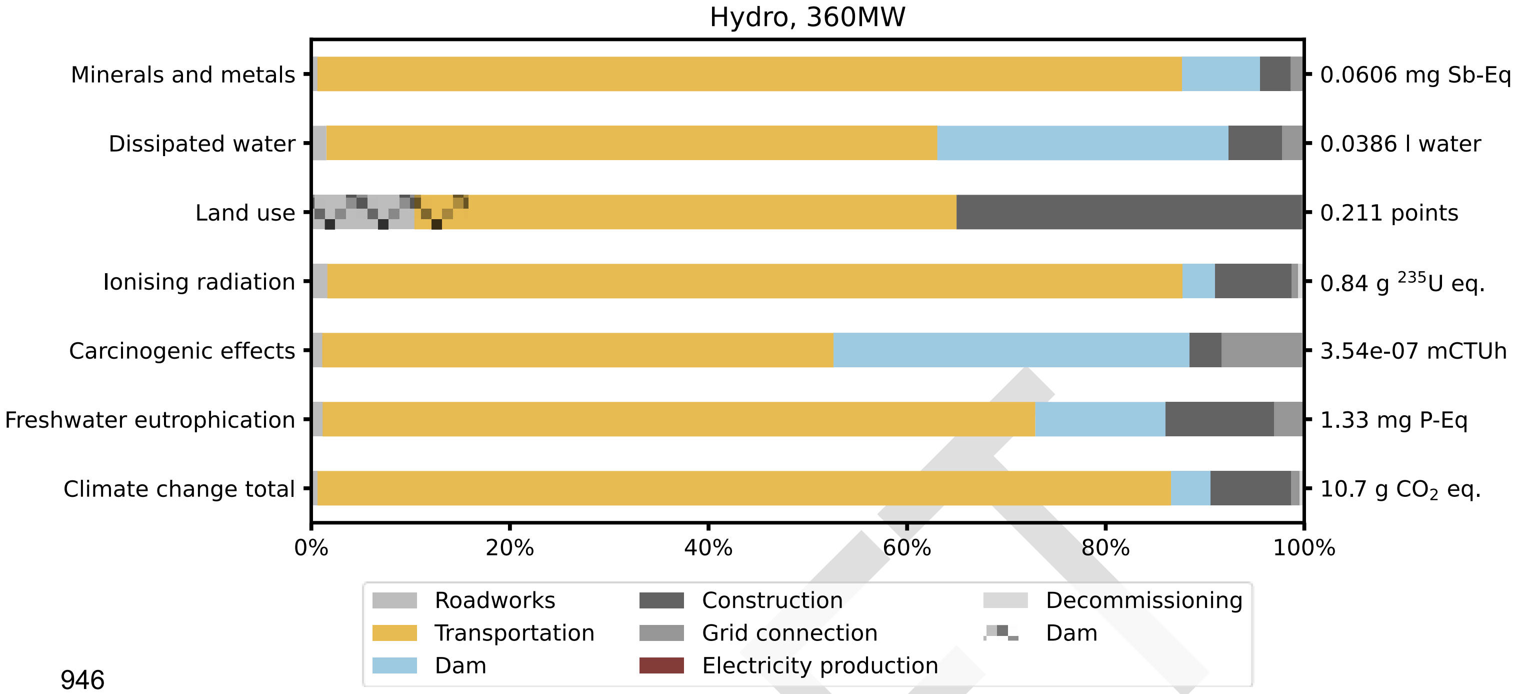


Figure 32. Life cycle impacts from 1 kWh of hydropower production, based on a 360-MW plant design, Europe, 2020.

3.7 Nuclear power: conventional

The term “conventional” nuclear power includes most of the fleet in operation today, i.e. pressurized water reactors, pressurized heavy-water reactors, boiling water reactors, or light water graphite-moderator reactors. As of early 2021, 443 of these nuclear power plants are in operation, providing 393 TW of power capacity [105]. The installed fleet delivered 2.6 PWh of electricity to the global grid in 2019, almost exactly 10% of the total that year. The IPCC characterizes nuclear power as able to deliver long-term low-carbon electricity at scale. However, nuclear power faces obstacles to its further deployment in some countries, among which public acceptance, high upfront costs, and challenges to the disposal of radioactive waste.

3.7.1 Technology description

Nuclear power reactors come in various designs, commonly classified into four categories, based on maturity, technology-readiness level, and more generally, the history of nuclear power development. **Generation I** reactors include the first prototypes operational in the 1950s and 1960s, which are no more in use today. **Generation II** include the majority of reactors in operation in 2021, namely light water reactors, with their two main variants, pressurised water reactors (PWR) and boiling water reactors (BWR), which dominate the market (see Figure 33). Generation II also includes heavy water reactors (such as the Canadian CANDU), fast neutron reactors (FNRs) or light water graphite reactors reactors (LWGRs) and advanced gas-cooled reactors (AGRs). **Generation III** designs build on Gen II reactors, with increased safety and economics. **Generation III** designs build on Gen II reactors, with increased safety and economics.

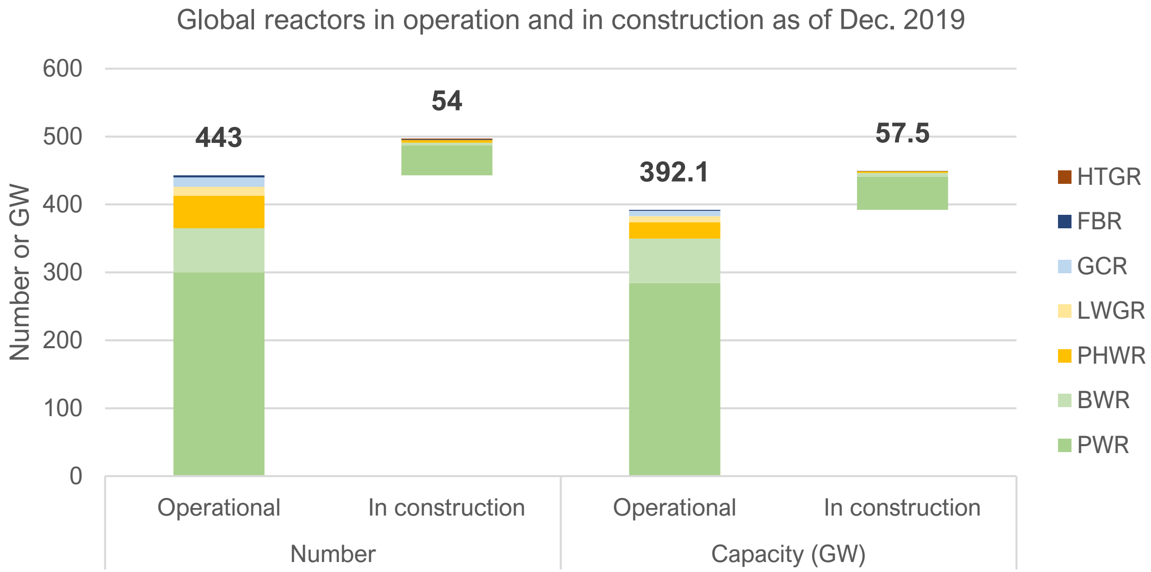


Figure 33. Snapshot of global nuclear power reactors, operational and in construction, as of December 2019. Source: IAEA [106].

Finally, the **Generation IV** category includes six main technologies under development, which offer various operational and environmental improvements over existing designs – the very-high-temperature reactor (VHTR), molten salt reactor (MSR), lead-cooled fast reactor (LFR), supercritical-water-cooled reactor (SCWR), sodium-cooled fast reactor (SFR) and the gas-cooled fast reactor (GFR). The last two of these designs are fast neutron reactors (FNRs) which have a common objective of "closing" the fuel cycle by design, thereby allowing the reuse of nuclear fuel for power generation, by reprocessing spent fuel. Several FNRs have operated historically and two are currently operating. These have all essentially been prototype units.

The present study aims at modelling the average conventional reactor in use as of 2020, in its two main variants, BWR and PWR. Some elements from Generation III reactors will be considered in the life cycle inventory (e.g. the amount of bulk materials in construction), mainly for information and comparative purposes.

The nuclear power fuel cycle involves the following steps:

- **uranium mining and milling**, extracting ore and then separating out the uranium for transport as a uranium oxide,
- **uranium conversion and enrichment**, converting the solid uranium oxide into gaseous UF6 for enrichment, which increases the concentration of the useful isotope ^{235}U,[5]
- **fuel fabrication**, converting the enriched uranium into a highly stable compound before loading into manufactured assemblies,
- **used fuel** management,
- high-level radioactive **waste management and disposal**.

The first steps, from mining to fuel fabrication, are commonly called "front end", while "back end" refers to the retreatment of the fuel. It is also possible to "reprocess" used fuel to recover useful isotopes and recycle uranium and plutonium as new fuel, However for simplicity reprocessing was not included in this study. "Core" processes generally refer to all operations occurring at the nuclear power plant site.

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[5] In physics and chemistry, the mass number A is conventionally noted as an upper-left exponent, it is the sum of neutrons and protons. Element ^{238}U has 146 neutrons and 92 protons, with A = 92 + 146 = 238, while its isotope ^{235}U only has 143 neutrons. The mass number is not to be confused with the number of atoms in a molecule noted as an index, e.g. CO_2 contains two oxygen atoms.

3.7.2 Life cycle inventory

This following section gives both a description of the various steps of the lifecycle as well as a description of the nuclear power life cycle inventory. Due to its centralised nature, and the scope of the work, we have chosen to model an average PWR reactor, representative of the global production in 2020. The frontend market (mining, milling, conversion, enrichment, fuel fabrication) is indeed shared between a a few suppliers, distributing their products globally. Only site-specific activities (core processes, i.e. plant construction and decommissioning, as well as operation) have been regionalised. The general parameters assumed for the modelled reactor and front-end global estimates are detailed in Table 11.

The premise of the study was to use inventories from the ecoinvent database version 3.7. However, it was recognized that for the nuclear power cycle, and especially for the first steps, this data is inaccurate. Therefore, supplemental data was provided regarding energy inputs, water requirements, chemicals in use, as well as for the high-level radioactive waste such as interim storage, encapsulation, and deep waste repository.

Table 11. Main parameters used for the nuclear LCA model. Front end values are calibrated on the global efficiency of the uranium supply chain as reported by the WNA.

Constants	Parameter	Unit	Value
Mining	Waste-to-ore ratio	-	5
	Ore grade	t U/t ore	0.21%
		t U308/t ore	0.25%
Milling	Extraction losses	-	4.05%
Conversion	Losses	-	0.00%
Enrichment	Enrichment rate	-	4.21%
	Tails assay	-	0.22%
	Cut	kg U/kg U	0.12
	SWU per kg feed	SWU/kg	0.82
	SWU per kg product	SWU/kg	6.67
Fuel fabrication	Losses	-	0%
	SWU per kWh	SWU/kg	6.74
Power plant	Burnup rate	GW-day/ton	42
	Efficiency	-	34%
	Nameplate capacity	MW	1000
	Lifetime	years	60

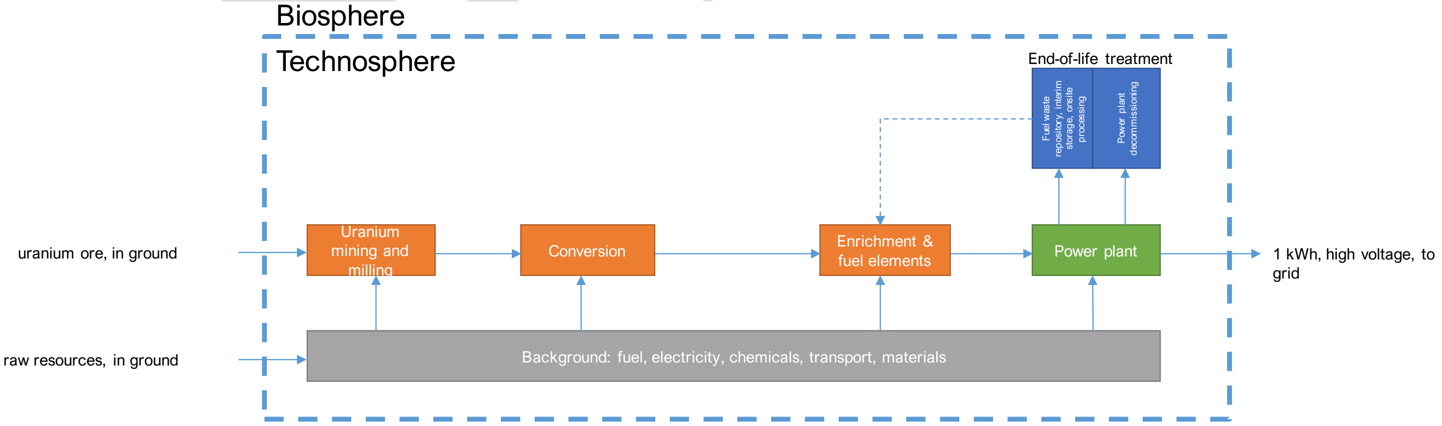

Figure 34. System diagram for conventional nuclear power technologies.

All data collected through scientific literature, technical reports, LCI databases and expert elicitation through consultations with the WNA is described in Annex, section 7.3.

3.7.3 Environmental impact assessment

From an environmental life cycle perspective, nuclear power has been shown to be low carbon, but also presents a number of co-benefits. It causes low land occupation and transformation over the life cycle, due to the high energy density of fuel elements, which minimizes mining area per kWh, and to the relatively low occupation of power plant sites. Human health and biodiversity impacts are overall low for PWR and BWR technologies.

On the other hand, nuclear electricity generation – as is routine in thermal plants – requires significant amounts of water primarily for cooling purposes. If open cycle cooling is used 1 kWh of output requires the withdrawal of up to 200 litres of water taken from and returned to the environment after a cycle. Between 1 and 3 litres will be lost due to downstream evaporation. If closed cycle cooling such as a cooling tower is used then 3-4 litres of water will be evaporated and consumed per kWh with withdrawal matching consumption Life cycle assessment studies have also shown moderate potential toxicity impacts from mining and milling. Finally, nuclear power is one of two technologies to show significant amounts of ionising radiation over its supply chain. Ionising radiation is an impact category included in most impact assessment methodologies to convey the potential impact due to radioactive emissions of materials, processes or products. Box 5 provides more details about ionising radiation modelling.

For every step in the lifecycle, global average data is used, meaning that the system diagram and material balance matches the various rates and efficiencies of the global industry, specifically averaged over the 2016-2020 period.

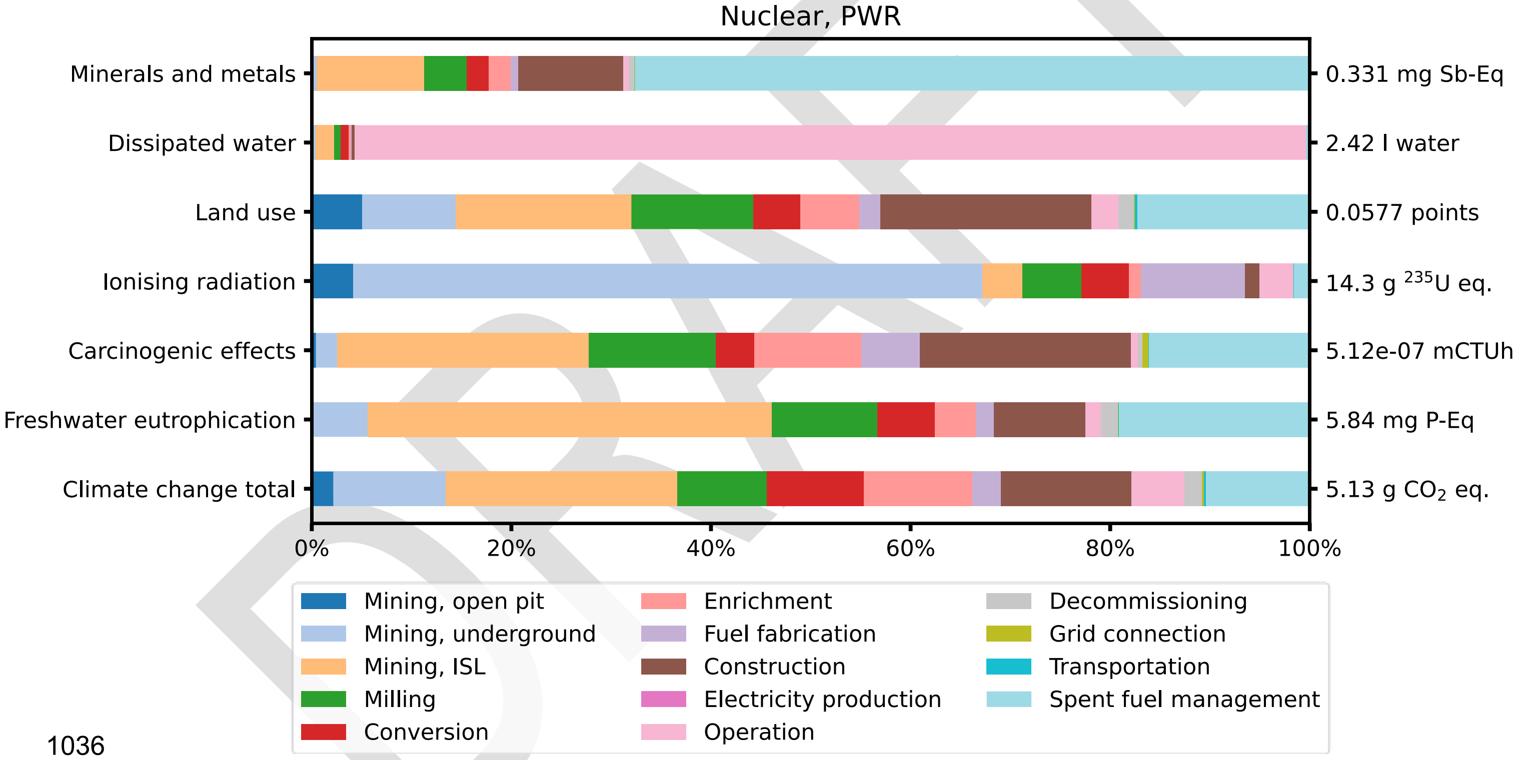

Figure 35. Lifecycle impacts of nuclear power, global average reactor, per kWh and activity.

As shown on Figure 35, front end processes, and especially mining, are main contributors to the overall life cycle impacts of nuclear power. Depending on the indicator, core processes and back-end activities come next, but do not contribute more than 30% and 10% to overall impacts, respectively. Energy use on site, mainly from diesel generators, are the main cause of GHG emissions for mining and milling processes.

Each MJ of fuel use (diesel, petrol, light fuel oil) contributes 86–105 g CO_2 eq./MJ. This translates into 0.22–0.26 g CO_2 eq./kWh for every 100 MJ of fossil energy inputs at the mining stage (at 25 mg U in ore per kWh), over the full lifecycle. These fossil fuel inputs are assumed to be 306 and 381 MJ/kg U in ore for open pit and underground mining, respectively, and 141 MJ/kg U in U_3O_8 for ISL mining.

3.8 Nuclear power: small modular reactors

3.8.1 Technology description

About 70 designs of SMRs are under development today. There is no strict definition of SMRs, but in practice they include **reactors under 300 MW** in size, as well as a high degree of modularity, for example, whole reactors can be designed to be transported by truck and installed on any site with minimal preparation. This flexibility theoretically reduces the time of construction and upscaling. Some designs can also follow load, more effectively than conventional nuclear plants and this make SMRs attractive regarding grid integration challenges. Overall, the development of SMRs provides access to nuclear power to countries that cannot accommodate large nuclear power plants for various reasons, be it costs or energy policy planning. It is recognised that deploying SMRs commercially would unlock access to nuclear power in new sectors and regions [107].

Four main categories of SMR can be differentiated, Water Cooled Small Modular Reactor, High-temperature gas-cooled reactors (HTGRs), Sodium-Cooled Fast Reactor (SFR) Technology and Molten Salt Reactor (MSR), but the variety of designs and the complexity of each technology reveal that building average and representative Life Cycle Inventory for each would be time consuming and overpass the objectives of the current project.

Water-cooled SMRs are among the most advanced designs for SMR (Locatelli et al. 2014), and few scientific papers are available in the literature, allowing to efficiently build a screening LCI representative of this technology. To do so, papers from Carless et al. 2016 and Godsey et al. 2019 were considered and compared in order to set an average LCI for water cooled SMR, considering the production of 1MWh electricity as the reference flow. The construction, operation and decommissioning of the SMR has been considered. Table 12 presents the main technical characteristics of the technologies respectively considered in each of the two papers investigated. The average inventory flows for water cooled SMR were derived first from Carless et al. 2016 and completed with inputs from Godsey et al. 2019, especially in regard to direct emissions during SMR operation and inputs – other than concrete – required for decommissioning.

Table 12: Technical characteristics for water cooled SMR technologies.

	Godsey et al. 2019	**Carless et al. 2016**	**Unit**
Technology	**LWR (NuScale Power)**	**Westinghouse-SMR (integrated) Pressurised Water Reactor**	
Electrical output	720	225	MWe
Lifetime electricity produced	360	114	TWh
Thermal output	2400	800	MWt
Capacity factor	95%	97%	
Thermal efficiency		28%	
Lifetime	60	60	years
Refueling cycle	24	24	months
Replaced fuel assemblies / modules per refueling	4	30	unit
Refueling outages duration		9	days
Total core load (U)	55	26.3	tons
Total fuel assemblies / modules	12	89	unit
Assembly/module electrical output	60	3	MWe/assembly
Construction duration	28.5	24	months

No life cycle inventory has been built for this exercise, due to a scarcity of data for non-LWR SMR reactors. Results from literature are presented in the next section.

3.8.2 Environmental impact assessment

Godsey [108] carried out a life cycle assessment for the NuScale SMR design, finding that per kWh of electrical output, the system would emit 4.6 g CO_2 eq./kWh. This is sensibly lower than the value reported by Carless, Griffin [109], of 8.4 g CO_2 eq./kWh. Both reactors being smaller versions of conventional light water reactors, this range of emissions coincides with commonly reported lifecycle GHG emissions of 1000 MW-scale reactors, including the value in this report, 5.6 g CO_2 eq./kWh under European (core and backend) conditions. Beyond GHG emissions, the same profile occurs for SMR and LWR, as shown on Figure 36, which can be roughly compared with Figure 35 (caveat: impact assessment methods are different). The mining and milling processes dominate the ionising radiation and toxicity indicators, and the uranium fuel chain in general dominates resource depletion and climate change impacts.

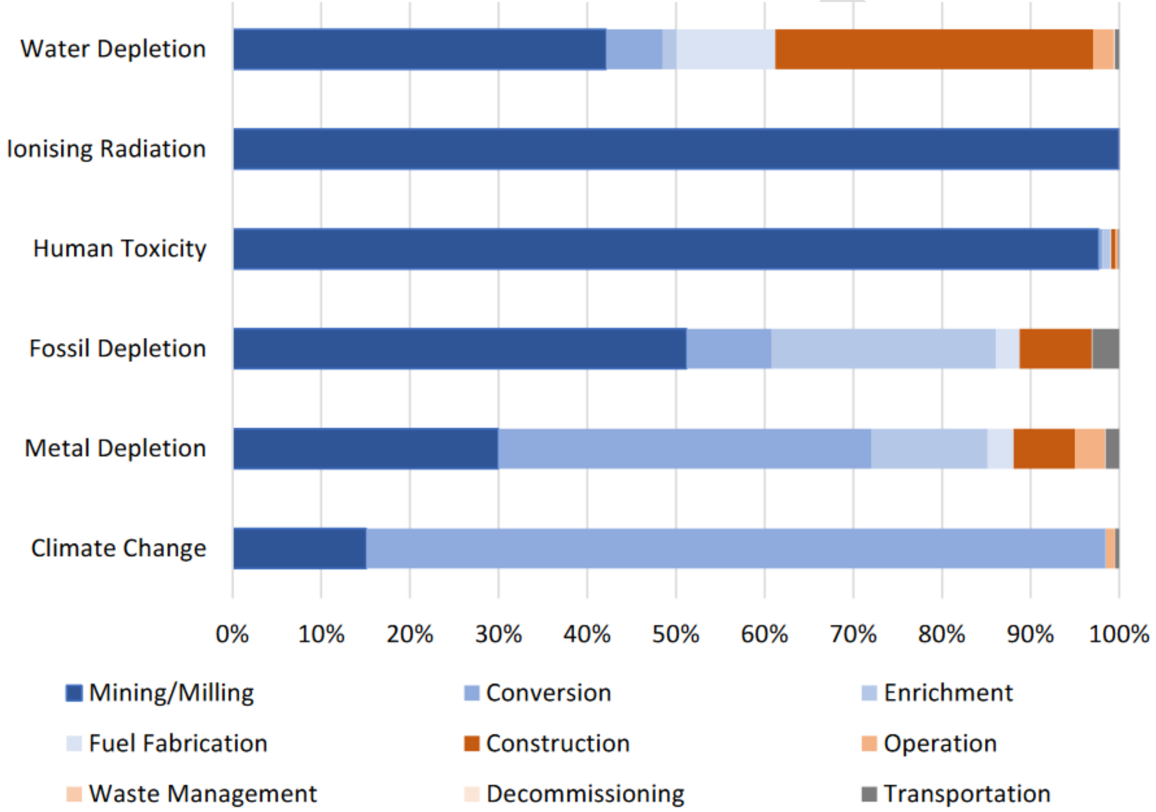

Figure 36. Lifecycle impacts of SMR technology, distribution across life cycle stages. Adapted from Godsey [108].

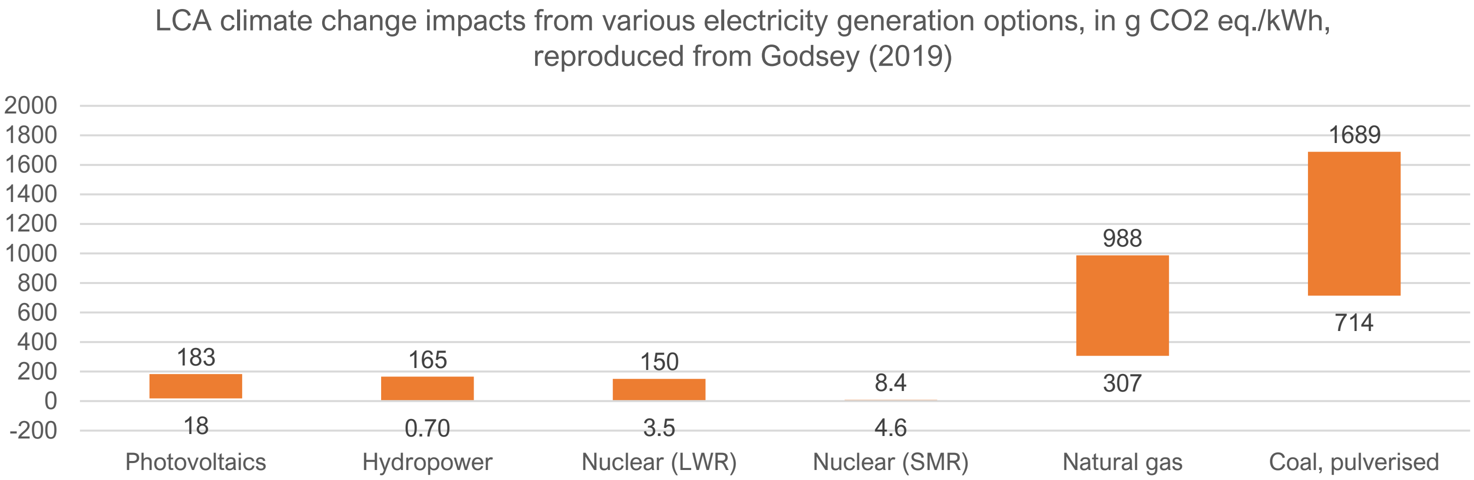

Figure 37. Maximum and minimum LCA climate change impacts of various electricity generators (kg CO2-eq/MWh). The nuclear SMR minimum is from Godsey [108] and the maximum is from Carless et al. [109].

4 Overall comparison

The impact indicators selected are climate change, freshwater eutrophication, ionising radiation, human toxicity (carcinogenic and non-carcinogenic impacts are shown in this section, although only carcinogenic is shown in technology-specific charts), land occupation, dissipated water, resource use (materials, non-renewable energy). Additional results for aggregated indicators are also shown at the end of the section, namely the single score results (normalisation and weighting) as well as two endpoint indicators, damage to ecosystems, and damage to human health.

4.1 Climate change

4.1.1 Regional differences

While the technology description is identical across regions, the site of operation plays a role for all technologies. The varying electricity mixes and industrial process efficiencies across world regions influence the environmental impacts of all systems, as energy inputs are a main contributor of infrastructure production. **Fossil fuel** extraction and supply are not described identically across regions – methane leakage rates indeed vary at the various stages (mostly for production and transportation), which plays a significant role on the results. Between 10% and 15% of greenhouse emissions are embodied in the fuel's supply chain in coal and gas systems, all variation occurs in that upstream phase for these technologies as plant efficiencies are assumed identical.

Hydropower emissions are mostly embodied in transport and infrastructure. The 660 MW plant should be considered as an outlier, as transportation for the dam construction elements is assumed to occur over thousands of kilometres (which is only representative of a very small share of hydropower projects globally). The 360 MW plant should be considered as the most representative, with fossil greenhouse gas emissions ranging from 6.1 to 11 g CO_2 eq./kWh. Biogenic emissions are not shown here, as they are highly site-specific. The absence of operational emissions, a long asset lifetime, and high load factors make hydropower perform relatively well regarding the GHG metric. For the same three reasons, **nuclear power**'s lifecycle emissions are estimated at 5.5 g CO_2 eq./kWh on a global average, with most of the emissions occurring in the front-end processes (extraction, conversion, enrichment of uranium and fuel fabrication). This value is comparable to the lower range of literature values because of the following assumptions: revised energy inputs for mining and milling, including electricity inputs for ISL, centrifugation-only enrichment, longer lifetime assumed for nuclear power plant (60 years instead of 40).

Concentrated solar power plants show high variability because of local conditions. In fact, the higher values correspond to regions where CSP would not be economically viable, such as Northern Europe or Japan. Under enough solar irradiation, CSP production emits 35-40 g CO_2 eq./kWh on the life cycle. **Solar PV** and **wind** technologies display low emissions too, with most GHG embodied in infrastructure. With the exception of polycrystalline silicon PV in certain regions, no technology surpasses 35 g CO_2 eq./kWh. Wind turbines offer consistently low emissions (under 16/23 g CO_2 eq./kWh for onshore and offshore respectively), regardless of their location.

These scores do not account for downstream supply of electricity, only connection to the grid is accounted for – transformation to lower voltages, incurred losses, and distribution lines to residential or commercial areas are not included. There is only one exception to this rule: roof-mounted PV, which technically delivers low-voltage electricity to households, readers should be aware that **the assessment scope is therefore different for roof-mounted PV technologies**.

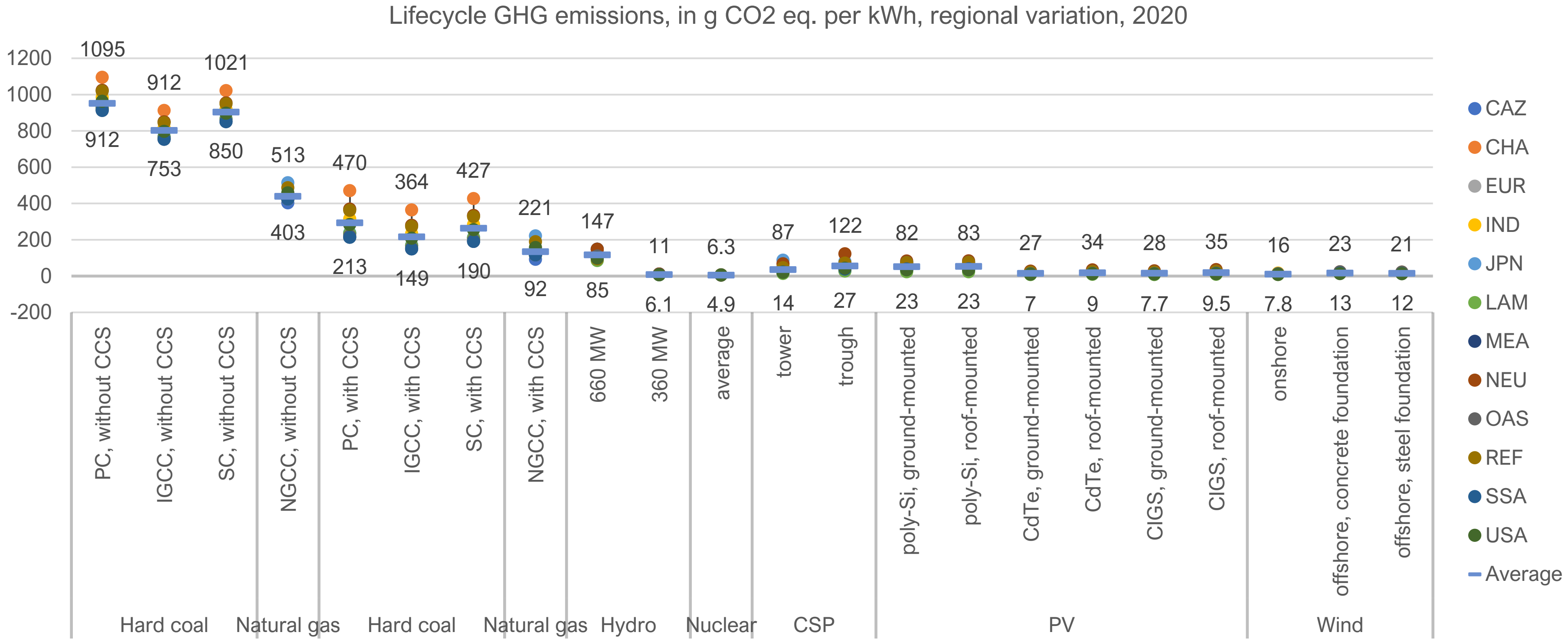

Figure 38. Lifecycle greenhouse gas emissions' regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), methane leakage rates (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average except for back-end.

4.1.2 Prospective assessment

The evaluation of environmental impacts in 2020 context is not enough to support long-term policies. As the energy transition is ongoing, modes of production (energy, industry) may undergo radical changes themselves, meaning that the very same electricity technologies assessed in this exercise may have a significantly different environmental profile by 2050, depending on the scenario followed.

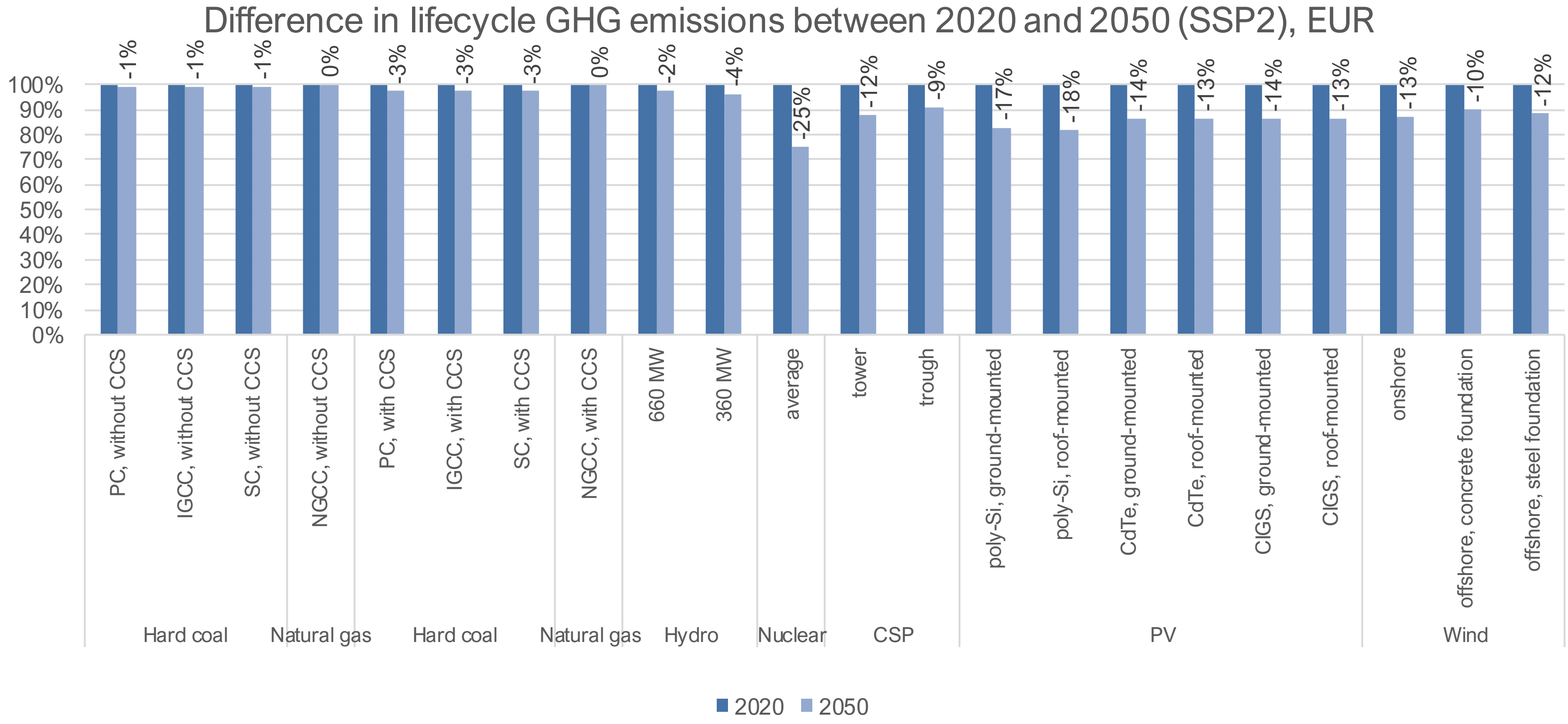

Figure 39. Differences in lifecycle greenhouse gas emissions between 2020 and 2050, due to the evolution of background electricity mixes and industrial processes. Please note that no change in the technology datasets themselves have been modelled for this figure.

4.2 Freshwater eutrophication

Freshwater eutrophication is caused by the emissions of phosphorus compounds to freshwater bodies (rivers or groundwater). The main source of phosphate emissions across all the studied systems is the treatment of spoil from coal mining. Depending on the coal source, variations occur: 1 kg of coal extracted in Australia requires the treatment of 15 kg of spoil from mining activities, this amount falls to about 5 kg in other world regions; which explains the 1:3 range in freshwater eutrophication between Japan, Australia and the rest of the world. On the other hand, coal extraction in China does not emit as much phosphate according to the ecoinvent data, hence the significantly lower value for that region. Non-coal technologies cause very low amounts eutrophication, principally through the use of coal electricity in the background, or from metal extraction (namely copper).

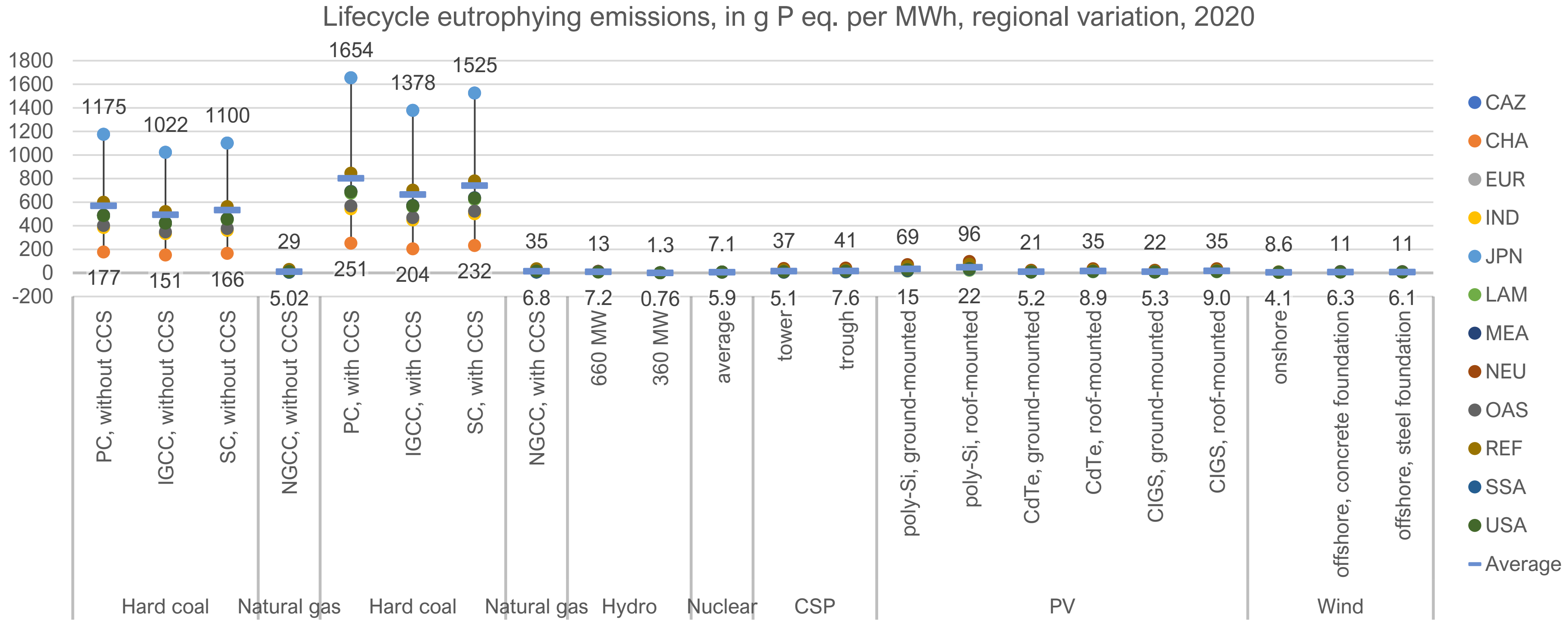

Figure 40. Lifecycle eutrophying emissions' regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), methane leakage rates (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average except for back-end.

4.3 Ionising radiation

Ionising radiation is caused by the exposure of humans to radioactivity. **As explained in Box 5, radioactive emissions from radionuclides are lumped sum regardless of the amount or time of exposure (as done with emissions of other substances)** *de facto* following a linear no-threshold approach. This approach has been criticised for being too simplistic [110]. Nuclear power is the only technology that uses radioactive material as a main fuel, and for which radioactive emissions are systematically measured and accounted for – consequently, it is the only technology in our portfolio that shows ionising radiation emissions with **475 g ^{235}U eq./kWh** (with conservative assumptions) or **14 g ^{235}U eq./kWh** (realistic assumptions)[6]. In comparison, coal power shows a range of **9-15 g ^{235}U eq./kWh**. Recent research suggests however that occupational exposure also occurs for other technologies (namely geothermal power over their life cycle, and to a lesser extent photovoltaics at the mining phase), this is also detailed in Box 5. The rest occurs, in small amounts (about a few grams per kWh) over the front-end chain, mostly conversion and enrichment. Other technologies' impact on ionising radiation originates in the use of nuclear power for electricity.

Box 5. Ionising radiation modelling, no-threshold linear model, and impact assessment

The LCA indicator "ionising radiation" encompasses all radiations that are energetic enough to detach electrons from molecules. The human environment has always been radioactive and exposure from natural sources accounts for up to 85% of the annual human radiation dose, with medical sources contributing most of the remainder. The worldwide average human dose is 2.4 mSv per year, but some regions natural background more than 10 times this value. High doses and high dose rates of ionising radiation are well-known to cause detrimental health effects and increase the incidence of certain cancers. At low doses (below 100 mGy) and low dose rates (below 0.1 mGy/min) however, there is insufficient statistical evidence to prove carcinogenic effects [111]. A conservative approach has nevertheless been adopted by the scientific community, extrapolating the dose vs cancer risk at high dose to the low-dose domain. This approach is called the Linear No-Threshold (LNT) model, and assumes a health detriment from ionising radiation regardless of how low the dose is. As a precautionary principle for nuclear power energy sources, the 103rd publication of the International Commission on Radiological Protection (ICRP 103) advises a maximum dose limit of 20 mSv per year for nuclear workers, and 1 mSv per year for the general public.

The "no lower threshold" assumption leads to the accounting of health effects from the first becquerel emitted by a radionuclide (or rather the first millisievert of received dose) – in other words, that if a certain dose of radiation is found to cause one extra case of cancer in a given population, then one-tenth of that dose will cause one extra case in ten times the population size. Since radiological studies need to be based on large enough sample sizes to be statistically significant, the question of the actual linear scalability of the dose-response relationship arises.

The LNT assumption, now a paradigm in radiology, has regularly been criticised for oversimplifying the health effects of radiation, and specifically for exaggerating the effects of small doses which would empirically be undetectable. Sacks, Meyerson [110] qualify the LNT hypothesis as "gigantic scientific oversight", which should therefore be interpreted with caution. UNSCEAR and ICRP both clearly advise that collective dose is not an appropriate tool for epidemiological studies and risk projection [2].

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[6] The original ecoinvent inventory shows emissions of ^{222}Rn from milling tailings include an integration time over 80000 years (roughly the half-life of ^{230}Th of which ^{222}Rn is a progeny), and the non-remediation of tailing repository sites – resulting in 35 TBq per kg of U_{nat} extracted (conservative assumptions). UNSCEAR publishes collective dose values with a 100-year integration, the time horizon we retain for the realistic assumptions. Plasma torch incineration emissions are adjusted to align with the latest data at the Zwilag plant (2017, as opposed to original ecoinvent data: 1993).

In life cycle impact assessment, ionising radiation from the decay of radionuclides is characterised using an impact pathway approach, following Dreicer, Tort [18], further refined in Frischknecht, Braunschweig [17] and Huijbregts, Steinmann [112]. Specifically, Frischknecht, Braunschweig [17] rely on data published in Dreicer, Tort [18] for the fate and exposure modelling, and also assume a "LNT behaviour for low doses of ionising radiation". Two main models are used to calculate the impact of airborne and waterborne radionuclides in the current LCIA method, although more are described in [18], namely for underground release and transportation accident. This modelling is based on a radionuclide's properties, and is therefore required for each of them. Current life cycle impact assessment methods (ILCD, ReCiPe, LC-IMPACT) have inherited the same modelling assumptions, including the one used in this study.

Collective dose from non-nuclear technologies. Exposition to radionuclides is not exclusive to nuclear power-related activities. Resource extraction in general is a source of exposition for workers due to the natural presence of radionuclides in ores. However, it has been shown that coal power plants also contribute significantly to the overall collective dose because of direct combustion and coal ash deposits. Likewise, geothermal power, also generate exposure during operation, showing the highest rate when calculated per unit of electricity generated, as shown on Figure 41.

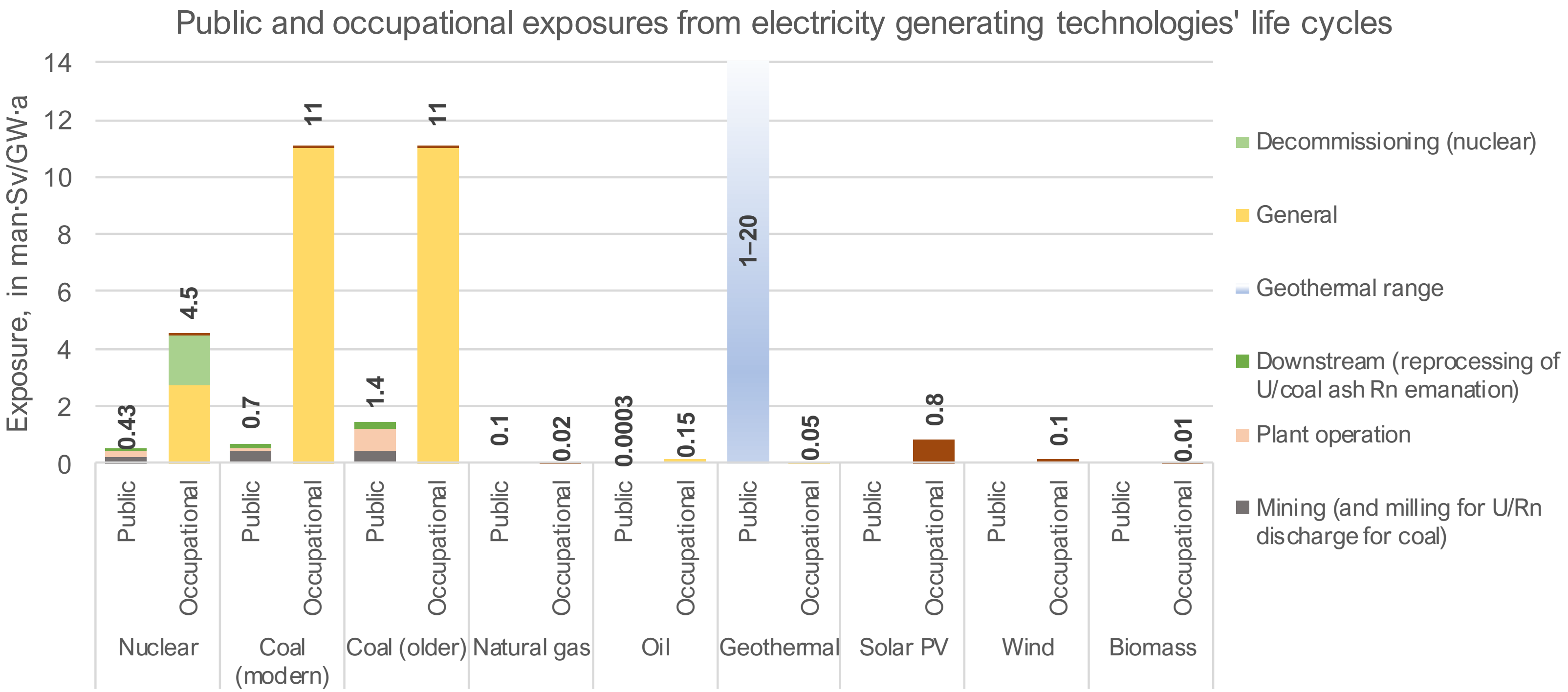

Figure 41. Public and occupational exposures from electricity generation, normalized to electricity generated, in man-Sievert per GW-annum (8760 GWh). Source: United Nations Scientific Committee on the Effects of Atomic Radiation [2].

4.4 Human toxicity

Human toxicity is assessed using two indicators: non-carcinogenic effects, and carcinogenic effects. Regarding **non-carcinogenic effects**, coal power displays the highest scores, with averages of 54-67 CTUh[7]/TWh and 74–100 CTUh/TWh without and with CCS respectively. The main contributing substance is arsenic (in ionic form), emitted to surface and groundwater, from coal extraction and treatment of hard coal ash at landfill. The next highest average is photovoltaic, poly-Si roof-mounted, with 14 CTUh/TWh, due to relatively high copper inputs, inducing arsenic ion emissions from the treatment of copper slag in landfills. The rest of technologies also emit small amounts of arsenic ion to water through the production of cast iron, ferronickel, and steel alloys.

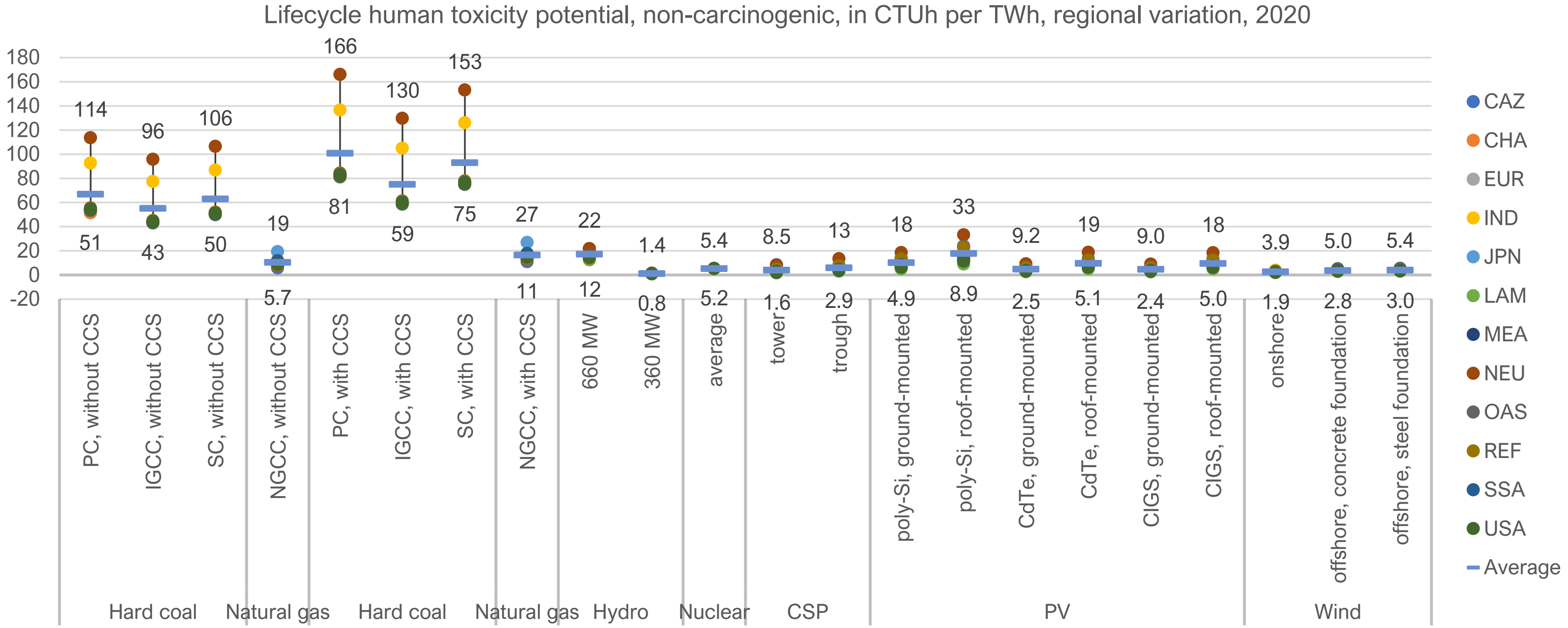

Figure 42. Lifecycle human toxicity (non-carcinogenic)' regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), region of extraction rates (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average except for back-end.

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[7] Comparative toxic units indicate the estimated increase in morbidity in the total human population.

Arsenic ion emitted to water has one of the highest factors for this category (0.0273 CTUh/kg). Regional variation is highly influenced by the share of coal imported from South Africa in each region's supply mix. This finding is supported by studies showing abnormally high arsenic content in South Africa and other African countries' waters, due to coal mining operations and other industrial activities [113, 114]. This is true for African regions, India, but also Europe, which imports about 6% of its hard coal consumption from South Africa and Mozambique.

As for **carcinogenic effects**, no average score surpasses 8.0 CTUh/TWh. This value is reached by the CSP trough plant, and due to the relatively high amount of stainless steel required for the infrastructure (also seen in section 4.7). The main substance contributing to this potential impact is hexavalent chromium (chromium VI), emitted to water (0.0106 CTUh/TWh). In fact, practically all technologies' human toxicity impact is linked with the amount of Cr(VI) emitted in water over their lifecycles, which is tied to the used of alloyed steel and the treatment of electric arc furnace slag (landfilling), a process that emits about 6 g of Cr(VI) in water for every kg of slag treated. Residual chromium emissions to air and arsenic (ion) emissions to water from waste treatment processes also contribute (<10%) to this impact category.

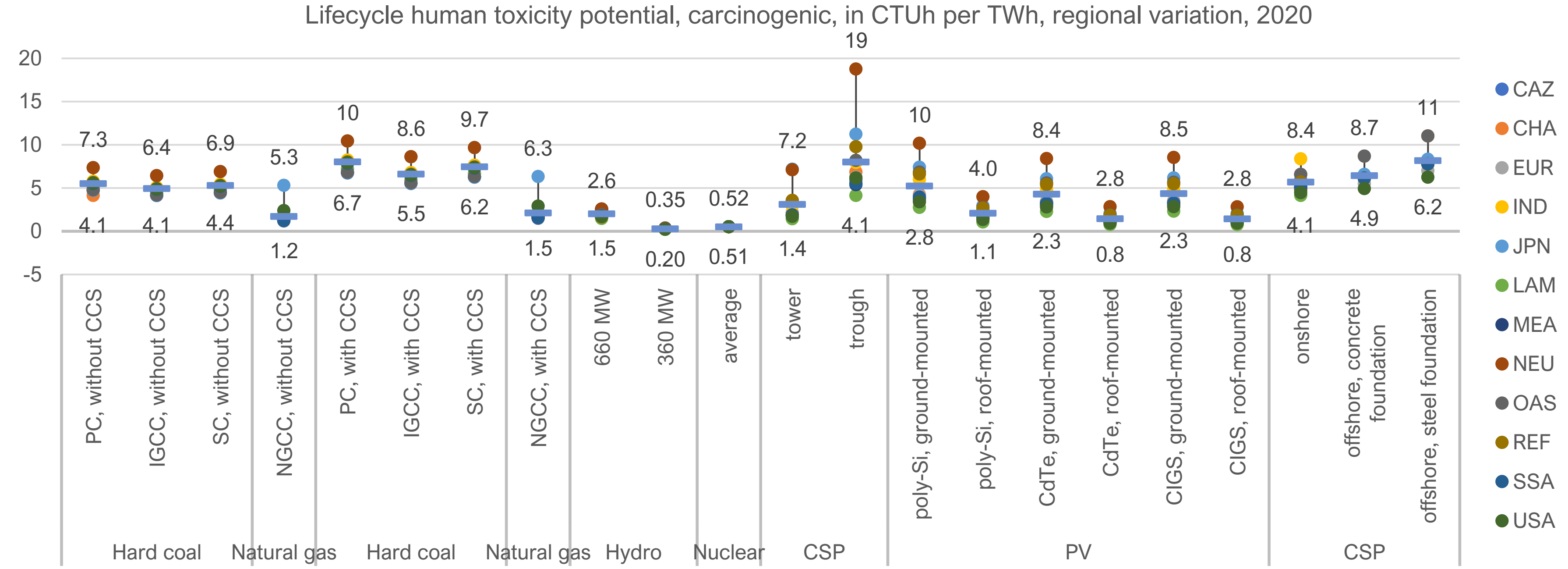

Figure 43. Lifecycle human toxicity (carcinogenic)' regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), region of extraction (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average except for front-end.

4.5 Land occupation

Land occupation (or use) includes both agricultural and urban land occupation, direct and indirect. For coal power, land occupation occurs mostly at the extraction phase, either through the mining infrastructure itself (open pit or underground) and the use of timber props in underground mines (timber is still a popular choice of material for roof support in mines [115]), which entails land use impacts from forestry. Natural gas does not entail high amount of land use, as natural is extracted from underground, and power plants do not use significant space. Hydropower projects, again, have site-specific characteristics, including for land occupation; the river, valley, and reservoir topology can make the land use indicators vary by orders of magnitude. This indicator is expressed in points, yielding a score for land quality[8] (see factors in Table 32). For the raw occupation values in m2a, see section 7.2.2.

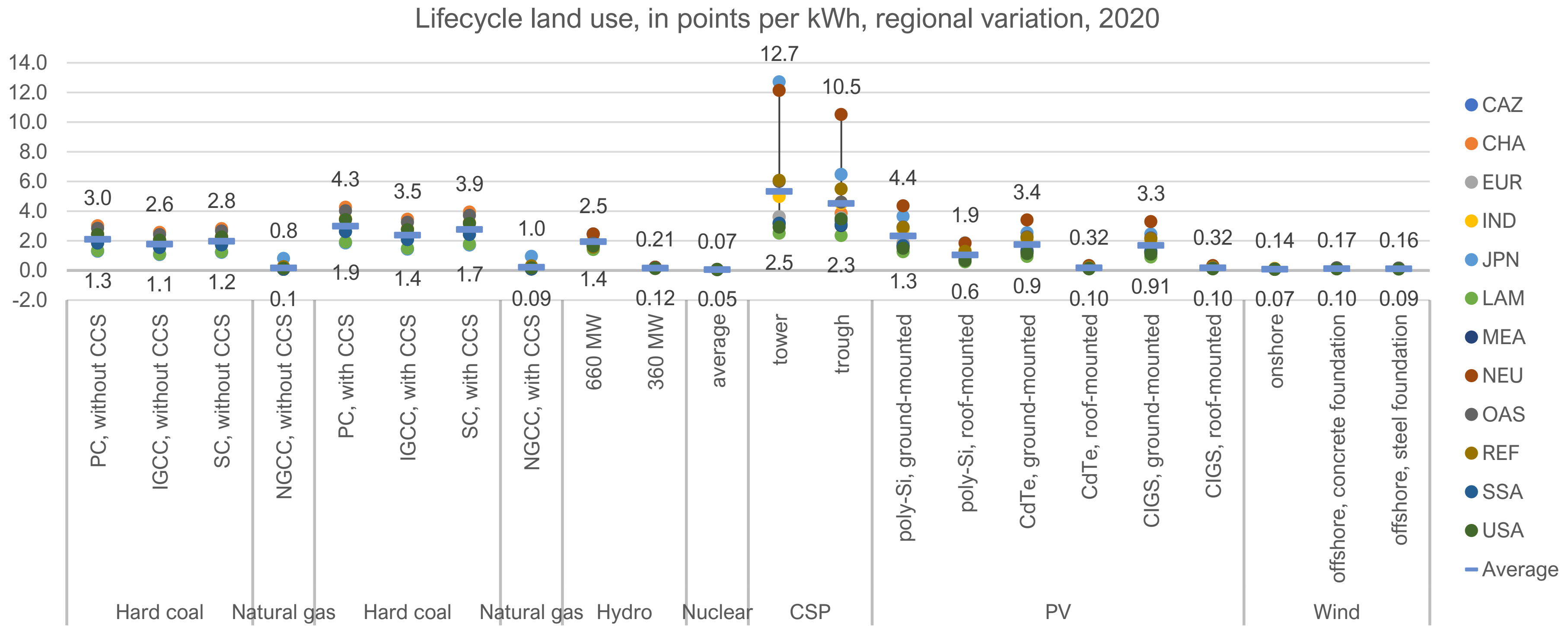

Figure 44. Lifecycle land use regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), methane leakage rates (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average and therefore does not see any variation.

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[8] Namely: erosion resistance, mechanical filtration, physicochemical filtration, groundwater regeneration, and biotic production.

4.6 Dissipated water

Dissipated water includes all uses that immediately deprive the local environment of using water, this indicator indicates scarcity of the water resource. For example, water immediately returned to the environment (in river, ocean, or groundwater) is not accounted towards “dissipated water”; while water used as an ingredient for a chemical product, or evaporated, is. Thermal power plants show high requirements of dissipated water as they deprive their immediate environment of readily available water for cooling. These requirements (on average) range from 1.0 m^3 per MWh, or l/kWh (natural gas without CCS), to 2.4 m^3 per MWh (nuclear power), to 5.0 m^3 per MWh (pulverised coal with CCS). For renewables, solar technologies have a moderate water footprint, which is mostly due to the use of electricity as backup (CSP) or the manufacturing of silicon cells (PV).

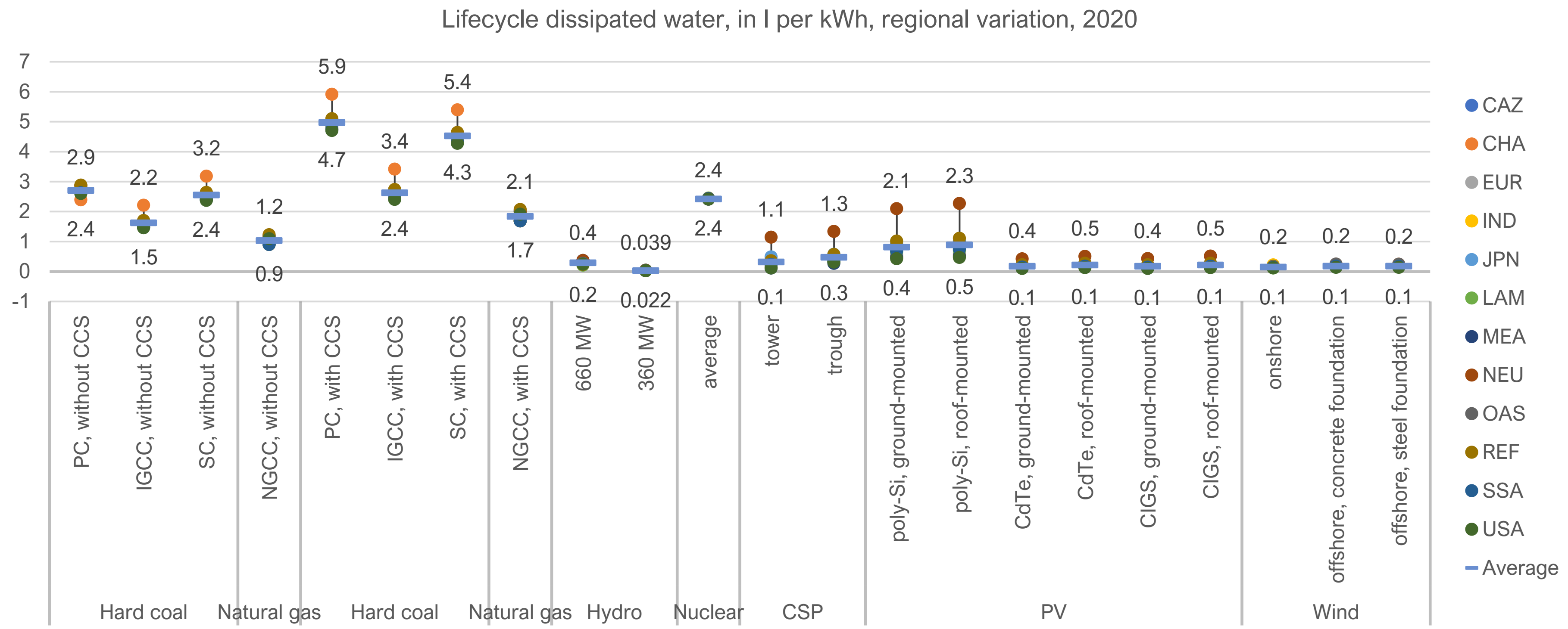

Figure 45. Lifecycle water requirement regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), methane leakage rates (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average except for back-end.

4.7 Resource use, materials

The resource use indicator characterises the elementary flows of resources extracted from the ground with a coefficient of scarcity. It aims at conveying one dimension of the criticality of materials, namely the supply risk (see Box 2 for a short explainer on material criticality). This coefficient is calculated from the estimated reserves of each element (e.g. gold, copper, chromium…) and compared to that of antimony, hence the unit in kg Sb equivalents. Photovoltaic systems contain slight amounts of gold and silver, used in power electronics, which shows the high score for this indicator as these elements have a factor orders of magnitude higher than copper or aluminium. No rare earth element is accounted for in the characterisation method, and using bulk materials like gravel, iron, and even aluminium barely has no influence on this indicator – which supports the low score of some infrastructure-intensive technologies such as hydropower.

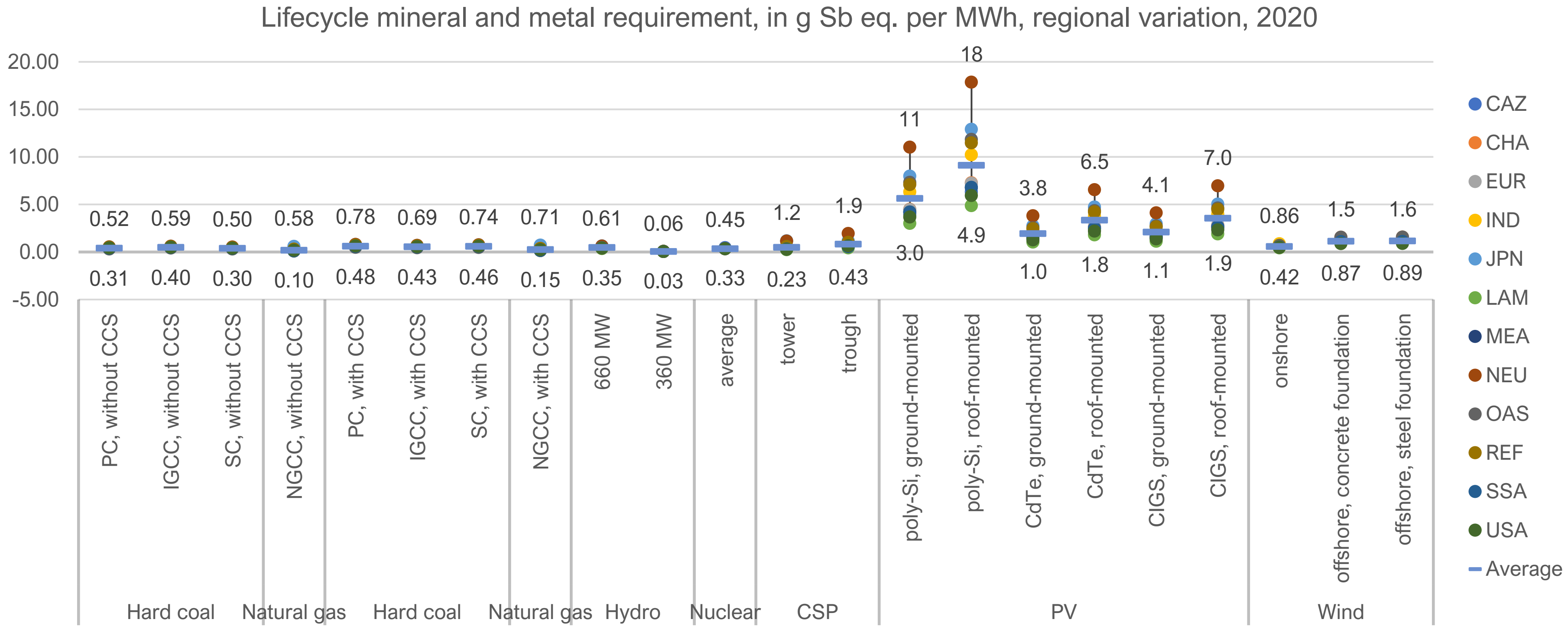

Figure 46. Lifecycle water requirement regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), methane leakage rates (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average except for back-end.

With the “scarcity” caveat in mind, another way to represent resource use is to list the uncharacterised inventory for each technology, i.e. to lump sum the list of materials directly from the life cycle inventories. Figure 47 shows the lifecycle amount of materials required, in g per MWh, using the same selection as International Energy Agency [24], namely: chromium, cobalt, copper, manganese, molybdenum, nickel, silicon, and zinc – to which we choose to add aluminium, given its very low abiotic depletion characterisation factor (i.e. it has virtually no influence on the results in Figure 46). Results exhibit wide disparities between technology. Regarding chromium, concentrated solar power consumes the most of it due to the stainless steel embodied in the infrastructure, namely the solar field for the trough design (300 g/MWh). Wind turbines are relatively steel-intensive and show a demand of 60-70 g of chromium per MWh. All technologies demand aluminium and copper, for infrastructure, connections and cabling. Photovoltaics appear as the most copper-intensive technology of the portfolio, because of electric equipment (general installation, inverter). Copper demand for nuclear appears through the use of copper canisters for high-level waste deep repository disposal and reflects the data sources used for this report.

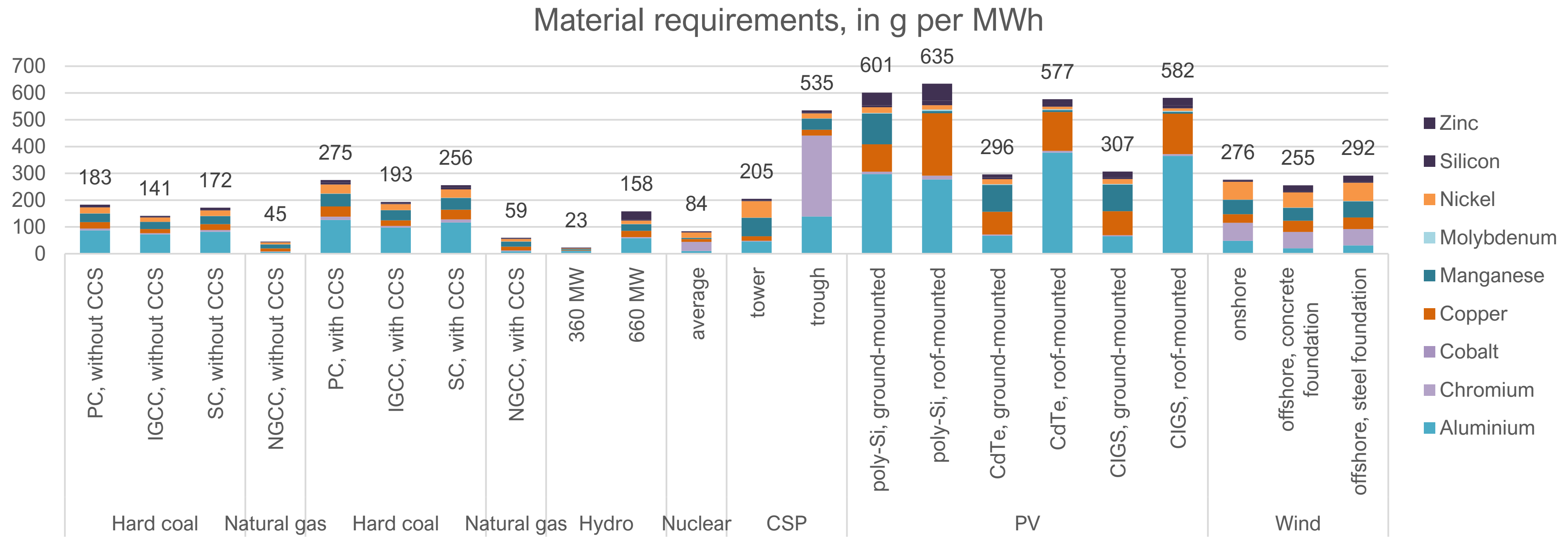

Figure 47. Lifecycle requirements of select materials for electricity technologies, in g per MWh.

4.8 Resource use, fossil energy carriers

Cumulative energy demand is calculated from lump summing primary energy carriers' energy content over the lifecycle of a system. Fossil technologies show a high score, slightly exceeding the inverse of the efficiency of a power plants, because of losses along the fuel supply chain. For CCS-equipped power plants, the energy penalty due to the capture facility, transport of carbon dioxide, and infrastructure of storage is clearly visible on Figure 48.

In the "cumulative energy demand" methodology, uranium is accounted as "fossil", which is technically not correct – **therefore it was removed from the list of elementary flows**. Uranium is accounted as a non-renewable primary energy resource with a characterisation factor of 560 GJ/kg of uranium ore[9] [117]. Note that uranium can be reprocessed after nuclear fuel is spent, as opposed to fossil energy carrier which undergo non-reversible dissipation (in other terms, coal, gas, or oil are not recoverable after combustion).

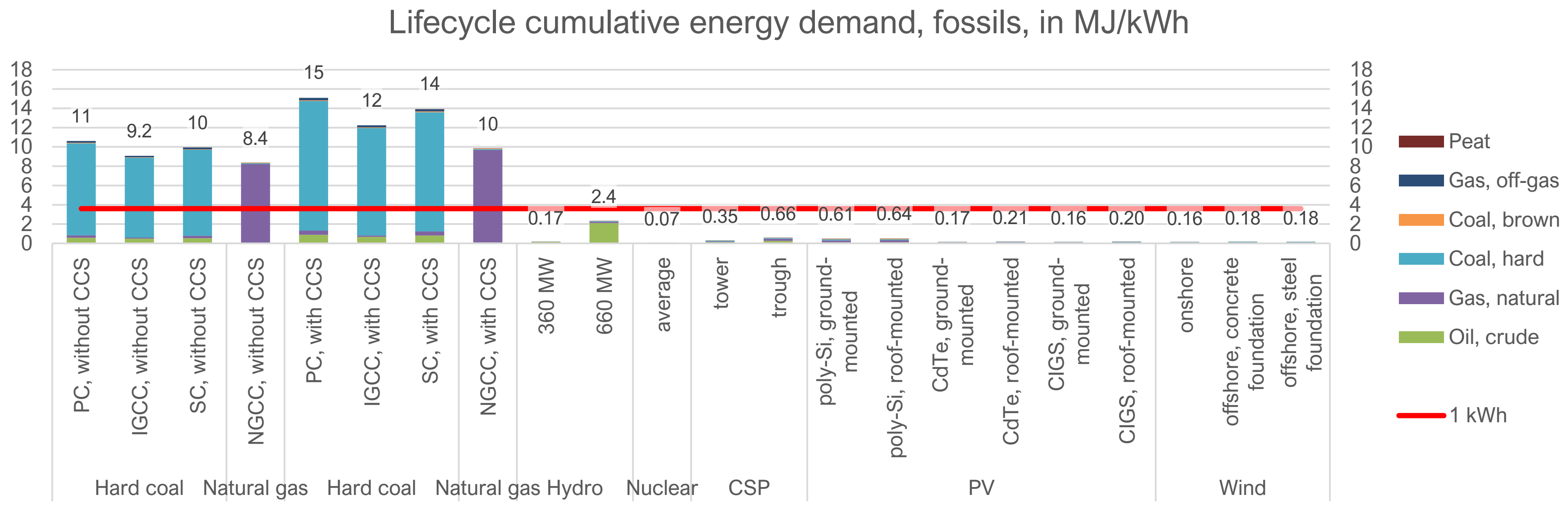

Figure 48. Cumulative energy demand, all energy carriers, in MJ per kWh electricity.

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[9] This value is the standard average used in the characterisation method. For information, the amount of uranium ore required per kWh is about 25-30 mg/kWh_e at plant – which would translate to 8.3-10 mg/kWh_{th} or 7.0-8.3 mg U_{nat}/MJ_{th}. This suggests a heating value of 140 GJ/kg ore, all losses excluded. The discrepancy between this estimate and the primary factor given to uranium in the "cumulative energy demand" method is identified [116].

4.9 Additional results for EU28

4.9.1 Endpoint indicators

Ecosystems

Endpoint indicators relate to the actual consequences of environmental impacts on three areas of protection: human health, ecosystem quality, and resources. They are not recommended by the latest JRC guidelines, but provide a different way of presenting aggregated results. Figure 49 displays impacts on ecosystems, in points, the result of normalisation and weighting. Climate change is overwhelmingly contributing to impacts on ecosystems, with slight impacts from natural land transformation for hydropower. The influence of CCS on fossil fuel plants is clear as it reduces ecosystem damage by 60–77%. Land occupation barely appears, yet it is the next contributor after climate change, as discussed in the next paragraph.

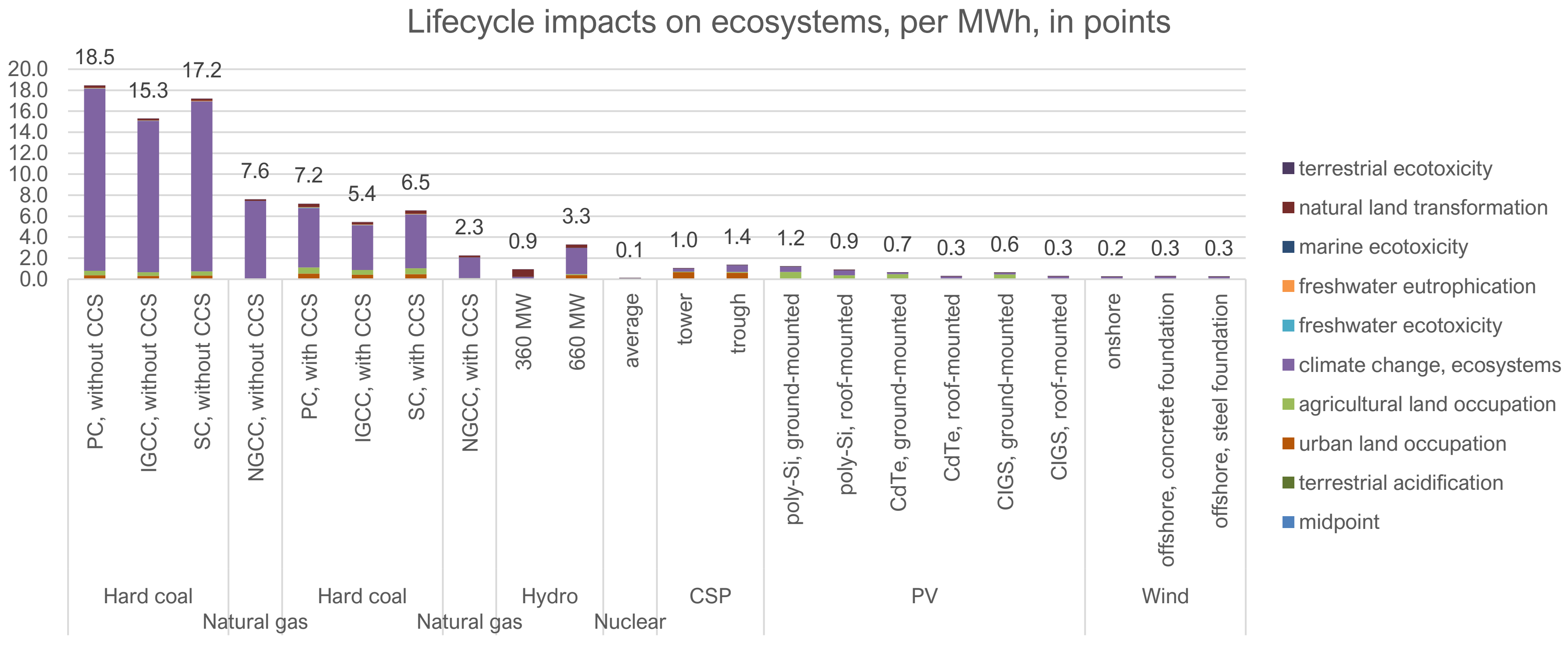

Figure 49. Life cycle impacts on ecosystems, in points, including climate change. Note on unit: 1 point is equivalent to the impacts (in species-year) of 1 person (globally) over one year.

When excluding climate change (Figure 50), land use categories explain most of the ecosystem damage, these are urban land occupation, agricultural land occupation, and natural land transformation. Transformation only occurs for fossil fuels and hydropower – as their lifecycle will generate a permanent change in land areas. Occupation without transformation occurs for renewable technologies, which have been assumed to be readily built on various land types without heavy modifications (such as land sealing, mountaintop removal, flooding, …). Roof-mounted PV, wind power, and nuclear power show a very low score on the ecosystem damage indicator.

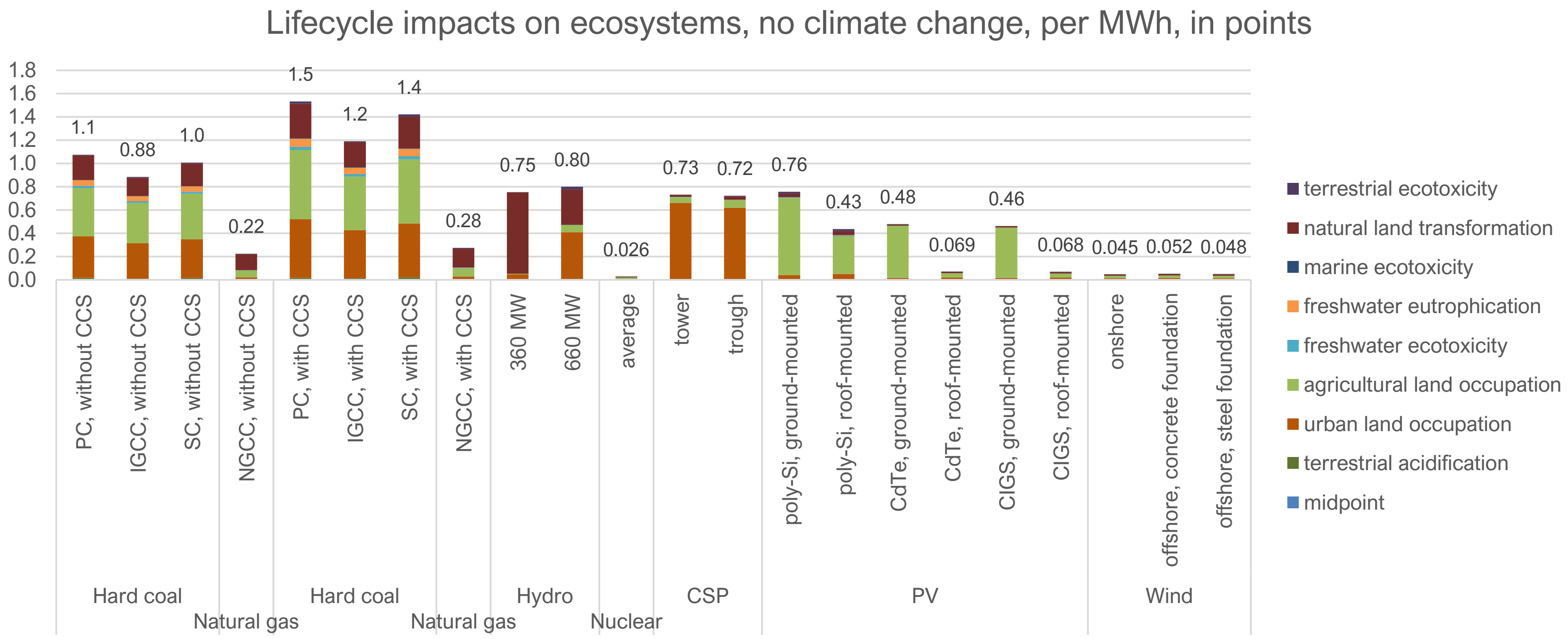

Figure 50. Life cycle impacts on ecosystems, in points, excluding climate change. Note on unit: 1 point is equivalent to the impacts (in species-year) of 1 person (globally) over one year.

Human health

The endpoint indicator for damage on human health is also dominated by climate change (>75% for all technologies) except for CCS-equipped plants, where human toxicity and particulate matter emissions are significant. Particulate matter emissions are significant for hard coal only, as the combustion of natural gas does not emit substantial amount of particles (unlike results from Gibon, Hertwich [11]). When excluding climate change, only human toxicity and particulate matter emissions remain as the main contributors to human health damage. It is important to note that **these results are normalized and weighted**, as is proposed in ReCiPe 1.13 – which marks a change in endpoint indicator units from ReCiPe 1.03.

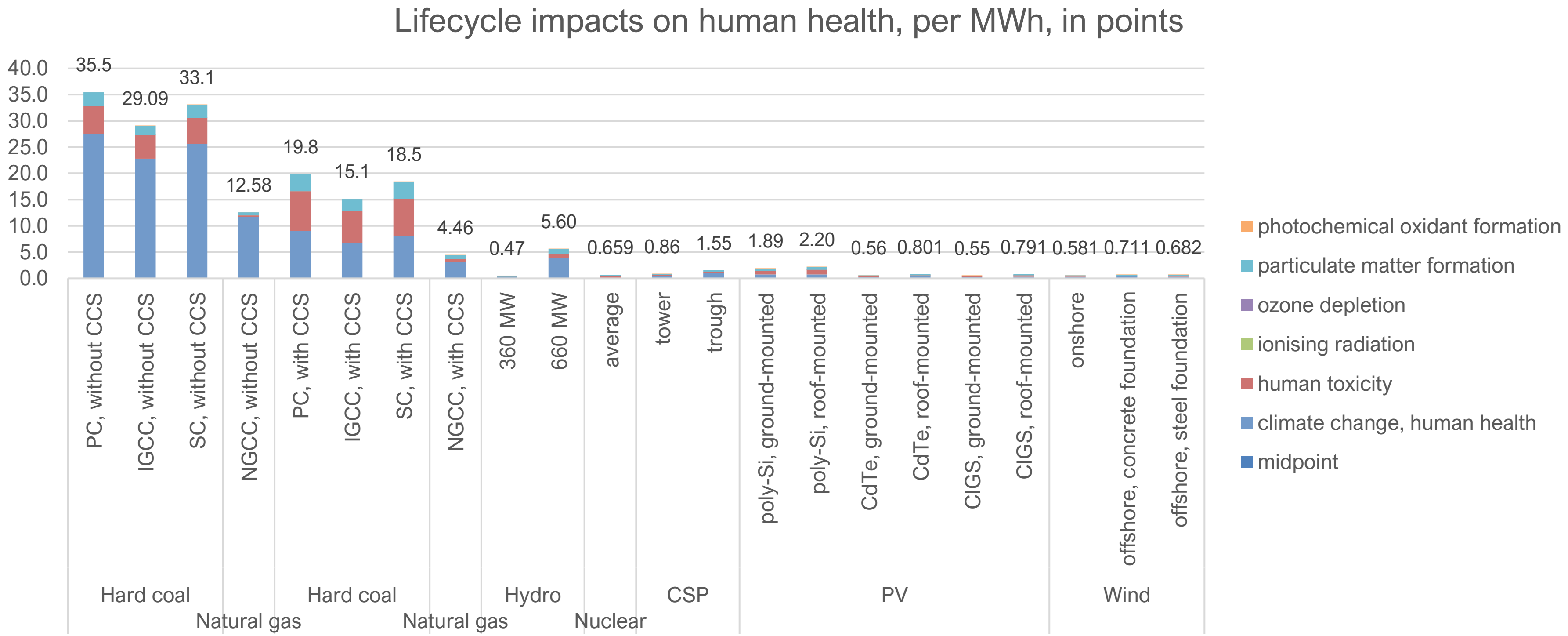

Figure 51. Life cycle impacts on human health, in points, including climate change. Note on unit: 1 point is equivalent to the impacts (in disability-adjusted life years, DALY) of 1 person (globally) over one year.

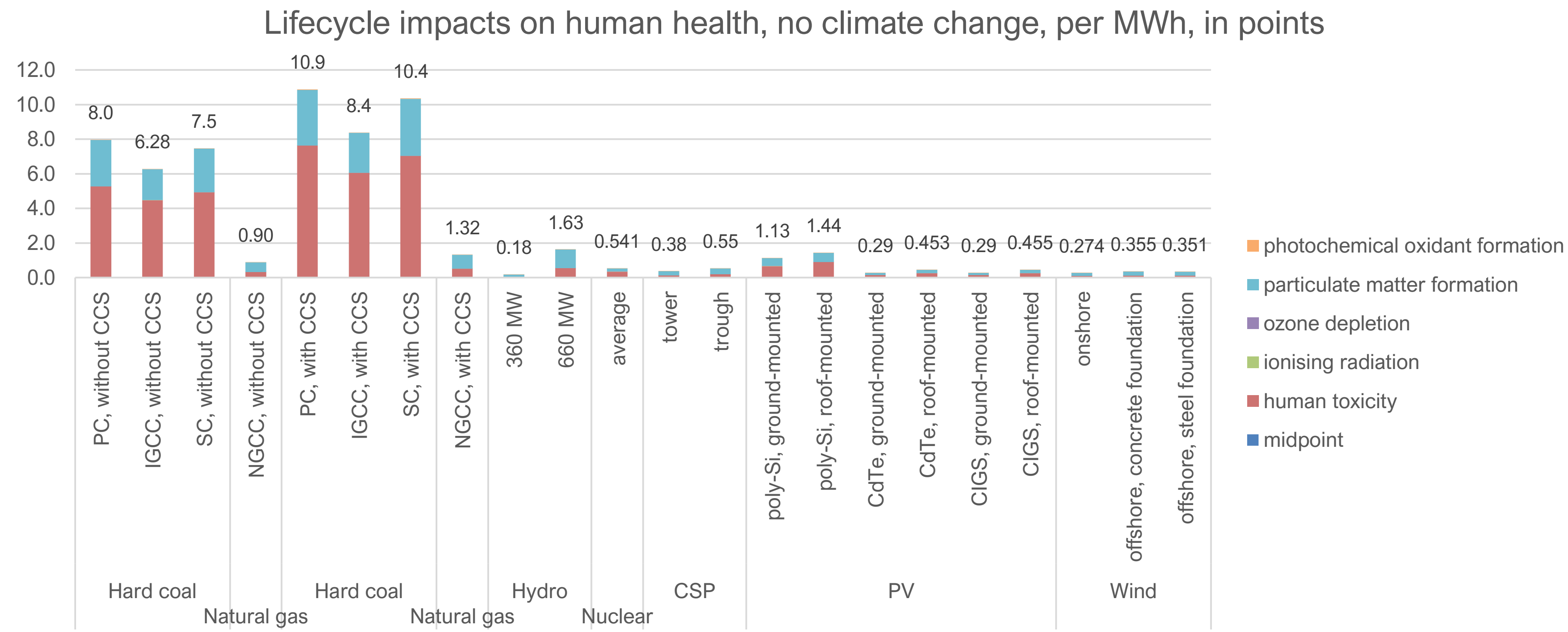

Figure 52. Life cycle impacts on human health, in points, excluding climate change. Note on unit: 1 point is equivalent to the impacts (in disability-adjusted life years, DALY) of 1 person (globally) over one year.

4.9.2 Single score: normalisation and weighting

Normalisation and weighting allow the hierarchisation of life cycle impact categories. By relating the environmental impact scores of each technology option to the global footprint of human activities, either total or per capita, all indicators can be aggregated as one score. Figure 53 shows the results of this normalisation for region Europe, in 2020. Hard coal displays the highest scores, namely 86–137 capita-equivalent per TWh (i.e. producing 1 TWh generates as much environmental impact as the footprint of 100 persons over one year, averaged over all categories). Most of this averaged impact is due to freshwater eutrophication, then resource use (fossils) and ionising radiation equally contribute. Nuclear power shows a low score (when not accounting for uranium as "fossil", see section 4.7). For renewables, human toxicity is the main contributor, with mineral use (PV only).

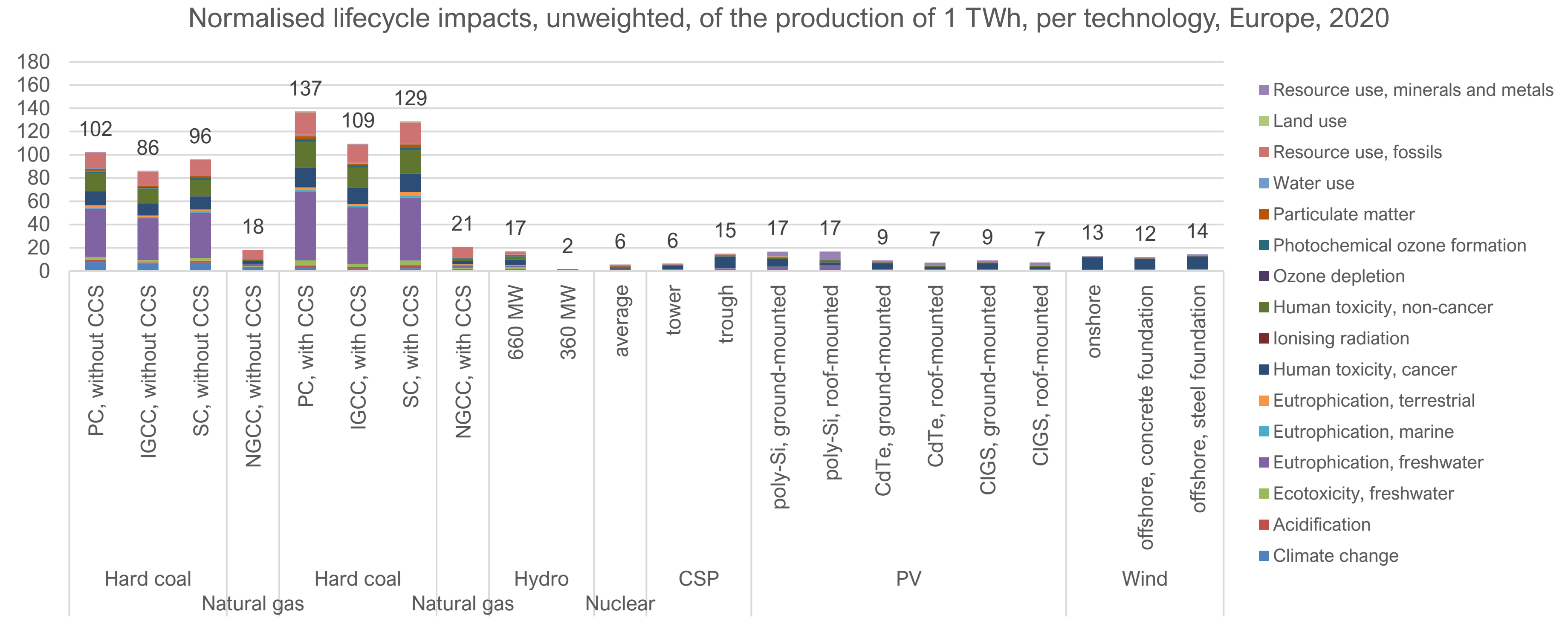

Figure 53. Normalised, unweighted, environmental impacts of the generation of 1 TWh of electricity.

To increase the relevance of normalisation, indicators can be hierarchised further, namely through a expert-defined weighting set composed of criteria such as spread of impact, reversibility, or level of impact compared to planetary boundary. This weighting set is then corrected with robustness factors, indicative of the uncertainty inherent to the impact assessment model behind each impact category. Details can be found in [13].

When weighted, normalisation scores decrease, chiefly because of the lesser weight given to eutrophication or toxicity effects. On the other hand, climate change contribution to the overall scores increase. These results, shown in Figure 54, have been used to establish a hierarchy used to select the environmental impact indicators to explore in detail in the study (see section 2.4).

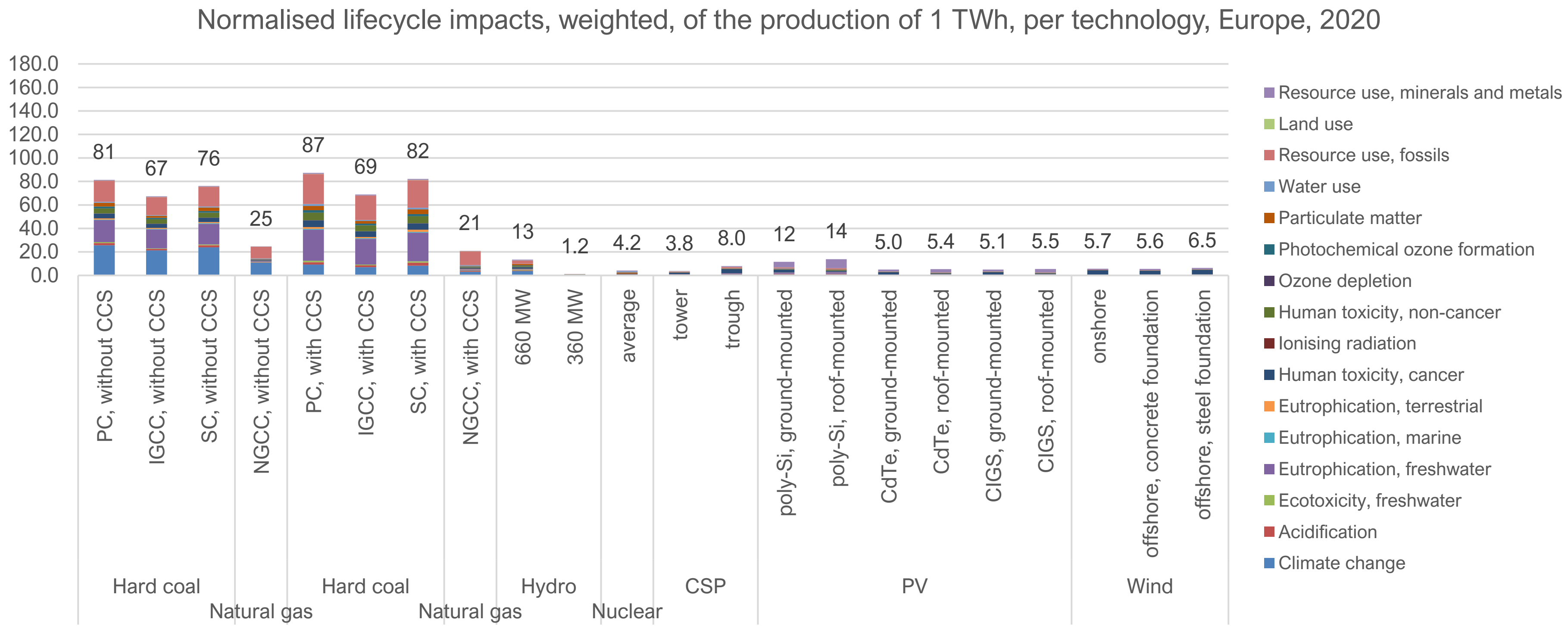

Figure 54. Normalised, weighted, environmental impacts of the generation of 1 TWh of electricity.

5 Conclusions

5.1 Discussion

The overarching objective of this report is to assess the **lifecycle environmental impacts of electricity generation options**. This has been performed by performing an LCA on updated life cycle inventories of select technologies. Specifically, hard coal, natural gas, hydropower, concentrated solar power, photovoltaics, wind power, as well as nuclear, have been evaluated regarding the following indicators: climate change, freshwater eutrophication, ionising radiation, human toxicity, land occupation, dissipated water, as well as resource use.

Regarding **GHG emissions**, coal power shows the highest scores, with a minimum of 751 g CO_2 eq./kWh (IGCC, USA) and a maximum of 1095 g CO_2 eq./kWh (pulverised coal, China). Equipped with a carbon dioxide capture facility, and accounting for the CO_2 storage, this score can fall to 147–469 g CO_2 eq./kWh (respectively). A natural gas combined cycle plant can emit 403–513 g CO_2 eq./kWh from a life cycle perspective, and anywhere between 49 and 220 g CO_2 eq./kWh with CCS. Nuclear power shows less variability because of the limited regionalisation of the model, with 5.1–6.4 g CO_2 eq./kWh. On the renewable side, **hydropower** shows the most variability, as emissions are highly site-specific, ranging from 6 to 147 g CO_2 eq./kWh. As biogenic emissions from sediments accumulating in reservoirs are mostly excluded, it should be noted that they can be very high in tropical areas. Solar technologies show GHG emissions ranging from 27 to 122 g CO_2 eq./kWh for CSP, and 8.0–83 g CO_2 eq./kWh for photovoltaics, for which thin-film technologies are sensibly lower-carbon than silicon-based PV. The higher range of GHG values for CSP is probably never reached in reality as it requires high solar irradiation to be economically viable (a condition that is not satisfied in Japan or Northern Europe, for instance). Wind power GHG emissions fluctuate between 7.8 and 16 g CO_2 eq./kWh for onshore, and 12 and 23 g CO_2 eq./kWh for offshore turbines.

Most of renewable technologies' GHG emissions are embodied in infrastructure (up to 99% for photovoltaics), which suggests high variations in lifecycle impacts due to variations in raw material origin, energy mix used for production, the transportation modes at various stages of manufacturing and installation, etc.

Notable deviations from published literature occur for several technologies, as shown on Figure 57. First, hard coal, without CCS, is shown to have an impact of over 911 g CO_2 eq./kWh in all cases (across technologies and regions), while the IPCC gives a *maximum* value of 910 g CO_2 eq./kWh. Differences in assumed power plant efficiencies explain this difference, as discussed in Box 1. Second, results for nuclear power are within the lower range of published literature. Several reasons explain this discrepancy: the assumed lifetime of 60 years for the power plant (instead of more commonly used 40 years), the absence of energy-intensive diffusion enrichment (mainly centrifuges are in use today), and revised energy inputs for mining and milling (increased share of ISL extraction).

All technologies display very low **freshwater eutrophication** over their life cycles, with the exception of coal, the extraction of which generates tailings that leach phosphate to rivers and groundwater. CCS does not influence these emissions as they occur at the mining phase. Average P emissions from coal range from 600 to 800 g P eq./MWh, which means that coal phase-out would virtually cut eutrophying emissions by a factor 10 (if replaced by PV) or 100 (if replaced by wind, hydro, or nuclear).

Ionising radiation occurs due to radioactive emissions from radon 222, a radionuclide present in tailings from uranium mining and milling – as a consequence, only nuclear power shows a contribution to this indicator. Coal power may be a significant source of radioactivity, Growing evidence that other energy technologies emit ionising radiation over their life cycle has been published, but data was not collected for this exercise (see Box 5 and [2]).

Human toxicity, non-carcinogenic, has been found to be highly correlated with the emissions of arsenic ion linked with the landfilling of mining tailings (of coal, copper), which explains the high score of coal power on this indicator. Carcinogenic effects are found to be high because of emissions of chromium

VI linked with the production of chromium-containing stainless steel – resulting in moderately high score for CSP plants, which require significant quantities of steel in solar field infrastructure relatively to electricity generated.

Land occupation is found to be highest for concentrated solar power plants, followed by coal power and ground-mounted photovoltaics. Variation in land use is high for climate-dependent technologies as it is mostly direct and proportional to load factors: 1-to-5 for CSP, 1-to-3.5 for PV, and 1-to-2 for wind power. The same variations can be found for water and material requirements.

Water use (as dissipated water) was found high for thermal plants (coal, natural gas, nuclear), in the 0.90–5.9 litres/kWh range, and relatively low otherwise, except for silicon-based photovoltaics, as moderate water inputs are required in PV cell manufacturing.

Material resources are high for PV technologies (5–10 g Sb eq. for scarcity, and 300–600 g of non-ferrous metals per MWh), while wind power immobilises about 300 g of non-ferrous metals per MWh. Thermal technologies are within the 100–200 g range, with a surplus when equipped with carbon capture. Finally, fossil resource depletion is naturally linked with fossil technologies, with 10–15 MJ/kWh for coal and 8.5–10 MJ/kWh for natural gas.

5.2 Limitations

ISO-compliant LCAs conventionally contain uncertainty and sensitivity analyses, in order to understand and quantify the influence of certain parameters over the LCIA results. This has not been systematically applied due to a stringent timeline, but should be investigated in order to increase the robustness of results. That being said, literature provides a rather clear overview of the sensitivity of electricity generation LCAs to certain assumptions – at least for GHG emissions. Regarding renewables, assumed lifetimes and load factors are two main parameters [118]. Fossil fuel inventories, on the other hand are generally sensitive to power plant efficiency assumptions, linked with the turbine technology and type of feedstock (e.g. for coal: anthracite, bituminous coal, subbituminous coal, lignite), as well as origin of feedstock (e.g. for gas: conventional vs. shale gas) and corresponding fugitive emissions. As for nuclear power, lifecycle GHG emissions depend chiefly on front end assumptions: mining mix and techniques, uranium ore grade, enrichment method, as well as power plant technology and expected lifetime (load factor is usually assumed very high and does not vary significantly across plants). Back end processes also influence results to a lower extent.

5.3 Outlook

The work presented in this report aims at providing an overview of known environmental impacts of select electricity generating technologies. However, it is certainly not complete as a few gaps remain, both in data and methodology.

A first main challenge was to address **uncertainty** as required per the ISO 14040 series of standards. Due to resource constraints and a concern for a balanced output (it is necessary to provide uncertainty and sensitivity analyses for the whole set of technologies equally), this has not been carried out. Regionalisation brings variability in results, but this variability is known and inherent to local conditions, not to data (accuracy of collected input information) or model uncertainty (e.g. linearity assumption).

A need for **refining data** was identified during this work. Robust data was unavailable for potential leakage in CCS systems (yet a key challenge [49]), ionising radiation from non-nuclear technologies (see Box 5) with the partial exception of coal mining and combustion [2], and the characterisation of the criticality of novel materials such as rare earth metals (see Box 2). The proper accounting of land occupation has also arisen as a potential challenge, specifically in the case of wind power (methodological question of accounting for wind farm or turbine-only occupation), and hydropower (absence of water body characterisation in the impact modelling). The end-of-life treatment of renewable infrastructure has not been identified has a challenge, at least for regions where recycling infrastructure is to scale, but issues may arise regarding the potential complexity of wind turbine blades (inherent to the recycling of glass-fibre reinforced plastics) and PV cells (addressed in Box 3) –

processes for which more robust data is needed. Regarding the nuclear fuel cycle, further work is required on modelling closed-loop recycling of spent fuel (excluded from this exercise), and deep waste repository practices, as only Swedish data was accounted for – while repository strategies may differ significantly across regions in the future.

Proper system modelling would also include storage technologies, which are described in Box 4. To a large extent, storage requirements depend on the degree on renewable penetration in a grid, which makes the modelling relatively complex. It can be estimated that at the project level, adding storage to a PV system would increase lifecycle GHG emissions by 15%–45%, depending on battery chemistry and local conditions local conditions [83]. Finer modelling (relying on hourly data and fine load models) is required to assess storage need with a high accuracy.

Finally, many potential impacts of energy technologies are known but unquantifiable through a strict LCA approach. These aspects have been mentioned in technology-specific sections, they include acceptance, costs, aesthetic impacts, or biodiversity threats. Risks are excluded from LCA, as LCA only assess routine operations of a system. Risk analysis is a well-developed discipline that can inform decision-making with, in our case, analysing accidents from energy supply chains [119, 120].

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7 Annex

7.1 Short literature review of electricity generation portfolio assessments

Electricity systems have been explored thoroughly through the life cycle assessment lens. Challenges in phasing out fossil fuel power has been leading to developing abundant literature describing and analysing the environmental impacts of electricity-generating technologies [5, 11, 35, 121-125]. Regular reviews are proposed by the IPCC (AR5, SRREN). Harmonization efforts to summarize results on a fair comparison basis (e.g. identical lifetimes, load factors…) have been led by NREL [126]. A summary of the NREL findings is shown in Figure 55, specifically for lifecycle greenhouse gas emissions, as well as a comparison with the IPCC AR5 values [127] for reference. Data from [128] has been collected for a broader overview, available in the Annex (Figure 56). Studies also exist at the country scale, as shown by [128], who carried out a comprehensive assessment of available technology in the policy, historical, geographical… context of Switzerland. More recently, finer analyses have also been proposed to account for regional variability or future changes in the energy and industrial systems [129] or for their full-scale deployment at the global level [84].

A general conclusion of the existing literature is that, with rare exceptions, renewable technologies show lifecycle GHG emissions one order of magnitude lower than fossil-based technologies (10-100 instead of 100-1000 g CO_2 eq./kWh), principally embodied in infrastructure. Nuclear power, neither renewable nor fossil in nature, shows very low emissions due to the energy density of nuclear fuel and the absence of any combustion for electricity generation. Biopower's lifecycle GHG emissions may vary significantly depending on its feedstock, as purpose-grown crops may yield significantly higher emissions than residual waste from forestry activities. Hydropower can offer very low GHG scores, which may however be partially offset by sedimentation of organic matter in reservoirs, releasing (biogenic) GHG.

Compared with fossil-fuelled electricity, a few impact categories show higher results with renewable power plants. A first concern often raised is material intensity – not only in terms of bulk materials [47] but potentially specialty materials [24]. Second, land use is another challenge for ground-mounted technologies such as concentrated solar power or utility-scale photovoltaics. To a lesser extent, wind power and biomass projects may also lead to significant land occupation, depending on how "occupation" is accounted for wind power plant (see section 3.2), and on the biomass feedstock, respectively. Biomass may indeed require substantial amounts of land if using purpose-grown crops, which can be reduced by using residues from forestry (same conclusion as for GHG emissions in the previous paragraph). This technology however still relies on combustion, which generates potential emissions of particulate matter and nitrogen oxides, contributing to photochemical ozone creation.

Prospective exercises show that low-carbon electricity technologies can contribute to mitigating GHG emissions globally to reach climate targets, if deployed fast enough, together with proper storage technologies, and grid reinforcement [84]. Different pathways can lead society to decarbonising the global grid in time in compliance with 2°C scenarios – yet none is without potential adverse effects, be they on land use, materials, or water stress, to name a few.

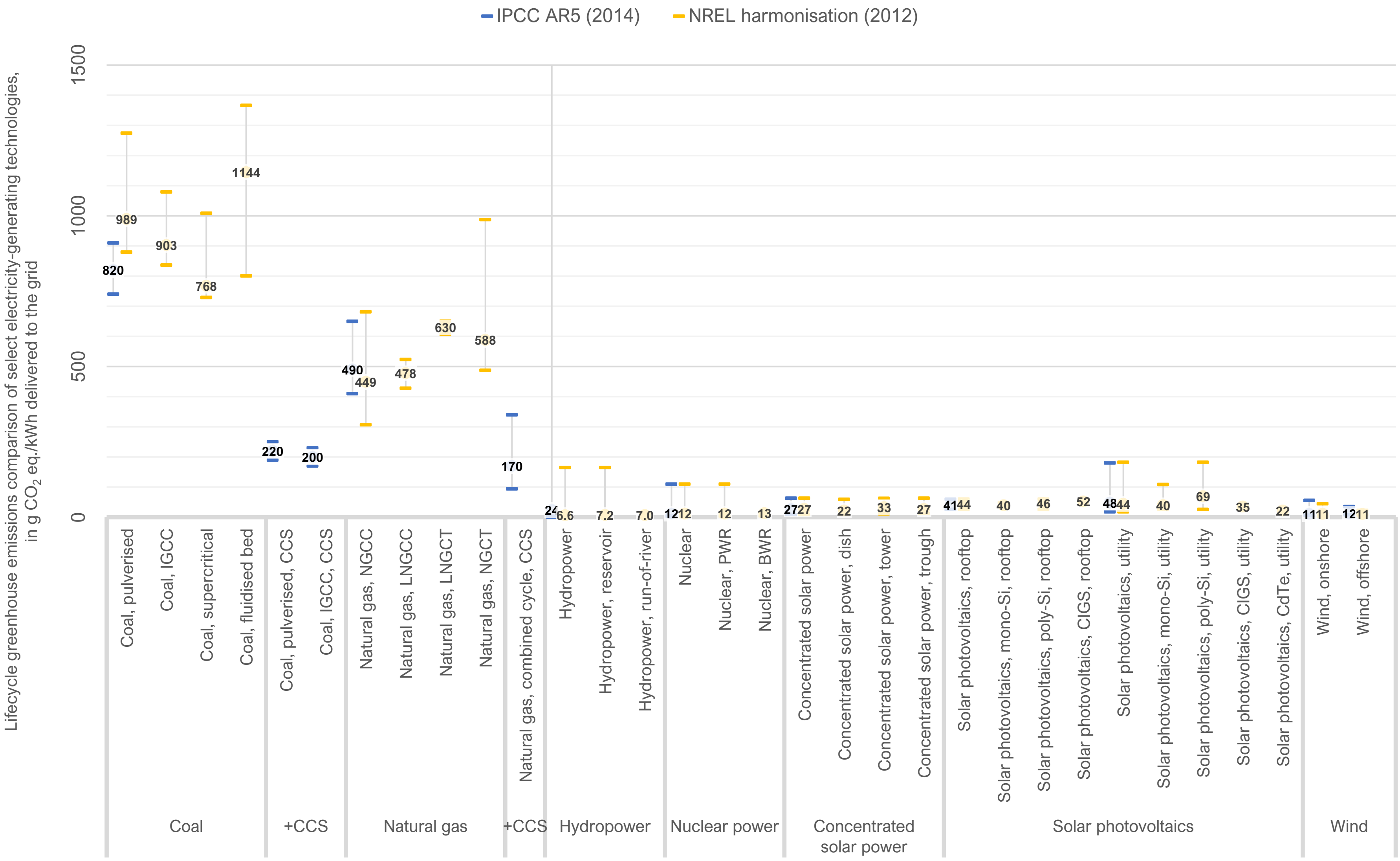

Figure 55. Lifecycle GHG emissions from electricity generation technologies, based on IPCC AR5 (2014) and the NREL harmonisation project (2012).

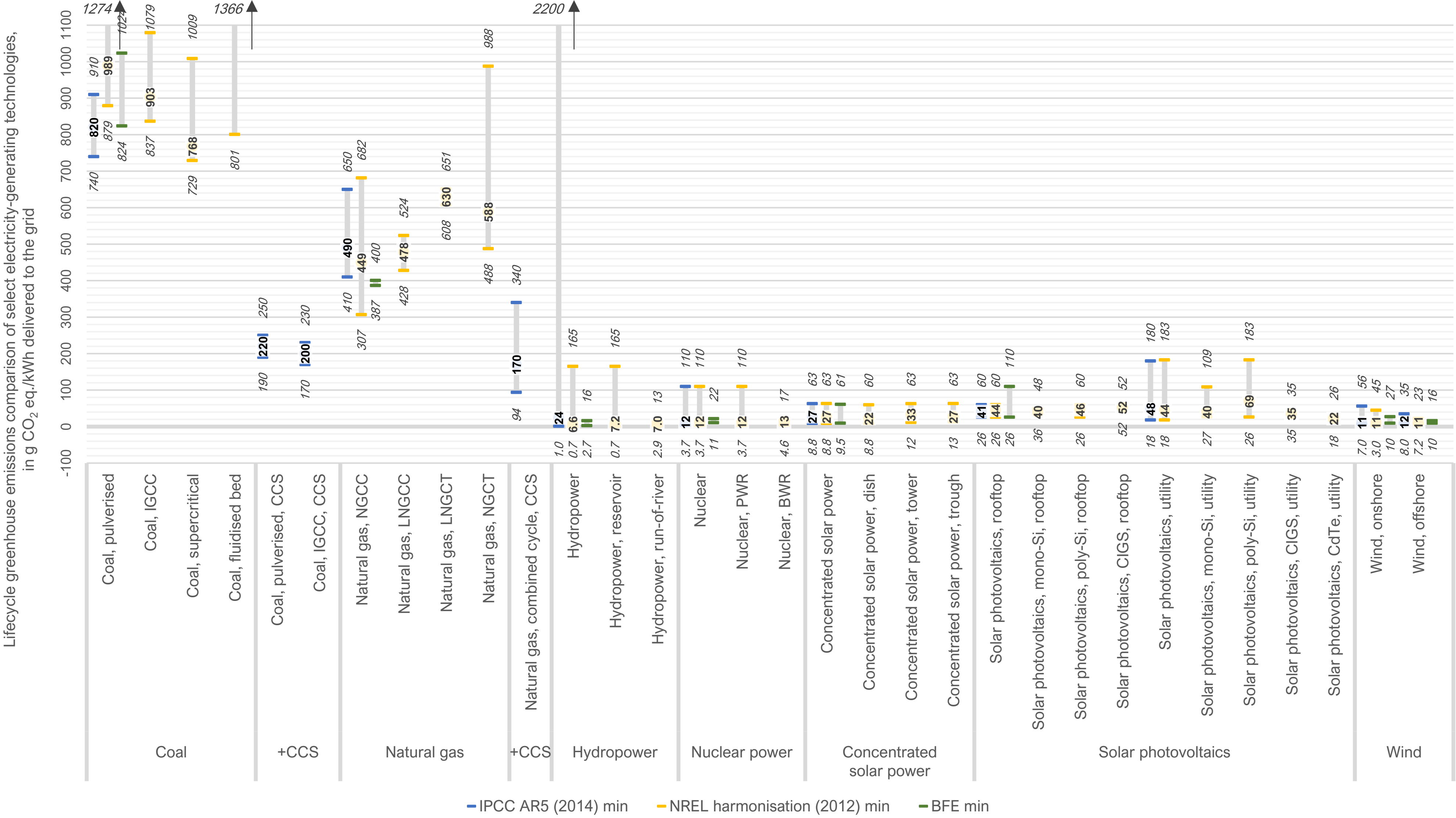

Figure 56. GHG values for electricity-generating technologies from [126-128].

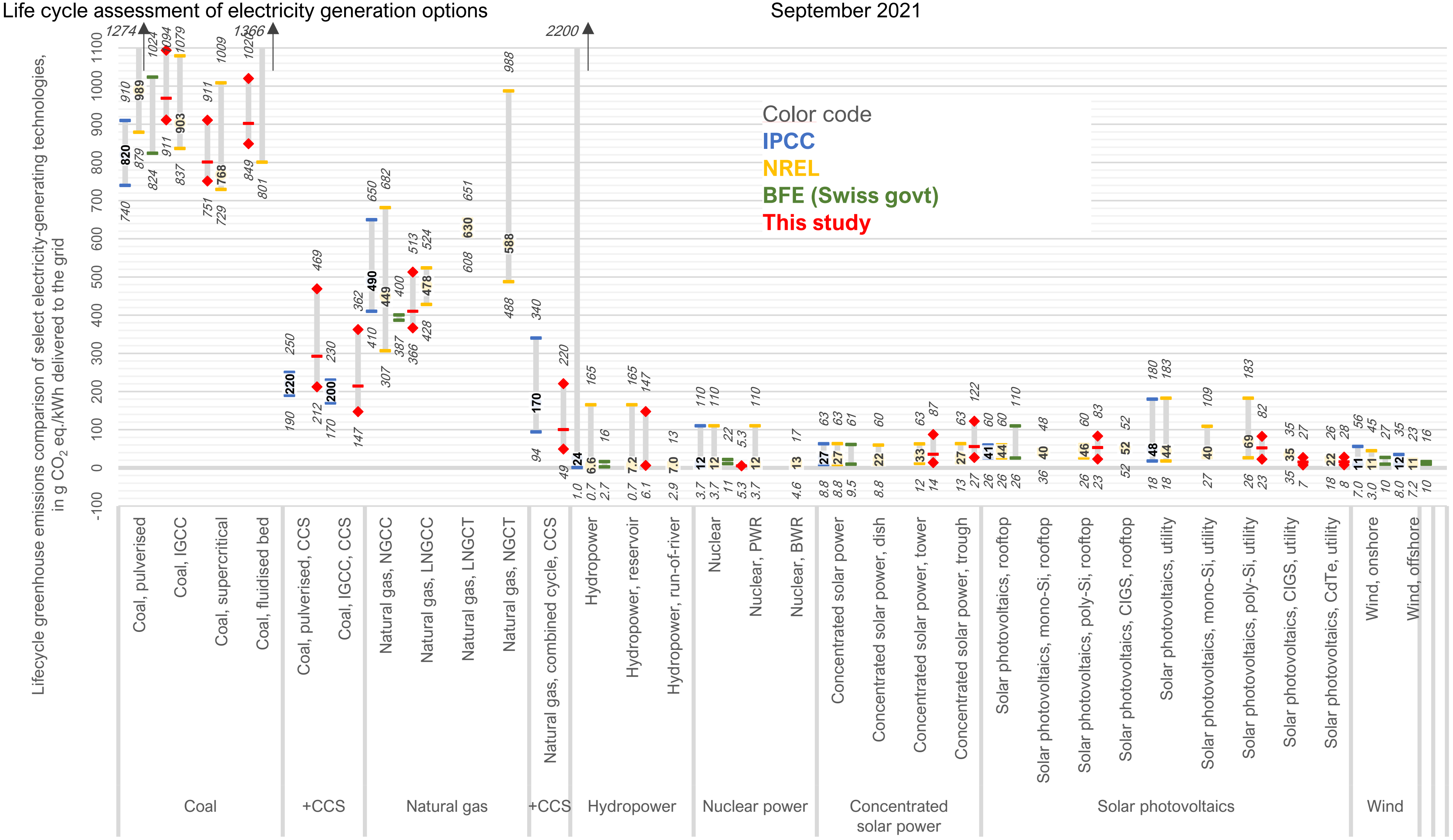

Figure 57. GHG values for electricity-generating technologies from [126-128] and this study.

7.2 Additional results

7.2.1 Full results as formatted tables

Table 13. LCIA results for region EUR (Europe EU28), per kWh, in 2020, for select indicators, rounded to two significant figures.

Per kWh		Climate change g CO_2 eq.	Freshwater eutrophication mg P eq.	Carcinogenic effects µCTUh	Ionising radiation g ^{235}U eq.	Land use points	Dissipated water l	Minerals and metals µg Sb eq.
Hard coal	PC, without CCS	1000	490	7.3	9.1	2.4	2.9	520
Hard coal	IGCC, without CCS	850	420	6.4	7.5	2.1	1.7	590
Hard coal	SC, without CCS	950	460	6.9	8.2	2.3	2.6	500
Natural gas	NGCC, without CCS	430	20	1.3	9.2	0.2	1.2	240
Hard coal	PC, with CCS	370	690	10	13	3.4	5.1	780
Hard coal	IGCC, with CCS	280	570	8.6	10	2.8	2.7	690
Hard coal	SC, with CCS	330	640	9.7	12	3.2	4.6	740
Natural gas	NGCC, with CCS	130	24	1.7	11	0.24	2.00	310
Hydro	660 MW	150	13	2.6	12	2.5	0.37	610
Hydro	360 MW	11	1.3	0.35	0.84	0.21	0.039	61
Nuclear	average	5.1	5.8	0.51	14	0.058	2.4	330
CSP	tower	22	11	2.1	4.5	3.6	0.18	340
CSP	trough	42	14	6.3	6.1	3.5	0.34	650
PV	poly-Si, ground-mounted	37	28	4.1	9.1	1.9	0.58	4500
PV	poly-Si, roof-mounted	37	39	1.6	9.8	0.86	0.63	7200
PV	CdTe, ground-mounted	12	8.8	3.4	1.9	1.4	0.13	1500
PV	CdTe, roof-mounted	15	14	1.1	1.9	0.15	0.16	2600
PV	CIGS, ground-mounted	11	8.8	3.4	1.8	1.3	0.13	1700
PV	CIGS, roof-mounted	14	14	1.1	1.8	0.15	0.16	2800
Wind	onshore	12	6.7	6.6	1.0	0.11	0.18	680
Wind	offshore, concrete foundation	14	7.0	5.5	1.2	0.11	0.16	980
Wind	offshore, steel foundation	13	6.8	7	1.2	0.099	0.16	990

Table 14. LCIA results for region EUR (Europe EU 28), in 2020, all ILCD 2.0 indicators, three significant figures[10]. Climate change (total) in bold. ***TO UPDATE***

Per kWh		Climate change biogenic	Climate change fossil	Climate change land use and land use change	**Climate change total**	Freshwater and terrestrial acidification	Freshwater ecotoxicity	Freshwater eutrophication	Marine eutrophication	Terrestrial eutrophication	Carcinogenic effects	Ionising radiation	Non-carcinogenic effects	Ozone layer depletion	Photochemical ozone creation	Respiratory effects, inorganics	Dissipated water	Fossils	Land use	Minerals and metals
		kg CO_2-Eq	kg CO_2-Eq	kg CO_2-Eq	**kg CO_2-Eq**	mol H+-Eq	CTU	kg P-Eq	kg N-Eq	mol N-Eq	CTUh	kg U235-Eq	CTUh	kg CFC-11.	kg NMVOC-.	disease i.	m3 water-.	megajoule	points	kg Sb-Eq
Hard coal	PC, without CCS	6.87E-05	1.02E+00	1.67E-04	**1.02E+00**	1.73E-03	4.72E-01	4.89E-04	5.14E-04	4.97E-03	7.34E-09	8.74E-03	1.14E-07	1.04E-08	1.25E-03	2.51E-08	1.23E-01	1.41E+01	2.43E+00	5.25E-07
Hard coal	IGCC, without CCS	5.38E-05	8.49E-01	1.40E-04	**8.49E-01**	1.05E-03	3.46E-01	4.24E-04	4.18E-04	4.00E-03	6.43E-09	7.47E-03	9.57E-08	8.74E-09	9.78E-04	1.36E-08	7.23E-02	1.21E+01	2.06E+00	5.89E-07
Hard coal	SC, without CCS	6.45E-05	9.53E-01	1.56E-04	**9.53E-01**	1.63E-03	4.33E-01	4.58E-04	4.82E-04	4.69E-03	6.90E-09	8.19E-03	1.06E-07	9.76E-09	1.16E-03	2.36E-08	1.12E-01	1.32E+01	2.28E+00	5.00E-07
Natural gas	NGCC, without CCS	7.78E-05	4.34E-01	8.21E-05	**4.34E-01**	3.26E-04	1.16E-01	1.97E-05	4.96E-05	7.49E-04	1.33E-09	9.24E-03	7.49E-09	6.66E-08	2.25E-04	1.33E-09	5.02E-02	7.86E+00	1.95E-01	2.43E-07
Hard coal	PC, with CCS	1.06E-04	3.68E-01	2.47E-04	**3.69E-01**	1.80E-03	8.26E-01	6.90E-04	7.29E-04	6.82E-03	1.04E-08	1.32E-02	1.66E-07	1.57E-08	1.68E-03	2.93E-08	2.18E-01	2.00E+01	3.45E+00	7.83E-07
Hard coal	IGCC, with CCS	7.23E-05	2.79E-01	1.89E-04	**2.79E-01**	1.35E-03	4.94E-01	5.71E-04	5.36E-04	5.10E-03	8.62E-09	1.01E-02	1.30E-07	1.18E-08	1.25E-03	1.72E-08	1.16E-01	1.63E+01	2.77E+00	6.85E-07
Hard coal	SC, with CCS	9.90E-05	3.33E-01	2.34E-04	**3.33E-01**	2.25E-03	7.51E-01	6.37E-04	6.92E-04	8.93E-03	9.66E-09	1.23E-02	1.53E-07	1.49E-08	1.55E-03	3.13E-08	1.98E-01	1.84E+01	3.18E+00	7.43E-07
Natural gas	NGCC, with CCS	9.39E-05	1.28E-01	9.93E-05	**1.28E-01**	6.07E-04	2.34E-01	2.40E-05	7.42E-05	1.87E-03	1.67E-09	1.11E-02	1.30E-08	7.81E-08	2.70E-04	3.14E-09	8.59E-02	9.26E+00	2.40E-01	3.14E-07
Hydro	660 MW	5.32E-05	1.47E-01	1.09E-04	**1.47E-01**	4.15E-04	3.97E-01	1.26E-05	9.54E-05	1.04E-03	2.56E-09	1.16E-02	2.17E-08	3.40E-08	3.85E-04	9.45E-09	1.58E-02	2.24E+00	2.45E+00	6.06E-07
Hydro	360 MW	1.80E-05	1.07E-02	9.21E-06	**1.07E-02**	4.45E-05	2.73E-02	1.33E-06	1.23E-05	1.43E-04	3.54E-10	8.40E-04	1.39E-09	2.37E-09	4.30E-05	8.07E-10	1.66E-03	1.63E-01	2.11E-01	6.06E-08
Nuclear	average	2.56E-05	5.24E-03	2.26E-05	**5.29E-03**	4.28E-05	2.70E-02	6.45E-06	8.20E-05	9.70E-05	5.51E-10	1.43E-02	5.50E-09	4.62E-10	2.65E-05	2.21E-09	1.31E-01	1.64E+01	6.25E-02	3.33E-07
CSP	tower	3.02E-05	2.16E-02	3.36E-05	**2.17E-02**	9.24E-05	3.65E-02	1.11E-05	2.21E-05	2.46E-04	2.09E-09	4.46E-03	2.61E-09	2.69E-09	7.54E-05	8.82E-10	7.60E-03	3.91E-01	3.62E+00	3.36E-07
CSP	trough	4.57E-05	4.19E-02	5.60E-05	**4.20E-02**	1.51E-04	1.10E-01	1.38E-05	2.88E-05	3.61E-04	6.25E-09	6.12E-03	4.61E-09	5.61E-09	1.05E-04	1.86E-09	1.47E-02	6.88E-01	3.54E+00	6.45E-07
PV	poly-Si, ground-mounted	3.43E-04	3.62E-02	1.51E-04	**3.67E-02**	3.01E-04	7.91E-02	2.84E-05	4.62E-05	4.48E-04	4.12E-09	9.14E-03	7.83E-09	6.97E-09	1.30E-04	2.21E-09	2.49E-02	6.43E-01	1.87E+00	4.45E-06
PV	poly-Si, roof-mounted	3.34E-04	3.67E-02	1.69E-04	**3.72E-02**	3.34E-04	6.99E-02	3.93E-05	5.12E-05	5.10E-04	1.63E-09	9.76E-03	1.38E-08	7.18E-09	1.43E-04	2.31E-09	2.72E-02	6.64E-01	8.57E-01	7.21E-06
PV	CdTe, ground-mounted	8.86E-05	1.18E-02	2.54E-05	**1.19E-02**	6.27E-05	5.59E-02	8.75E-06	1.27E-05	1.39E-04	3.44E-09	1.86E-03	3.67E-09	1.03E-09	4.16E-05	6.40E-10	5.63E-03	1.83E-01	1.39E+00	1.53E-06
PV	CdTe, roof-mounted	5.59E-05	1.45E-02	4.38E-05	**1.46E-02**	8.82E-05	3.96E-02	1.42E-05	1.54E-05	1.73E-04	1.14E-09	1.89E-03	7.46E-09	9.49E-10	4.86E-05	7.68E-10	7.05E-03	2.20E-01	1.48E-01	2.64E-06
PV	CIGS, ground-mounted	8.58E-05	1.13E-02	2.52E-05	**1.14E-02**	6.11E-05	5.58E-02	8.76E-06	1.25E-05	1.36E-04	3.39E-09	1.75E-03	3.77E-09	9.91E-10	4.08E-05	6.20E-10	5.64E-03	1.75E-01	1.35E+00	1.66E-06
PV	CIGS, roof-mounted	5.47E-05	1.40E-02	4.33E-05	**1.41E-02**	8.64E-05	4.02E-02	1.42E-05	1.52E-05	1.71E-04	1.14E-09	1.79E-03	7.59E-09	9.10E-10	4.79E-05	7.48E-10	7.08E-03	2.12E-01	1.47E-01	2.81E-06
Wind	onshore	1.87E-05	1.24E-02	1.99E-05	**1.24E-02**	5.28E-05	7.48E-02	6.67E-06	1.39E-05	1.26E-04	6.56E-09	1.03E-03	2.98E-09	6.71E-10	4.63E-05	7.06E-10	7.52E-03	1.75E-01	1.08E-01	6.75E-07
Wind	offshore, concrete foundation	1.74E-05	1.42E-02	2.58E-05	**1.42E-02**	1.00E-04	6.62E-02	6.98E-06	2.84E-05	2.93E-04	5.52E-09	1.19E-03	3.17E-09	1.24E-09	8.99E-05	6.57E-10	6.74E-03	1.97E-01	1.11E-01	9.77E-07
Wind	offshore, steel foundation	1.87E-05	1.33E-02	2.46E-05	**1.33E-02**	9.45E-05	7.94E-02	6.84E-06	2.69E-05	2.76E-04	7.00E-09	1.19E-03	3.41E-09	1.18E-09	8.44E-05	6.19E-10	6.67E-03	1.90E-01	9.94E-02	9.93E-07

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[10] Results should not be considered robust to the third significant figure, for information only.

7.2.2 Land use results from ReCiPe method

To facilitate interpretation, Figure 58 shows land occupation in m2-annum (1 square meter occupied over 1 year).

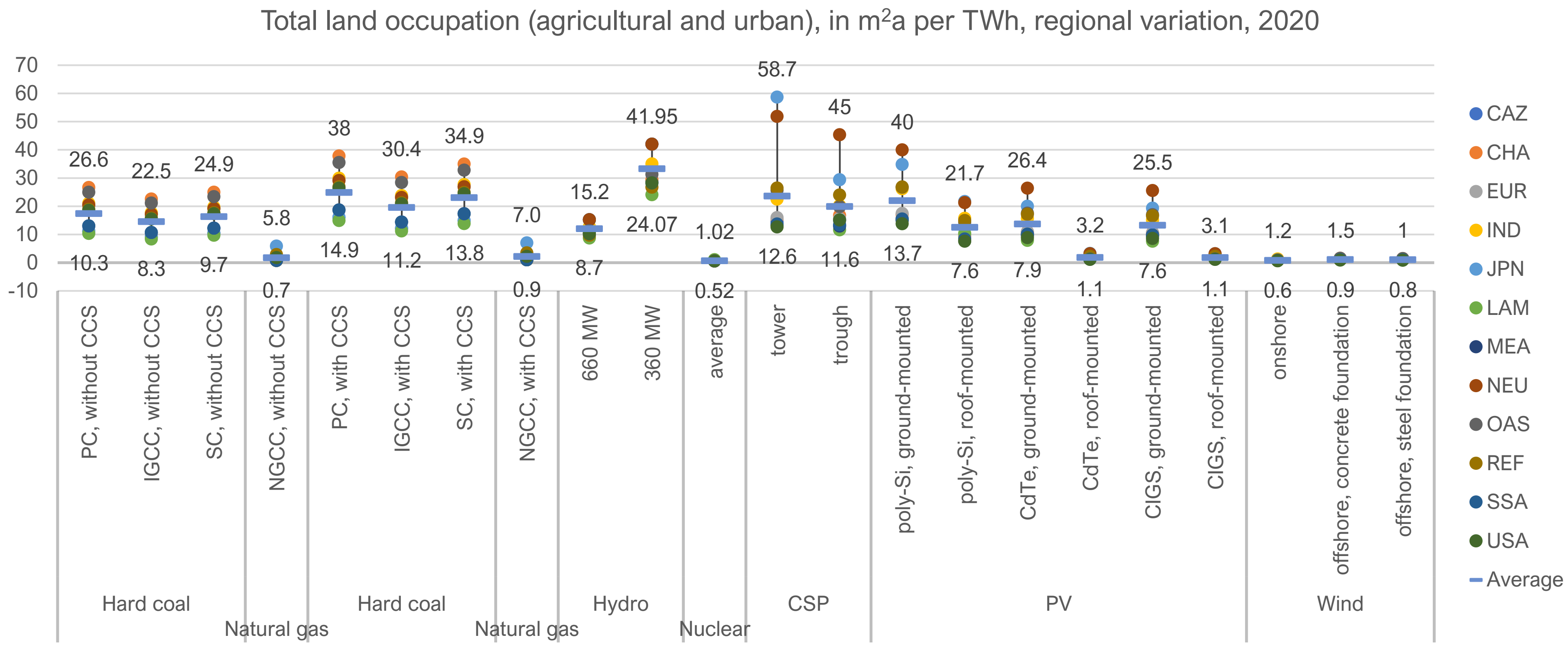

Figure 58. Lifecycle land use regional variations for year 2020. Variability is explained by several factors: electricity mix (all regions), origin of supply (fossil fuels), load factors (renewables). Nuclear power is modelled as a global average and therefore does not see any variation.

7.3 Nuclear power life cycle inventories

Nuclear power has been subject to a consultation process with the World Nuclear Association in order to build new life cycle inventories for the front-end, core, and back-end processes of the nuclear life cycle. Significant changes have been brought regarding the mining & milling, and spent fuel management, which reflects recent changes in the nuclear power industry.

Throughout this section, only inputs are indicated – emissions (of greenhouse gases, radionuclides, and other emissions are available in the full life cycle inventory file).

7.3.1 Uranium mining and milling

This step consists in the extraction of raw uranium from the ground, the ore milling, ending with the production of uranium oxide (or yellowcake), on site. Uranium is mined from surface or from underground. Globally, the study assumed that to produce electricity from nuclear power approximately 68% of uranium production is derived from surface mines and approximately 32% of uranium production is derived from underground mines.

Historically, the two main techniques used for uranium extraction are open pit and underground mining – depending on the depth of the ore. The market share of *in-situ leaching* (ISL), has been gradually increasing over the last decades – up to about half of all uranium extracted annually as of 2014. The fastest growth in ISL extraction has been in Kazakhstan, but other projects have started operation in Australia, China, Russia and Uzbekistan. Other production methods exist, namely "co-product" recovery from copper, gold and phosphate extraction, or heap and in-place leaching. These methods are more anecdotal and will be excluded from the present study. ISL involves leaving the ore physically undisturbed and recovering minerals from it by dissolving them in a solution, often sulphuric acid before pumping that to the surface where the minerals can be recovered. Consequently, there is little surface disturbance and no tailings or waste rock generated.

Mining extracts uranium from the uranium-containing ore deposit using a method that is appropriate to the geological conditions of the deposit and ensures the health and safety of workers and the public and protection of the environment. Ore grade may vary significantly between deposits / ore bodies that are mined, from <0.01% to >20%. Milling includes crushing and grinding the ore, separating the uranium from the rest of the rock, as well as further steps of refinement and purification. At this stage, the main uranium product is known as "yellowcake", a common name for uranium oxide (U_3O_8), the naturally occurring form of uranium. After milling, yellowcake is then transported to a conversion facility and the tailings are stored in a final repository. Milling tailings are notoriously the main source of radioactive emissions over the nuclear fuel cycle, as they are assumed to release 35 TBq/kg U_{nat} over 80000 years as reported in [27] a value reused in [130].

We assume natural attenuation instead of active remediation of site. Tests have been carried out at the Irkol deposit in Kazakhstan, showing that in "four years the ISL-affected area had reduced by half, and after 12 years it was fully restored naturally." More densely populated area require that groundwater be restored to baseline standards, and newer mines even include a water restoration circuit by design [131].

Globally, **we assume that 14% of all primary[11] uranium comes from open pit mines, 32% from underground, and 55% from in-situ leaching**. This assumption is valid over the 2016-2020 period and based on WNA global data. Co-product recovery is not accounted for, although it accounts for a few percentage points of the global supply – neglecting it is therefore a conservative assumption, as allocation rules would lead to calculating reduced impacts from uranium being a by-product from a larger multi-output process. Furthermore, almost all of co-product extraction occurs at a single polymetallic mine in South Australia, Olympic Dam, which revenue originates mostly from copper,

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[11] Uranium requirements are met essentially from primary uranium – extracted from the ground – but also from secondary resources – inventories, re-enrichment of depleted uranium, recycled uranium. warheads dismantling Those resources had been mined in any case and would represent less than 15% of the total yearly uranium requirements – for sake of simplification, this LCA considers only the equivalent primary production to meet the worldwide demand of all nuclear power reactors.

followed by uranium, silver and gold. The specificities of Olympic Dam are not considered representative enough with respect to the global mining mix – and allocating its environmental impacts to co-products for the building of life cycle inventories would require further analysis.

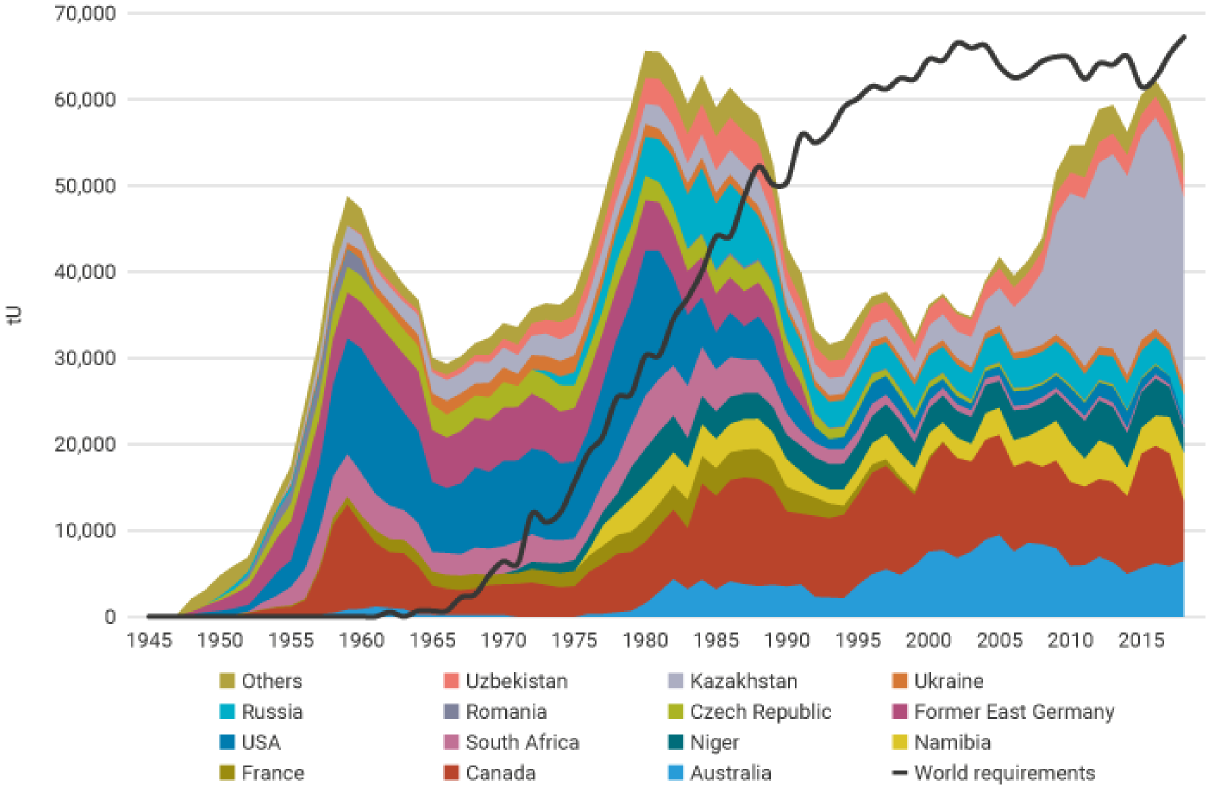

Figure 59. World primary uranium production and reactor requirements, in tonnes uranium. Source: [132]

The generic ecoinvent 3.7 dataset was considered for uranium ore underground mining and milling. Data is representative of US operation modes in the early 1980. It was compared to the Life Cycle Inventory data from Parker et al. (2016), which are representative of a weighted average between two underground mines (ore grade 0.74 and 4.53%), and one surface mine (ore grade 1.54%)in northern Saskatchewan, between 2006 and 2013 for two of them, and between 1995 and 2010 for the third one. However, the respective inventories present large disparities, limiting the possibility of comparison. Ecoinvent specifies the major harm from the uranium ore extraction (underground or open pit) and treatment is from milling, hence a lower priority was given to the characterisation of underground mining inventory which remain empty in terms of chemicals used (Table 19). Also, as shown in Table 19 the range of chemicals considered in ecoinvent dataset for milling does not include hydrogen peroxide, a main chemical used in the inventory from Parker et al. 2016 – although it includes a generic input of “chemicals, organic”. The consumption of energy is also disparate. The dataset from Parker et al. (2016) accounts for electricity consumption, as the specific mine is grid-connected, unlike the ecoinvent model mine. Last, ecoinvent accounts for heat inputs (more than 3 times higher than electricity requirements from Parker) generated from fuel oil, hard coal and wood chips, while Parker et al. (2016) lists diesel, gasoline, and propane as inputs.

The ecoinvent 3.7 LCI dataset representative of uranium in yellow cake from uranium mining through ISL seems incomplete. Indeed, ecoinvent specified that no consideration of chemical mining was attempted due to the high variety of geological conditions and the few literature available on the related environmental impacts. The partially complete inventory from Haque et al. (2014) is given in Annex (section 7.2.2) as indicative. It is representative of ISL practice in Australia for the early 2010, uranium ore grade 0.24%. High variations are observed between ecoinvent dataset and that of Haque et al. (2014), for sulphuric acid, diesel and water consumption. The inventory from Parker et al. (2016) and

Haque et al. (2014) do not quantify the direct emissions released into air, water and soil during mining and milling operations, while it is available in the ecoinvent datasets.

Table 15. Inputs for surface, open pit mining, per kg of uranium in ore.

Inputs	Amount	Unit	Comment
blasting	1.52	kg	WNA consultation
diesel, burned in building machine	12.2	MJ	WNA consultation
diesel, burned in diesel-electric generating set, 10MW	293.9	MJ	WNA consultation
mine infrastructure construction, open cast, uranium	6.17E-08	unit	ecoinvent assumption

Table 16. Inputs for underground mining, per kg of uranium in ore.

Inputs	Amount	Unit	Comment
blasting	0.29	kg	WNA consultation
diesel, burned in diesel-electric generating set, 10MW	133.4	MJ	WNA consultation
heat, district or industrial, other than natural gas	247.5	MJ	WNA consultation
electricity, medium voltage	68.1	MJ	WNA consultation
mine infrastructure, underground, uranium	2.78E-07	unit	ecoinvent assumption

Table 17. Inputs for surface mining, in-situ leaching, per kg of U in yellowcake.

Inputs	Amount	Unit	Comment
ammonium nitrate	2.5	MJ	WNA consultation
electricity, medium voltage	43.4	kg	WNA consultation
diesel, burned in diesel-electric generating set, 10MW	32.95	kg	WNA consultation
petrol, unleaded, burned in machinery	4.1	kg	WNA consultation
heat, central or small-scale, other than natural gas	103.9	kg	WNA consultation
steel, chromium steel 18/8	0.108	kg	ecoinvent assumption
sulfuric acid	65.5	kg	WNA consultation
water, decarbonised	173.2	kg	WNA consultation
hydrogen peroxide, without water, in 50% solution state	0.61	kg	Haque et al. (2014)
phosphoric acid, industrial grade, without water, in 85% solution state	0.23	kg	Haque et al. (2014), D2EHPA
hydrochloric acid, without water, in 30% solution state	0.03	kg	Haque et al. (2014)
sodium bicarbonate	0.3	Kg	Haque et al. (2014)
sodium hydroxide, without water, in 50% solution state	1.37	kg	Haque et al. (2014)
sodium chlorate, powder	8.21	kg	Haque et al. (2014)

Table 18. Inputs for milling, per kg of uranium in yellowcake.

Inputs	Amount	Unit	Comment
Electricity, medium voltage	22.5	kWh	WNA consultation
Tailing, from uranium milling	-0.25	m^3	ecoinvent assumption
Sulfuric acid	55	kg	WNA consultatio
Diesel, burned in diesel-electric generating set, 10MW	57	kg	WNA consultatio
Uranium mine operation, open cast, WNA	30%	kg	WNA consultation
Uranium mine operation, underground, WNA	70%	kg	WNA consultation

Box 6. Ore grade

Mining impacts are technically highly dependent on ore grade, as the efforts required to extract a fixed quantity of ore is proportional to the amount of rock to be extracted, therefore inversely proportional to the grade. This is true at the individual mine level, for which such a model could be derived; more importantly, this assumption is valid for open pit and underground mines. Warner and Heath [133] test this relationship and its influence over the full life cycle of the technology, showing that a lowering ore grade may lead to tripling lifecycle GHG emissions by 2050 in case of a sustained growth of installed nuclear capacity (assuming that primary uranium remains the main source up to 2050). In the case where uranium is mined together with other elements, it is also plausible that energy inputs may be overestimated [134].

7.3.1.1 Mining inventories

Table 19: Life Cycle Inventory of uranium (underground & open pit) mining and milling

			Uranium ore underground mining and milling			
Chemicals	**Parker et al. 2016 - Weighted average for underground / open pit / raisebore mining + Milling**		**Ecoinvent3.7 - Uranium ore, as U [135]| uranium mine operation, underground | Cut-off, U**		**Uranium, in yellowcake [135]| production | Cut-off, U**	
Ammonia	0.404	kg/kg U3O8			0.9	kg/kg
Lime/Quicklime	2.91	kg/kg U3O8				
Hydrogen peroxide	0.202	kg/kg U3O8				
Diluent (kerosene)	n.a.	kg/kg U3O8				
D2EHPA (Di-(2-ethylhexyl)phosphoric acid)	n.a.	kg/kg U3O8				
Amine	n.a.	kg/kg U3O8				
TBP (tributyl phosphate)	n.a.	kg/kg U3O8				
Hydrochloric acid	n.a.	kg/kg U3O8				
Sodium carbonate	n.a.	kg/kg U3O8				
Sodium hydroxide	n.a.	kg/kg U3O8			0.026	kg/kg
Sulphuric acid	n.a.	kg/kg U3O8			35	kg/kg
Sodium chlorate	n.a.	kg/kg U3O8			1	kg/kg
Ammonium sulfate					0.106	kg/kg
Chemical inorganic					0.26	kg/kg
Chemical organic					0.315	kg/kg
Ethylenediamine					0.012	kg/kg
Soda ash					2.5	kg/kg
Sodium chloride					2.5	kg/kg
Other non chemical - for operation						
Bentonite						
Barite						
Blasting	0.0912	kg/kg U3O8	0.26	kg/kg ore		
Diesel	36.86	MJ/kg U3O8	300	MJ/kg ore	176	MJ/kg
Water			0.1	m3/kg ore	1	m3/kg
Electricity	22	kWh/kg U3O8				
Heat (other than gas)					250.8	MJ/kg

Table 20. Life Cycle Inventory of uranium (ISL) mining and milling

Chemicals	**Haque et al. 2014 - In situ leaching - Australia**		**Ecoinvent3.7 - Uranium, in yellowcake (GLO)| uranium production, in yellowcake, in-situ leaching | Cut-off, U**
Ammonia	-	kg/kg U3O8 as yellow cake	
Lime/Quicklime	-	kg/kg U3O8 as yellow cake	
Hydrogen peroxide	0.61	kg/kg U3O8 as yellow cake	
Diluent (kerosene)	0.88	kg/kg U3O8 as yellow cake	
D2EHPA (Di-(2-ethylhexyl)phosphoric acid)	0.23	kg/kg U3O8 as yellow cake	
Amine	0.23	kg/kg U3O8 as yellow cake	
TBP (tributyl phosphate)	0.23	kg/kg U3O8 as yellow cake	
Hydrochloric acid	0.03	kg/kg U3O8 as yellow cake	

Sodium carbonate	0.3	kg/kg U3O8 as yellow cake		
Sodium hydroxide	1.37	kg/kg U3O8 as yellow cake		
Sulphuric acid	7.87	kg/kg U3O8 as yellow cake	20.0	kg/kg
Sodium chlorate	8.21	kg/kg U3O8 as yellow cake		
Other non chemical - for operation				
Bentonite	0.08	kg/kg U3O8 as yellow cake		
Barite	0.21	kg/kg U3O8 as yellow cake		
Blasting				
Diesel	11.66	MJ/kg U3O8 as yellow cake	886.6	MJ/kg
Water			9.1229347	m3/kg
Electricity (pumping)	28	kWh/kg U3O8 as yellow cake		
Heat (other than gas)				

7.3.2 Conversion and enrichment

Conversion involves a series of processes aiming at producing uranium hexafluoride (UF_6), from yellowcake and other chemicals. Up to this stage, the share of uranium-235 (^{235}U) in the uranium product is about 0.7% (its natural abundance), with 99.2% of uranium-238 (^{238}U), the dominant, non-fissile, isotope, making up most of the rest of natural uranium. As the manipulation of gases is easier for enrichment, uranium atoms are combined with fluorine to produce UF_6, which sublimes at 56°C, a temperature that makes it usable as a stable gas for the subsequent step of enrichment. Yellowcake is first purified through a series of chemical processes: dissolution in nitric acid, solvent extraction, washing, and concentration by evaporation. The resulting solution is then calcined to produce uranium trioxide or dioxide. A reduction process is necessary to obtain pure UO_2. This UO_2 then reacts with gaseous hydrogen fluoride in a kiln to produce uranium tetrafluoride (UF_4), which finally reacts with gaseous fluorine (F_2) to produce uranium hexafluoride (UF_6). At this point, uranium is still made of about 0.7% of ^{235}U.

The global conversion market is shared between a few sites, we assume here that all plants are supplied by this global market, namely **from CNNC (China), Rosatom (Russia), Cameco (Canada), and Orano (France)** – another company, ConverDyn, represents 12% of global capacity but has been idle for several years [136]. The exact shares are not communicated in this report for confidentiality reasons. A main assumption is that all uranium converted over a year is used on the same year, which does not exactly reflect reality as stocks may be kept. We provide the conversion-specific electricity mix used in the model in Figure 61.

Table 21. Inputs for conversion, per kg UF_6 (non-enriched).

Inputs	Amount	Unit	Comment
Ammonia	0.25	kg	ecoinvent 3.7
Cement	0.81	kg	ecoinvent 3.7
Chemical, organic	0.03	kg	ecoinvent 3.7
Chemical, inorganic	0.052	kg	ecoinvent 3.7
Electricity, high voltage	11.8	kWh	From WNA consultation
Heat	26	MJ	From WNA consultation
Hydrogen fluoride	0.59	kg	ecoinvent 3.7
Nitric acid	0.9	kg	ecoinvent 3.7
Quicklime, milled, loose	0.5	kg	ecoinvent 3.7
Uranium, in yellowcake	1.04	kg	Global average estimate
Water, decarbonised	500	kg	ecoinvent 3.7

To start and sustain a chain reaction in a conventional nuclear reactor, the ^{235}U share must increase to 3–5%, which is achieved by the enrichment process. The vast majority of commercial enrichment process in use today is centrifugation, whereby the slightly heavier molecules of $^{238}UF_6$ are separated from the lighter $^{235}UF_6$ by rotating centrifuges at a very high speed. The process needs to be repeated multiple times, by cascading centrifuges, until the uranium element has reached the desired enrichment rate. Other techniques exist, for example gaseous diffusion, which also exploits the slight differences in UF6 molecules by forcing them through a membrane (much more energy-intensive than centrifugation), aerodynamic processes, or electromagnetic separation. Gaseous diffusion has been phased out globally in 2013. In addition to energy inputs required for the high-speed rotations of centrifuges, heat is also needed to keep UF6 in its gaseous state.

Conversion generates low-level radioactive waste, 90% of which is directed to interim storage, while 9% is incinerated (plasma torch) and 1% is surface or trench-deposited. The original ecoinvent model assume the same shares, with the plasma torch incineration being modelled on the Zwilag treatment plant in Würenlingen, Switzerland[12]. Radioactive emissions from the waste treatment were adjusted from 1.66 and 3.04 GBq/m3 of carbon-14 and tritium, respectively (1993 data) to 0.04 and 8.40 GBq/m3 (2017 data, from [137], assuming a constant throughput of waste, i.e. 5 m^3/year).

Globally, enriched uranium is supplied by roughly the same operators as for conversion, as reported in Table 22. All enrichment activity is assumed to use centrifuges, consuming a global average of 40 kWh/SWU, see Figure 60 for a comparison with existing studies. The weighted average electricity mix used for this process is shown in Figure 61.

Table 22. Global enrichment capacity as of 2018. Source: World Nuclear Association [138].

Operator	Region	Capacity (in SWU, 2018)	Market share
CNNC	China	6750	11%
Rosatom	Russia	28215	46%
Orano	France	7500	12%
Cameco	Canada	46	0%
Urenco	Netherlands, United Kingdom, Germany, United States	18600	30%

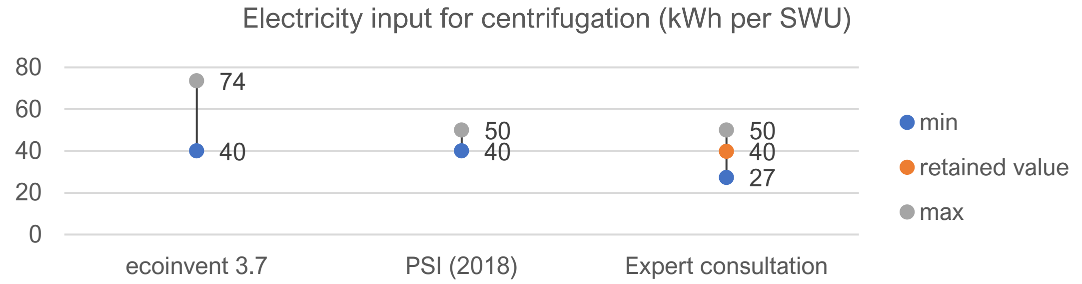

Figure 60. Review of electricity input value for the centrifugation step, in kWh per SWU of enriched uranium (see Box 7 for an explanation of that unit). Sources: ecoinvent 3.7, Zhang and Bauer [139], and consultation with WNA experts.

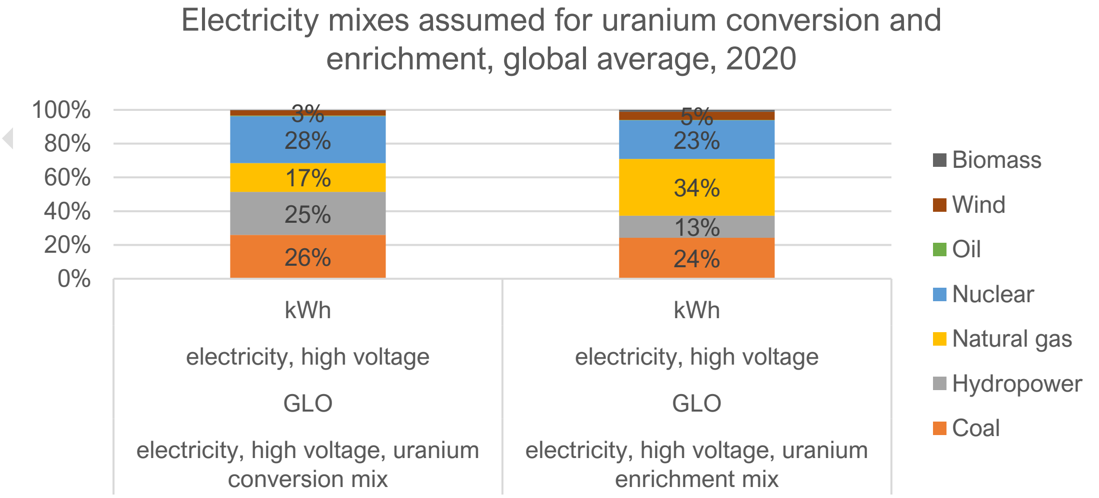

Figure 61. Electricity mixes specific to the conversion and enrichment of uranium, as a result of the weighted average of global suppliers as of 2019.

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[12] More details on the facility at https://www.zwilag.ch/en/function-of-facility-_content---1--1065.html

Table 23. Inputs for conversion, per kg UF_6 (non-enriched).

Inputs	Amount	Unit	Comment
acetylene	0.000025	kg	ecoinvent assumption
aluminium, wrought alloy	0.05	kg	ecoinvent assumption
argon, liquid	0.0018	kg	ecoinvent assumption
brass	0.0018	kg	ecoinvent assumption
chemical, organic	0.00082	kg	ecoinvent assumption
chemicals, inorganic	0.0311	kg	ecoinvent assumption
concrete, normal	0.00029	m3	ecoinvent assumption
diesel, burned in diesel-electric generating set, 10MW	1.28	MJ	ecoinvent assumption
Electricity, high voltage, uranium enrichment mix	**40.0**	**kWh**	**WNA consultation**
heat, district or industrial, natural gas	13.68	MJ	ecoinvent assumption
hydrochloric acid, without water, in 30% solution state	0.0002	kg	ecoinvent assumption
hydrogen peroxide, without water, in 50% solution state	0.00068	kg	ecoinvent assumption
hydrogen, liquid	0.000011	kg	ecoinvent assumption
low level radioactive waste	-0.00063	m3	ecoinvent assumption
lubricating oil	0.0092	kg	ecoinvent assumption
methanol	0.00032	kg	ecoinvent assumption
nitric acid, without water, in 50% solution state	0.0015	kg	ecoinvent assumption
nitrogen, liquid	0.00039	kg	ecoinvent assumption
oxygen, liquid	0.000036	kg	ecoinvent assumption
phosphoric acid, fertiliser grade, without water, in 70% solution state	0.00012	kg	ecoinvent assumption
polyvinylchloride, bulk polymerised	0.00087	kg	ecoinvent assumption
soap	0.00088	kg	ecoinvent assumption
sodium hydroxide, without water, in 50% solution state	0.0028	kg	ecoinvent assumption
spent anion exchange resin from potable water production	-0.058	kg	ecoinvent assumption
steel, low-alloyed, hot rolled	0.15	kg	ecoinvent assumption
uranium enrichment centrifuge facility	2.22E-08	unit	ecoinvent assumption
uranium hexafluoride, WNA	**1.20**	**kg**	**Global average (WNA 2019)**
waste mineral oil	-0.0024	kg	ecoinvent assumption
treatment of municipal solid waste, sanitary landfill	-0.235	kg	ecoinvent assumption

Box 7. Separative work units

Enrichment processes involve the separation of a feed of UF6 into two outputs with different $^{235}U/^{238}U$ isotope concentrations, the enriched product and the depleted tails. Depending on the feed assay (the original concentration), the desired enrichment rate and the tails assay, a centrifuge, or more likely an array thereof, will provide a variable amount of work. Following Glaser (2008), we write the mass balance of the enrichment process as:

$$\boldsymbol{FN_F = PN_P + WN_w}$$

We use the notations of Glaser (2008) where $\boldsymbol{F}$, $\boldsymbol{P}$, and $\boldsymbol{W}$ are the feed, product, and tails streams, typically in kg/year, and $\boldsymbol{N_X}$ are the respective fraction of the fissile material ^{235}U, in each stream. We define the *cut* $\boldsymbol{\theta}$ as the proportion of the feed exiting the process as product, i.e. $\boldsymbol{P = \theta F}$. It can be shown that the cut is dependent on the various rates $\boldsymbol{N_X}$, and is therefore fixed for a given configuration. The work (energy) needed to enrich or deplete a flow is defined through the function $\boldsymbol{V(N)}$, which obeys the following equation:

$$\boldsymbol{\delta U = PV(N_P) + WV(N_W) - FV(N_F)}$$

Where $\boldsymbol{\delta U}$ is the separative power for producing quantity P from quantity F. There is no exact analytical expression for $\boldsymbol{V(N)}$ but using Taylor series, its second derivative can be estimated, from which $\boldsymbol{V(N)}$ is given the standard expression:

$$\boldsymbol{V(N) = (2N - 1)\ln\left(\frac{N}{1 - N}\right)}$$

Combining the two latter equations, the amount of SWU per enriched material can be computed as $\boldsymbol{\frac{\delta U}{P}}$, which after simplification yields the following expression:

$$\boldsymbol{\frac{\delta U}{P} = SWU = V(N_P) - V(N_W) + \frac{N_P - N_W}{N_F - N_W}\big(V(N_W) - V(N_F)\big)}$$

This value is used in the life cycle inventories.

A few examples:

- 1 kg UF6 at $\boldsymbol{N_P} = \mathbf{3.8}\%$ and $\boldsymbol{N_W} = \mathbf{0.20}\%$ tails assay requires 6.09 SWU, from 7.05 kg feed,
- 1 kg UF6 at $\boldsymbol{N_P} = \mathbf{5.0}\%$ and $\boldsymbol{N_W} = \mathbf{0.25}\%$ tails assay requires 7.92 SWU, from 10.3 kg feed.

Depending on the actual technique used, the energy value of a SWU can span from about 40 kWh/SWU for gas centrifugation, to more than 2 MWh/SWU in gas diffusion techniques. Most of diffusion facilities have now been retired, all enrichment in this study is considered performed via gas centrifugation.

7.3.3 Fuel fabrication

Fuel fabrication is the main step remaining before fissile uranium is ready to be used in a reactor. The enriched UF_6 is here transformed into uranium dioxide (UO_2), first as powder, and then in a format adapted to the reactor design, usually as small pellets. These pellets are ultimately piled up in long rods made of zirconium alloy that, once in place in the reactor, are at the heart of the chain reactions.

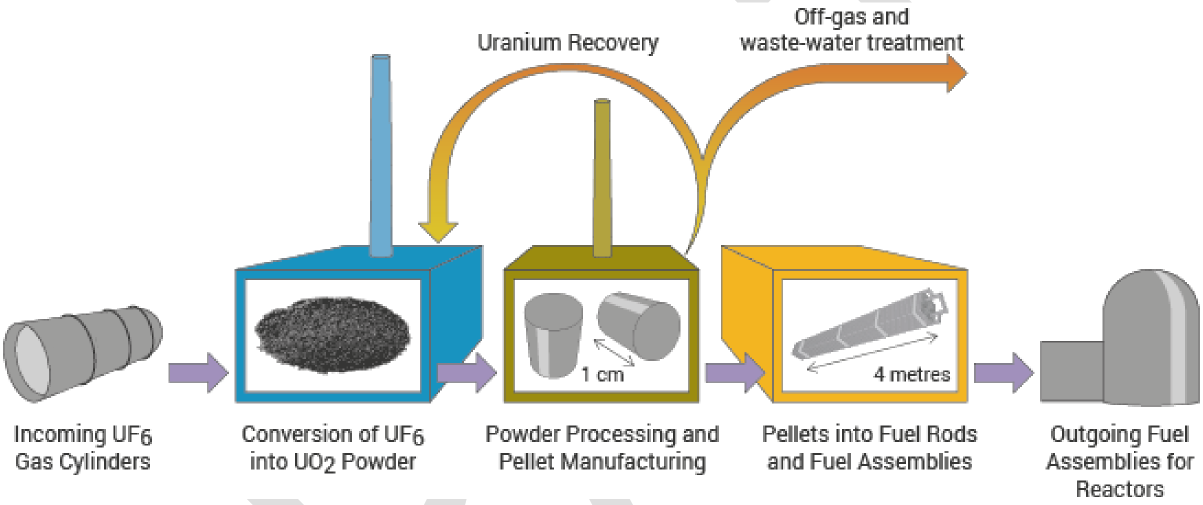

Figure 62. Fuel fabrication process. Source: World Nuclear Association [140]

The three main steps of fuel fabrication are: the powder conversion, which can be done either through a “wet” (using water and drying the slurry) or “dry” process (with steam), the pellet manufacturing (using a high temperature furnace), and the assembly. All these steps require significant energy inputs, reported in ecoinvent 3.7 as 36 kWh of electricity and 30 MJ of heat. Consultation with WNA experts show that electricity inputs could possibly reach **50 kWh per kg U** in fuel elements – which is the value retained for this LCA.

Table 24. Inputs for fuel fabrication, per kg fuel element.

Inputs	Amount	Unit	Comment
Cement	0.0065	kg	ecoinvent 3.7
Chromium	0.6	kg	ecoinvent 3.7
Electricity, medium voltage	50	kWh	From WNA consultation
Uranium, enriched, per SWU	6.74	SWU	See mass balance calculation
Water, decarbonised	300	kg	ecoinvent 3.7

7.3.4 Power plant construction

This step covers the processes of development, site preparation, construction of reactors, and infrastructure, as well as connection to the grid. The amount and variety of materials for a power plant construction is significant, inventory modelling is therefore done through collecting high-level data. Sources include both official documentation from NPP operators, but also estimates based on blueprints, whereby authors provide rough methods to calculate the total amount of bulk materials in a NPP from drawings. Such estimates carry high uncertainty, which leads to a significant variability in results, as seen in Figure 63**Error! Reference source not found.**. Bulk material requirements for the construction a NPP vary significantly from source to source also because of the multiple designs possible. For the current exercise, we retain average values (in magenta on Figure 63).

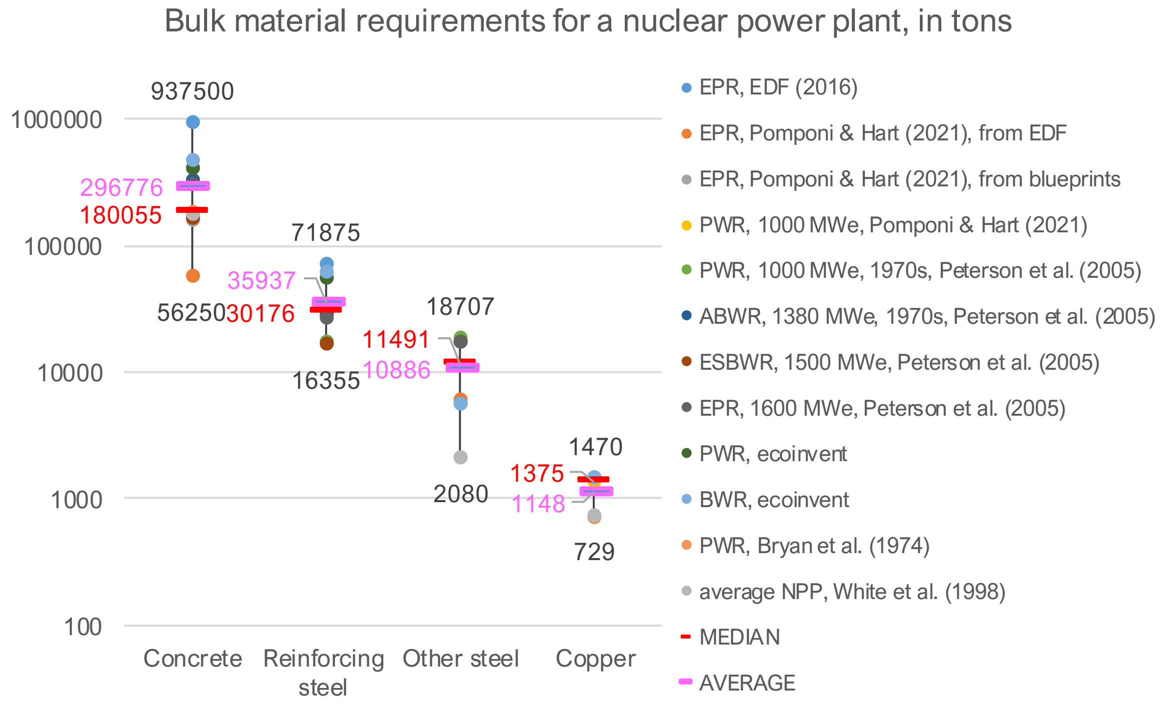

Figure 63. Bulk material requirements for the construction of a nuclear power plant, scaled to 1000 MWe, based on official documentation from EDF and various estimates made in the academic and grey literature. Concrete is usually given in volume, a density of 2.4 t/m³ was assumed for conversion. Sources: [141-144], and ecoinvent database.

Construction does not only require materials; the amount of energy and chemical inputs is also significant. Electricity, diesel, and heat are required for this energy investment, totalling 531 GWh, 190 TJ, and 136 TJ, respectively.

Table 25. Inputs for NPP construction, 1000 MW reactor.

Inputs	Amount	Unit	Comment
concrete production, normal	123657	m3	Average of literature (see Figure 63)
copper, cathode	1147600	kg	Average of literature (see Figure 63)
reinforcing steel production	35936572	kg	Average of literature (see Figure 63)
steel production, low-alloyed, hot rolled	10885813	kg	Average of literature (see Figure 63)
aluminium, cast alloy	64000	kg	ecoinvent assumption
excavation, hydraulic digger	85000	m3	ecoinvent assumption
electricity, low voltage	531000000	kWh	ecoinvent assumption
diesel, burned in building machine	190000000	MJ	ecoinvent assumption
inert waste, for final disposal	-322000000	kg	ecoinvent assumption
heat, district or industrial, other than natural gas	135850000	MJ	ecoinvent assumption

7.3.5 Power plant operation

Chemicals required during the operational phase are shown in Table 27. Furthermore, a comparison of sources is displayed in Figure 64.

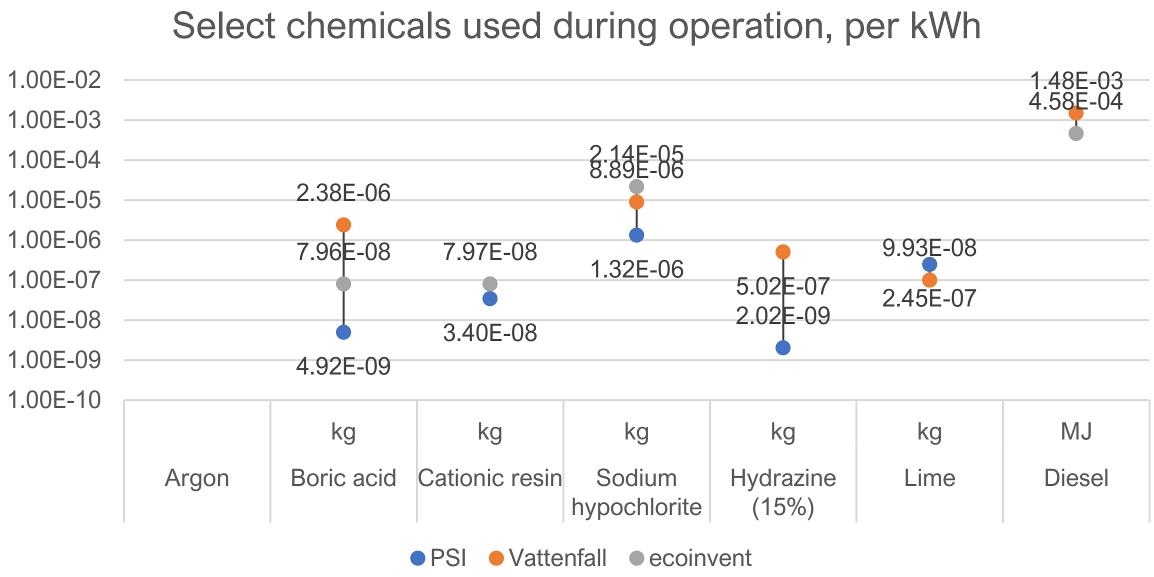

Figure 64. Select list of chemicals used during the operation of a NPP. Sources: [139] and ecoinvent database.

Water requirements (and emissions) may vary significantly depending on the site configuration, as exemplified by the French nuclear fleet [145]. Open-cycle power plants built on the seashore do not dissipate any water, as 100% of the cooling water (about 182 l/kWh) is returned to the water body (sea). In open-cycle power plants using freshwater (river), nearly all water (about 169 l/kWh) is also returned, only 0.2% are removed from the local environment. Finally, closed-cycle plants use much less water, and air-cooling towers to evaporate about 23% of the water taken from the immediate environment, or about 2.3 l/kWh from the 10 l/kWh required. With the conservative assumption that the average PWR plant evaporates at most as much as a closed-cycle cooling system does (**2.3 l/kWh**), we retain this value as an average – bearing in mind that this is a conservative assumption.

The amount of fuel elements required per unit of energy is embodied in the “discharge fuel burnup” (or “burnup rate”, or “fuel utilisation”), a quantity characterised as the amount of energy per ton of uranium contained in the fuel element. The burnup rate is expressed in GW-day per ton, expressing roughly how many days an average reactor (1 GW) can operate on one ton of fuel elements. Conventional values range from 40 to 50 GWd/ton, a value of **42 GWd** per ton is usual for current reactors [146] – this is the value retained for the modelling. An overview of literature values, explicit or recalculated, is given on Figure 65.

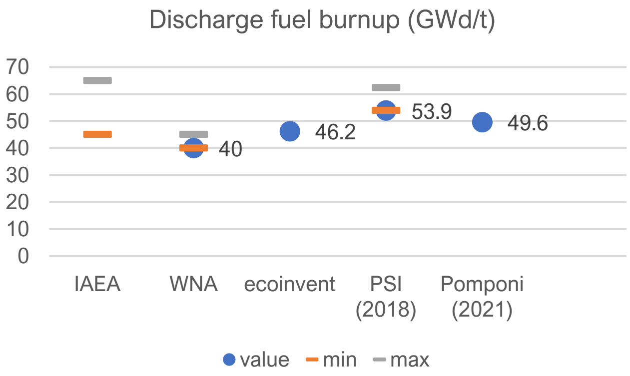

Figure 65. Common values for burnup rates as found in the literature. Sources: [139, 141]

Table 26. Chemical inputs for NPP operation, 1000 MW reactor.

Inputs	Amount	Unit	Comment
argon, liquid	3.23E-05	kg	ecoinvent assumption
boric acid, anhydrous, powder	2.38E-06	kg	WNA consultation
carbon dioxide, liquid	2.07E-07	kg	ecoinvent assumption
chemical, inorganic	2.90E-06	kg	ecoinvent assumption
hydrogen liquid, production mix	2.14E-05	kg	WNA consultation
hydrazine	5.02E-07	kg	WNA consultation
nitrogen, liquid	7.65E-05	kg	ecoinvent assumption
oxygen, liquid	2.07E-05	kg	ecoinvent assumption
sodium hypochlorite, without water, in 15% solution state	8.89E-06	kg	WNA consultation
sodium hydroxide, without water, in 50% solution state	8.94E-07	kg	WNA consultation
acetylene	4.46E-08	kg	ecoinvent assumption
anionic resin	7.97E-08	kg	ecoinvent assumption
cationic resin	7.97E-08	kg	ecoinvent assumption
chemical, organic	1.71E-06	kg	ecoinvent assumption
lubricating oil	2.01E-06	kg	ecoinvent assumption
cement, production mix	1.14E-06	kg	ecoinvent assumption
pitch	9.56E-07	kg	ecoinvent assumption
diesel, burned in diesel-electric generating set	1.48E-03	MJ	WNA consultation
paper, woodfree, coated	7.97E-08	kg	ecoinvent assumption

7.3.6 Power plant decommissioning

Decommissioning covers the deconstruction of the nuclear power plant, as well as the end-of-life treatment of generated waste, be it inert, hazardous, or radioactive. Decommissioning consists in three main distinct phases. First, 5 years are generally required after the final shutdown to remove the spent fuel in a wet storage building. Simultaneously, buildings are prepared for the decommission, which can surpass the 5-year period, preparation generally lasts from 7 (WNA consultation) to 9 years [103]. Finally, decommission itself occurs, including the equipment dismantling and demolition of buildings – processes that can last over 20 years (WNA consultation). The data used for the decommissioning phase is adapted from [139] and updated with data collected during the consultation with WNA experts.

Table 27. Inputs for NPP decommissioning, 1000 MW reactor.

Inputs	Amount	Unit	Comment
diesel, burned in building machine	53550000	MJ	170000 l/year for 9 years [139]
electricity, medium voltage	55188000	kWh	0.70 MW for 9 years [139]
heat, district or industrial, other than natural gas	14300000	MJ	ecoinvent assumption
transport, freight, lorry 20-28 metric ton, production mix	2420000	tkm	ecoinvent assumption
transport, freight train	1800000	tkm	ecoinvent assumption
scrap steel	-19776385	kg	WNA consultation
process-specific burdens, inert material landfill	4500000	kg	WNA consultation
low level radioactive waste for final repository	-5766	m3	WNA consultation

7.3.7 Reprocessing (excluded)

After being spent in reactors, a share of fuel elements is today being reprocessed so that they can be used as fuel again. Reprocessing of used fuel represents a significant opportunity to preserve natural resources and reduce amount and hazard of radioactive waste. The total reprocessing capacity for light water reactors today is about 6000 tonnes of heavy metal (tHM) per year (including about 1000 tHM/y in France, 2000 in the US). New reprocessing plants are expected to be launched, thus with the growth of nuclear the ratio seems to remain.

With the development and deployment of fast neutron reactors, fuel self-sufficiency of nuclear industry (without involvement of a natural component) will increase and can technically even tend to 100% - a scenario in which all fuel is secondary. While no reprocessing is included in this LCA, it is worth mentioning that, currently, the fuel cycle closing through spent fuel reprocessing and Gen IV reactors deployment seems to be a main objective of the global nuclear industry development.

Reprocessing is excluded from this LCA, i.e. all uranium used as fuel is primary (see [11] above). Recent LCA work suggests that closed-loop fuel cycle (with reprocessing) offers a sensibly lower lifecycle environmental profile as conventional open-loop front-end fuel cycle [130] – indicating that this present work relies on conservative assumptions.

7.3.8 Used fuel management

Used fuel management includes the storage at the nuclear plant site of spent fuel, before it is cooled enough to be stored outside of the reactor pools during an interim storage before it will be deposited in a final repository. Interim storage may be in the form of dry casks that will house several spent fuel assemblies with natural ventilation or in dedicated pools.

Table 28. Inputs for interim storage of spent fuel, per TWh of average NPP operation.

Inputs	Amount	Unit	Comment
petrol, low-sulfur	1.00E+01	kg	From WNA consultation
diesel, burned in building machine	8.41E+03	MJ	From WNA consultation
hazardous waste, for incineration	1.11E+02	kg	From WNA consultation
inert waste, for final disposal	1.88E+02	kg	From WNA consultation
water, decarbonised	3.12E+02	kg	From WNA consultation
electricity, high voltage	3.78E+05	kWh	From WNA consultation
chemicals, inorganic	2.11E-01	kg	From WNA consultation
acrylic dispersion, without water, in 65% solution state	1.34E-02	kg	From WNA consultation
butyl acetate	8.50E-02	kg	From WNA consultation
ethanol, without water, in 99.7% solution state, from fermentation	6.38E+00	kg	From WNA consultation
ethyl acetate	4.67E-02	kg	From WNA consultation
hydrazine	3.00E-01	kg	From WNA consultation
isopropanol	2.05E+00	kg	From WNA consultation
lubricating oil	9.05E-02	kg	From WNA consultation
methyl ethyl ketone	2.83E-03	kg	From WNA consultation
methyl methacrylate	1.59E-03	kg	From WNA consultation
refrigerant R134a	3.10E-01	kg	From WNA consultation
silicone product	5.56E-02	kg	From WNA consultation
soap	3.59E+00	kg	From WNA consultation
anionic resin	9.73E+01	kg	From WNA consultation
monoethanolamine	6.80E-03	kg	From WNA consultation
sodium chloride, powder	1.70E+00	kg	From WNA consultation
ethylene glycol	5.35E-01	kg	From WNA consultation

7.3.9 High-level radioactive waste management and disposal

This last phase of the backend part of the uranium chain will be the disposal of either spent fuel assemblies or high radioactive wastes resulting from the reprocessing of the assemblies in a deep geological repository. While deep geological sites for disposal have existed for decades at the research scale, no mature commercial repository is active as of 2021. The commercial site closest to operation is the Onkalo spent nuclear fuel repository, near the Olkiluoto power plant in Finland; operation is foreseen as soon as 2023. Another site in Sweden (Forsmark) is rather advanced, with 2030 as a possible operation date. The fact that no site is currently in exploitation means that lifecycle data has to be estimated from the current projects' advancements. These estimates are based on Vattenfall assumptions, and collected data so far, regarding the encapsulation of the spent fuel assemblies into canisters and their final disposal in a deep geological repository. The next decade will be key in radioactive waste treatment, as other projects are under development – experience feedback will then help refining lifecycle inventories.

Encapsulation is done by enclosing spent fuel in copper-cast iron canisters. Two designs exist depending on the copper-to-insert (cast iron) ratio, both designs can contain 3.6 tons of spent fuel for a total weight of 24.3-24.6 tons [147], we use the 50-mm copper design for the LCA model. Each canister can contain 3600 kg of spent fuel elements, consisting of UO_2 in their zirconium envelope. The uranium fuel chain model shows that 2.92 mg of uranium in fuel elements is required per kWh of electricity, which translates to 3.31 mg of UO_2, or 7.98 mg of fuel elements including the zirconium envelope. About 2.2 canisters are therefore needed per TWh of electricity output.

Table 29. Inputs for one spent fuel canister.

Inputs	Amount	Unit	Comment
copper, cathode	7400	kg	Hedman, Nyström [147]
cast iron	13600	kg	Hedman, Nyström [147]
welding, arc, aluminium	3.30	m	Assuming welding around the cap (diameter 1050 mm) and approximating fusion welding with arc welding

Table 30. Inputs for encapsulation of spent fuel from interim storage, per TWh of NPP operation.

Inputs	Amount	Unit	Comment
Spent fuel canister	2.2	unit	From WNA consultation
diesel, burned in diesel-electric generating set, 10MW	1448	MJ	From WNA consultation
ethanol, without water, in 95% solution state, from fermentation	0.028	kg	From WNA consultation
lubricating oil	0.81	kg	From WNA consultation
soap	4.4	kg	From WNA consultation
electricity, medium voltage	310282	kWh	From WNA consultation

Table 31. Inputs for deep waste repository, per TWh of NPP operation.

Inputs	Amount	Unit	Comment
market group for concrete, normal	2.59	m3	From WNA consultation
blasting	1140	kg	From WNA consultation
diesel, burned in diesel-electric generating set, 10MW	52640	MJ	From WNA consultation
light fuel oil	9984	kg	From WNA consultation
electricity, medium voltage	738766	kWh	From WNA consultation
reinforcing steel	113	kg	From WNA consultation

7.4 Characterisation factors

7.4.1 Land use

Table 32. Land use characterisation factors, in points.

Occupation or transformation by land type	Value	pts per
Occupation, annual crop	131	m2a
Occupation, annual crop, flooded crop	91.4	m2a
Occupation, annual crop, greenhouse	89	m2a
Occupation, annual crop, irrigated	131	m2a
Occupation, annual crop, irrigated, extensive	124	m2a
Occupation, annual crop, irrigated, intensive	136	m2a
Occupation, annual crop, non-irrigated	131	m2a
Occupation, annual crop, non-irrigated, extensive	124	m2a
Occupation, annual crop, non-irrigated, intensive	136	m2a
Occupation, arable land, unspecified use	131	m2a
Occupation, construction site	207	m2a
Occupation, dump site	158	m2a
Occupation, field margin/hedgerow	98.7	m2a
Occupation, forest, extensive	68.5	m2a
Occupation, forest, intensive	78.2	m2a
Occupation, grassland, natural (non-use)	98.5	m2a
Occupation, industrial area	244	m2a
Occupation, mineral extraction site	207	m2a
Occupation, pasture, man made	117	m2a
Occupation, pasture, man made, extensive	101	m2a
Occupation, pasture, man made, intensive	119	m2a
Occupation, permanent crop	131	m2a
Occupation, permanent crop, irrigated	131	m2a
Occupation, permanent crop, irrigated, extensive	124	m2a
Occupation, permanent crop, irrigated, intensive	131	m2a
Occupation, permanent crop, non-irrigated	131	m2a
Occupation, permanent crop, non-irrigated, extensive	124	m2a
Occupation, permanent crop, non-irrigated, intensive	131	m2a
Occupation, shrub land, sclerophyllous	78.5	m2a
Occupation, traffic area, rail network	244	m2a
Occupation, traffic area, rail/road embankment	192	m2a
Occupation, traffic area, road network	288	m2a
Occupation, unspecified	134	m2a
Occupation, urban, continuously built	301	m2a
Occupation, urban, discontinuously built	184	m2a
Occupation, urban, green area	121	m2a
Occupation, urban/industrial fallow (non-use)	243	m2a
Transformation, from annual crop	-131	m2
Transformation, from annual crop, flooded crop	-91.4	m2
Transformation, from annual crop, greenhouse	-89	m2
Transformation, from annual crop, irrigated	-131	m2
Transformation, from annual crop, irrigated, extensive	-124	m2
Transformation, from annual crop, irrigated, intensive	-136	m2
Transformation, from annual crop, non-irrigated	-131	m2
Transformation, from annual crop, non-irrigated, extensive	-124	m2
Transformation, from annual crop, non-irrigated, intensive	-136	m2
Transformation, from arable land, unspecified use	-131	m2
Transformation, from cropland fallow (non-use)	-243	m2
Transformation, from dump site	-158	m2
Transformation, from dump site, inert material landfill	-158	m2
Transformation, from dump site, residual material landfill	-158	m2
Transformation, from dump site, sanitary landfill	-158	m2
Transformation, from dump site, slag compartment	-158	m2
Transformation, from field margin/hedgerow	-98.7	m2
Transformation, from forest, extensive	-68.5	m2
Transformation, from forest, intensive	-78.2	m2
Transformation, from forest, primary (non-use)	-63.6	m2
Transformation, from forest, secondary (non-use)	-63.7	m2
Transformation, from forest, unspecified	-71	m2
Transformation, from grassland, natural (non-use)	-98.7	m2
Transformation, from heterogeneous, agricultural	-121	m2
Transformation, from industrial area	-244	m2
Transformation, from mineral extraction site	-207	m2
Transformation, from pasture, man made	-117	m2
Transformation, from pasture, man made, extensive	-101	m2
Transformation, from pasture, man made, intensive	-119	m2
Transformation, from permanent crop	-131	m2
Transformation, from permanent crop, irrigated	-131	m2
Transformation, from permanent crop, irrigated, extensive	-124	m2
Transformation, from permanent crop, irrigated, intensive	-131	m2
Transformation, from permanent crop, non-irrigated	-131	m2
Transformation, from permanent crop, non-irrigated, extensive	-124	m2
Transformation, from permanent crop, non-irrigated, intensive	-131	m2
Transformation, from shrub land, sclerophyllous	-78.6	m2
Transformation, from traffic area, rail network	-244	m2
Transformation, from traffic area, rail/road embankment	-192	m2
Transformation, from traffic area, road network	-288	m2
Transformation, from unspecified	-114	m2
Transformation, from unspecified, natural (non-use)	-103	m2
Transformation, from urban, continuously built	-301	m2
Transformation, from urban, discontinuously built	-184	m2
Transformation, from urban, green area	-121	m2
Transformation, from urban/industrial fallow (non-use)	-243	m2
Transformation, to annual crop	131	m2
Transformation, to annual crop, flooded crop	91.4	m2
Transformation, to annual crop, greenhouse	89	m2
Transformation, to annual crop, irrigated	131	m2
Transformation, to annual crop, irrigated, extensive	124	m2
Transformation, to annual crop, irrigated, intensive	136	m2
Transformation, to annual crop, non-irrigated	131	m2
Transformation, to annual crop, non-irrigated, extensive	124	m2
Transformation, to annual crop, non-irrigated, intensive	136	m2
Transformation, to arable land, unspecified use	131	m2
Transformation, to cropland fallow (non-use)	243	m2
Transformation, to dump site	158	m2
Transformation, to dump site, inert material landfill	158	m2
Transformation, to dump site, residual material landfill	158	m2
Transformation, to dump site, sanitary landfill	158	m2
Transformation, to dump site, slag compartment	158	m2
Transformation, to field margin/hedgerow	98.7	m2
Transformation, to forest, extensive	68.5	m2
Transformation, to forest, intensive	78.2	m2
Transformation, to forest, unspecified	71	m2
Transformation, to heterogeneous, agricultural	121	m2
Transformation, to industrial area	244	m2
Transformation, to mineral extraction site	207	m2
Transformation, to pasture, man made	117	m2
Transformation, to pasture, man made, extensive	101	m2
Transformation, to pasture, man made, intensive	119	m2
Transformation, to permanent crop	131	m2
Transformation, to permanent crop, irrigated	131	m2
Transformation, to permanent crop, irrigated, extensive	124	m2
Transformation, to permanent crop, irrigated, intensive	131	m2
Transformation, to permanent crop, non-irrigated	131	m2
Transformation, to permanent crop, non-irrigated, extensive	124	m2
Transformation, to permanent crop, non-irrigated, intensive	131	m2
Transformation, to shrub land, sclerophyllous	78.6	m2
Transformation, to traffic area, rail network	244	m2
Transformation, to traffic area, rail/road embankment	192	m2
Transformation, to traffic area, road network	288	m2
Transformation, to unspecified	114	m2
Transformation, to urban, continuously built	301	m2
Transformation, to urban, discontinuously built	184	m2
Transformation, to urban, green area	121	m2
Transformation, to urban/industrial fallow (non-use)	243	m2