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LETTER

Prospective analysis of external environmental costs of energy provision

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Abstract

Human activities, particularly energy supply, often lead to significant environmental impacts that are typically overlooked in economic markets and policy decisions. Here, we evaluate the environmental costs associated with different energy carriers and technologies using prospective life cycle assessment combined with monetary valuation across a broad spectrum of environmental indicators and monetization perspectives. Our findings reveal that, for most energy carriers and technologies, external costs are at least as high as production costs and higher than previously estimated, regardless of choice of monetization perspective used. However, that choice significantly affects the relative importance of environmental indicators and associated costs. Nonetheless, non-climate-related impacts often account for more than half of the total costs. Fossil-based electricity and heat generation exhibit high external costs, whereas renewable and nuclear energy show comparatively lower environmental costs. However, bio-based fuels often incur higher costs than their fossil counterparts due to substantial land use impacts.

1. Introduction

Human influence on Earth's systems has accelerated tremendously over the last few decades [1] and has led to a host of environmental concerns and crises that profoundly affect both natural and human processes. These include climate change [2], air pollution [3], biodiversity loss [4], water scarcity [5, 6], and chemical pollution [7], several of which are interlinked [8, 9]. The consequences of these impacts are unevenly distributed, and marginalized groups bear the brunt of the effects [2]. Planetary Boundaries [10] highlight the breadth of anthropogenic influence and relate it to the stability of the Earth system. Their most recent assessment [11] concluded that six out of nine boundaries have already been transgressed.

From the perspective of welfare economics, such environmental harm can be addressed in a welfare-optimal manner by taxing polluting activities at the level of the environmental damages caused [12]. Such a 'Pigouvian tax' internalizes the damages, leading to a reduction in environmental impacts. If such external costs are to be used for decision-making or pathway modeling, for example with Integrated Assessment Models (IAMs), they must be calculated prospectively to reflect changes in environmental impacts due to changes in the techno-economic system. Moreover, the inherent variability in economic valuation, driven by empirical uncertainties and evolving societal values, must be carefully addressed.

The ExternE project [13] and its successor, NEEDS [14], have been pivotal in assessing external environmental costs of energy and still constitute the backbone of this area of research. Their methodological guidance

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and databases have been adopted for many case studies [15–17], as well as recent reports [18, 19] that assess external costs across whole sectors. These reports estimate the external costs of energy at 6.6% of GDP in the EU and G20 countries [18], and transport-related externalities at 4.8% of GDP in the EU [19]. Based on a meta-analysis, Sovacool *et al* [20] further estimate that global externalities from electricity and transport amount to 28.7% of global GDP, highlighting that these external costs are often comparable to the levelized costs of electricity production. In a recent paper [21], Baumgärtner *et al* advance the granularity of external cost of electricity estimates by applying country-specific emission data and monetization factors in the EU, building on the NEEDS methodology. They find a large spread of costs between 2.1 and 22.4 cents/kWh in the time period 2010-2030, and that costs decrease in only 8 EU countries. In their comparative review, Bielecki *et al* [22] conclude that the lower externalities of renewable energy sources such as solar, wind, and geothermal can make them competitive against fossil sources with low production costs. They also highlight the heterogeneity and complexity of approaches to assess externalities.

Several studies on external costs use Life Cycle Assessment (LCA)-based methodologies [14, 18, 19]. Life Cycle Impact Assessment (LCIA) typically assesses potential biophysical impacts, which require monetary valuation to translate these impacts into environmental costs. The review by Pizzol *et al* [23] significantly contributed to the field by classifying various approaches and methods, advocating for specific strategies tailored to each impact category. Building on this foundation, the reviews by Arendt *et al* [24] and Amadei *et al* [25] offered further classification and qualitative assessments of different methodologies as well as quantitatively evaluated published monetization factors. Their studies reveal a considerable spread across methods and a notable imbalance in research focus between impact categories. This variation is partly due to uncertainties that could be reduced with more data collection, but it also results from differing value judgments across methods [24, 26]. For example, these differences include how costs are estimated, choice of discount rates, and using equity weighting to transfer values across regions.

Pehl *et al* and Arvesen *et al* advanced the field by introducing prospective Life Cycle Assessment (pLCA) based on IAM scenarios [27, 28], building on earlier work in scenario-based impact analysis [29, 30]. The integration between IAM and LCA has been further refined within the *premise* framework [31]. Building on this research, Luderer *et al* [32] extensively evaluated the environmental impacts across various scenarios, demonstrating that while climate change mitigation offers significant environmental co-benefits, it also shifts the burden from fossil fuel dependence to increased metal and mineral resource depletion and potentially greater impacts on ecosystems due to land use expansion. Rauner *et al*'s analysis [33] focused on coal exit scenarios, revealing that environmental benefits outweigh the associated policy costs.

This study presents several significant contributions to the field of external costs of energy assessments. The NEEDS project [14] already provided some prospective cost estimates that are used in follow-up studies [21]. We advance the prospective, scenario-based approach by using the tool *premise* [31], allowing us to assess costs for different climate policy scenarios. Most previous studies focused on power generation; we extend the coverage of energy carriers beyond electricity to also include heat and liquid and gaseous fuels, each with a diverse set of competing technologies. Combined with the operationalized approach using premise, this paves the way for internalization in full energy system or integrated assessment models. The inherent variability in monetary valuation is commonly addressed by applying a simple triangular sensitivity analysis. We instead apply different monetization perspectives with clearly defined approaches, and perform Monte Carlo simulations to derive full distributions. This nuanced analysis allows us to distinguish robust findings across valuation methods from those highly dependent on the chosen monetization perspective. Lastly, we further broaden the scope of impact pathways to include all pathways commonly found in life cycle impact assessment.

2. Methods

2.1. Energy-economy model

The starting point of our analysis is the integrated assessment model *REMIND* [34]. Its core comprises a multiregional general equilibrium model with Ramsey-type growth formulated as an optimization problem. Maximized is the inter-temporal welfare of the 12 world regions of *REMIND*, discounted with an effective rate of ∼4.5% (region-specific) [5]. This macroeconomic core is hard-linked to *REMIND*'s detailed energy system representation through induced energy demands. It features energy carriers on primary, secondary, and final energy levels and more than 50 conversion technologies between the levels. The final energy demand is split into three sectors: industry, transport, and buildings. Biomass supply and land-use emissions are provided by the land-use model MAgPIE [35]. The final energy demands originate from the EDGE models [36, 37].

[5] The discount rate is given by $d + ng$, with pure rate of time preference d, relative risk aversion n and growth rates g. *REMIND* uses $d = 0.015$ and $n = 1.5$, growth rates are region specific.

Prescribed greenhouse gas emission budgets and taxes and subsidies on energy activities allow the implementation of climate policies. This study uses the *REMIND* release version 3.2.1 (https://github.com/remindmodel/remind/releases/tag/v3.2.1). We consider two scenarios based on SSP2 assumptions: Firstly, a scenario capturing National Policies implemented ('NPi') and therefore assuming a continuation of current trends. Secondly, a stringent climate policy scenario consistent with the 1.5 °C target of the Paris Agreement through a peak budget of 500 Gt CO2 from 2015 to 2100 ('1.5 °C').

2.2. Prospective LCA

The output data from the *REMIND* runs feed into the prospective LCA (pLCA) framework *premise* [31] (version 2.0.2), which modifies the LCA database ecoinvent (https://ecoinvent.org/, version 3.9.1, using the 'Allocation, cut-off by classification' system model) according to the IAM projections, thereby creating prospective, time- and scenario-dependent versions. These modifications include the creation of regional markets, adapting fuel and electricity mixes, adding inventories relevant to the energy transition, and applying changes in the efficiency and emission factors of power and heat generation technologies. We create pLCA databases for the NPi and the 1.5°C scenarios for 2020, 2030, 2040, and 2050.

2.3. System boundaries, additional inventories and technology selection

The system boundaries of our analysis encompass the cradle-to-gate provision of secondary energy carriers electricity, heat, liquid fuels, gaseous fuels, and hydrogen by various production technologies. For combustable carriers, we further include the combustion stage.

The electrolysis-based synfuel and hydrogen production activities in the pLCA databases use grid electricity by default. They are used to create new activities that use only wind and solar power inputs to the electricity markets, which feed into the hydrogen production datasets.

The names and technologies selected for analysis in this study are listed in Supplementary table 1.

2.4. External environmental cost calculation

Calculating external environmental costs can be seen as a life cycle impact assessment (LCIA) with monetization as weighting. In a formula:

$$EEC = w \cdot Q \cdot B \cdot A^{-1} \cdot f, \tag{1}$$

where the quantities are defined as follows:

- f is the demand vector, which is equal to one in the row representing the activity whose EEC is to be calculated and zero otherwise
- A is the technosphere matrix, containing all inputs and outputs between activities in the techno-economic production system
- B is the biosphere matrix, which contains all exchanges from technosphere activities into the environment
- Q is the characterization matrix, in which each row defines one LCIA method that aggregates environmental flows into so-called midpoints
- w is the weighting vector, which, in our case, is the monetization factor for each LCIA method used.

We use 73 different LCIA methods, the union of all midpoint methods used in our collection of monetization studies. We further categorize the methods into the following impact categories (some with subcategories), aligned with the typical midpoints of well-established methods such as ReCiPe or Environmental Footprint ([38, 39]):

- acidification
- climate change (biogenic, fossil, land use)
- ecotoxicity (freshwater, marine, terrestrial)
- eutrophication (freshwater, marine)
- human toxicity (carcinogenic, non-carcinogenic)
- ionizing radiation
- land use

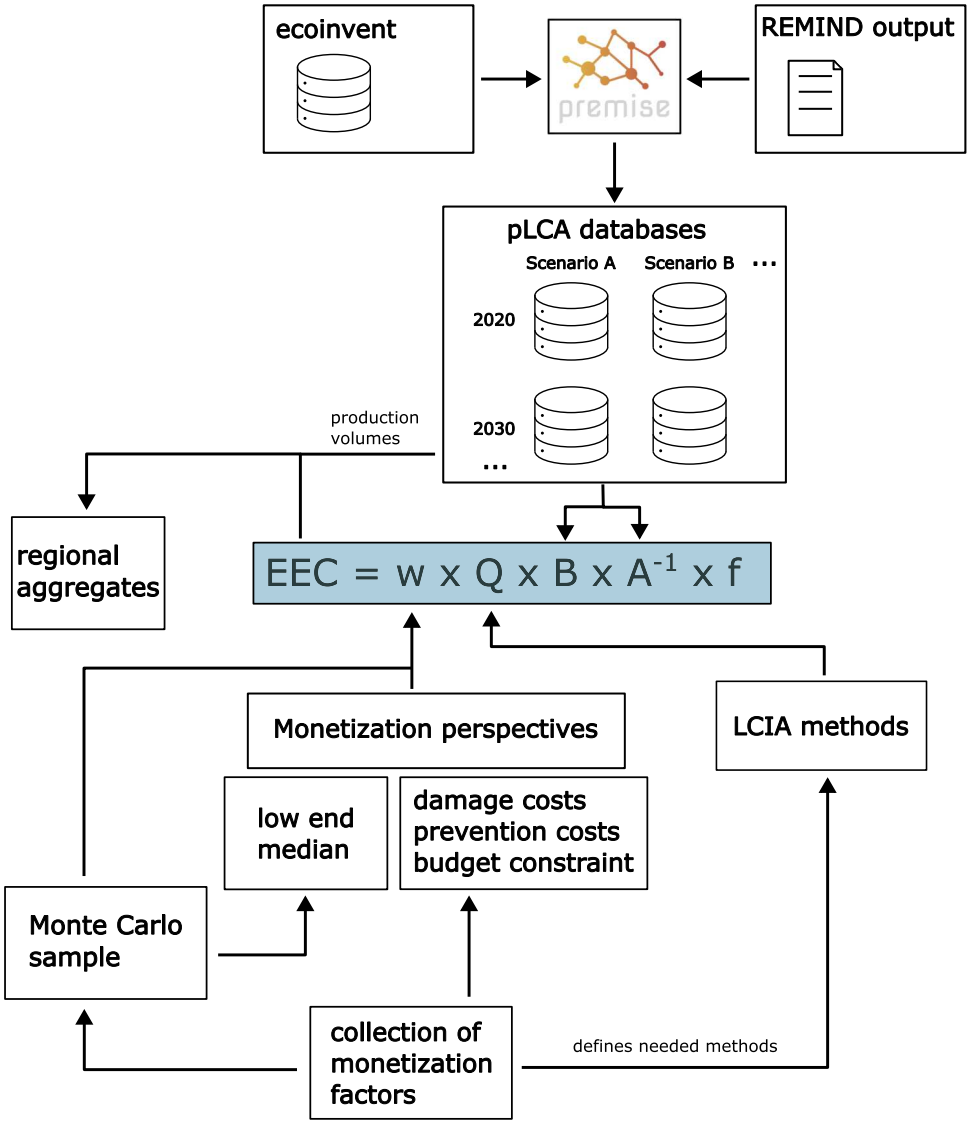

Figure 1. Data flow diagram for external environmental cost calculation. *REMIND* output data and the ecoinvent database are combined using *premise* to build prospective LCA databases. These provide the technosphere and biosphere matrices *A* and *B*. Our collection of monetization factors define the LCIA methods (with characterization factors *Q*) that need to be run. Based on the monetization factors, a Monte Carlo sample is built and monetization perspectives are defined, both of which are used to provide (distributions of) weighting factors *w*. Regional aggregates of individual EECs are calculated using production volumes that are included in the prospective LCA databases.

- metal/mineral resources
- ozone depletion
- particulate matter formation
- photochemical oxidant formation (human health, terrestrial ecosystems)
- water use

We don't consider the midpoint 'fossil resources' because, in all cases, the monetization factors represent market values of fuels and, therefore, not external costs. We perform calculations with the *brightway* LCA framework [40]. In figure 1 a data flow diagram for the EEC calculation is shown.

2.5. Impacts of fuel combustion

We add combustion impacts of liquid and gaseous fuels based on emission factors contained in *premise*. For biofuels, we scale the emission factors of corresponding fossil liquids by scale factors collected from literature sources [41, 42]. The full set of resulting emission factors is listed in Supplementary table 2.

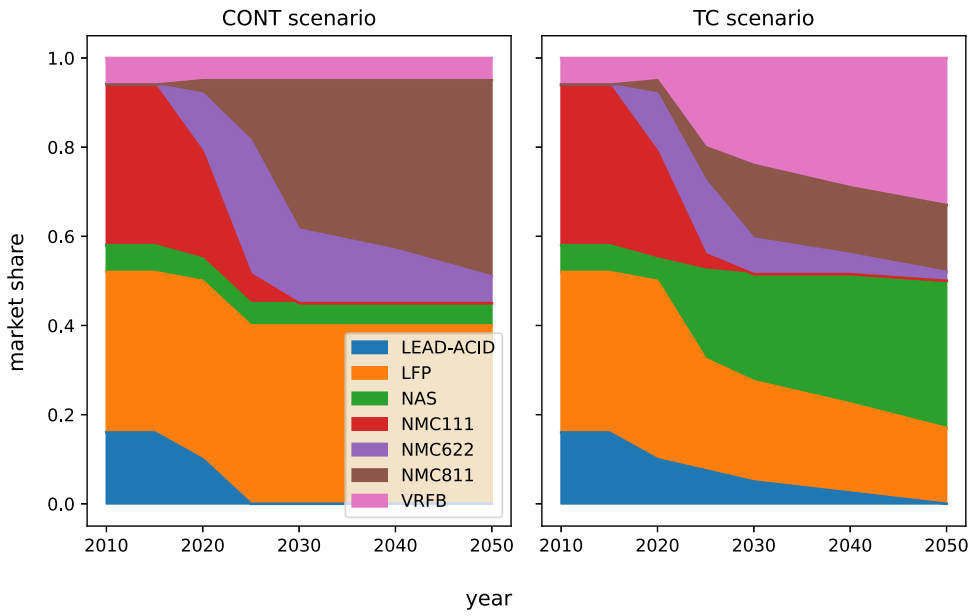

Figure 2. Scenarios for battery capacity market shares The scenarios are taken from Schlichenmaier *et al* [43]. The 'CONT' scenario describes a continuation of current trends without sudden jumps in market shares, whereas in the 'TC' scenario newer sub-technologies gain rapid increases in market shares. The considered battery chemistries/subtechnologies are lithium nickel manganese cobalt (NMC-111, NMC-622, NMC-811), lithium iron phosphate (LFP), lead-acid batteries, vanadium redox flow batteries (VRFB), and Sodium-sulfur and other molten salt batteries (NAS).

2.6. Costs of battery storage

We calculate costs of battery storage required for variable renewable electricity (VRE) generation. *premise* contains datasets that model the market for stationary battery capacity following two scenarios ('CONT' and 'TC') from Schlichenmaier *et al* [43]. The market shares of battery chemistries are shown in figure 2.

From the costs per battery capacity $EEC_{battery}$, we calculate the cost mark-up MU_t for a VRE technology t:

$$MU_t = EEC_{battery} \cdot CR_t \cdot EP \cdot f_t. \tag{2}$$

We only consider solar PV and wind as the main VRE technologies, and restrict ourselves to onshore wind turbines, as offshore wind typically has lower diurnal variations that are commonly balanced with battery storage [44, 45]. The capacity ratios CR_t (between added battery storage capacity and VRE installed capacity) are taken directly from the REMIND output for each year, region, and scenario. EP is the ratio between energy capacity and the nameplate (discharge) capacity of the batteries, the inverse of the battery C-rate. Based on literature [46], we assume values between 4 and 6 appropriate for the application case of diurnal load balancing. As we want to calculate the costs per energy unit VRE production, we need to scale by f_t, the fraction of one kWh relative to the total power production over the lifetime per unit of installed capacity. Thus,

$$f_t = \frac{1}{CF_t \cdot \tau_t \cdot 8760\ \text{h}} \tag{3}$$

where CF are the capacity factors by VRE technology, region, year, and scenario, again derived directly from REMIND output data, and τ are the lifetimes of wind turbines and solar PV installations. We assume 25 yr and 30 yr, respectively.

We derive similar cost mark-ups for hydrogen production through PEM electrolysis using VRE electricity:

$$MU_{h2} = (s_{\text{PV}} \cdot MU_{\text{PV}} + s_{\text{wind onshore}} \cdot MU_{\text{wind onshore}})/\eta_{\text{PEM}}, \tag{4}$$

with respective technology shares s_t in the VRE electricity markets (see section 2.3), and the PEM efficiency η_{PEM} as taken from the REMIND scenario outputs.

2.7. Regional aggregation

The ecoinvent database (updated by premise) contains regional datasets. We calculate regional averages weighted by production volumes (based on the IAM scenario data), an 'overall' average over all regions globally, and 'IAM region' averages for each of the 12 world regions of *REMIND*.

2.8. Monetization

Our monetization is based on a collection of monetization factors (Supplementary table 3), i.e., conversion factors from impact units to monetary units. This collection is based on a review by Amadei *et al* [25]; it includes 160 estimates from 22 sources. Each factor is referenced to an appropriate LCIA method, either given explicitly in the respective study or selected based on the impact units.

We implement a Monte Carlo approach to sample the entire collection of monetization factors. The algorithm used to generate one sample for a given impact category is as follows:

1. Choose uniformly at random from all sources providing estimates for that impact category.
2. Depending on the estimate(s) given by the chosen source, draw a monetization factor value from a distribution:
   - Only central estimate: single value
   - Minimum and maximum given: Draw from a symmetric triangular distribution
   - Minimum, central, and maximum given: Draw from a triangular distribution with a mode equal to the central estimate
   - Several values are given: Draw from them uniformly
3. If the first study chosen does not completely cover an impact category, repeat 1. and 2. for studies that provide estimates for the complementary subcategories (Example: The first study only provides estimates for freshwater eutrophication, so in the second step, choose between studies that provide estimates for marine eutrophication).

In addition, we consider four distinct monetization perspectives, as explained in table 1 below.

2.9. Sensitivity analysis

We perform a one-factor-at-a-time sensitivity analysis to determine the relative amounts of variations in external environmental costs introduced by variations in the technosphere, biosphere, and monetization factors. Starting from 2,000 samples of technosphere matrices, biosphere matrices, and monetization factors, we generate for each technology i distributions where only one of the factors A, B, and w is varoed. From these distributions, we compute the normalized variances to be used as sensitivity scores:

$$\begin{aligned} S^i_{\mathrm{A}} &= \mathrm{Var}^i_{\mathrm{A}}/(\mathrm{Var}^i_{\mathrm{A}} + \mathrm{Var}^i_{\mathrm{B}} + \mathrm{Var}^i_{\mathrm{w}}) \\ S^i_{\mathrm{B}} &= \mathrm{Var}^i_{\mathrm{B}}/(\mathrm{Var}^i_{\mathrm{A}} + \mathrm{Var}^i_{\mathrm{B}} + \mathrm{Var}^i_{\mathrm{w}}) \\ S^i_{\mathrm{w}} &= \mathrm{Var}^i_{\mathrm{w}}/(\mathrm{Var}^i_{\mathrm{A}} + \mathrm{Var}^i_{\mathrm{B}} + \mathrm{Var}^i_{\mathrm{w}}). \end{aligned} \tag{5}$$

2.10. Production costs

Production costs were collected from a variety of sources listed in table 4 below. All production costs are listed in Supplementary table 4.

2.11. Conversion of monetary units

Monetization factors were given in different currencies and for different reference years. To convert to the common unit of EUR2022 per impact unit, we first adjust for inflation in the local currency from base year b to target year $t = 2022$ using consumer prices indices (https://bls.gov/data/ and https://data.ecb.europa.eu/data/datasets/ICP/ICP.M.U2.N.000000.4.INX):

$$X_t = X_b \frac{CPI_t}{CPI_b}. \tag{6}$$

Then, the inflation-adjusted values are converted to euros using purchasing power parities (from https://data-explorer.oecd.org):

$$X_{\text{euro}} = X_{\text{other currency}} \frac{PPP_{\text{euro}}}{PPP_{\text{other currency}}}. \tag{7}$$

If a study does not provide a reference year, we assume it to be the publication year. Similarly, we convert production costs to EUR2022, but in this case, we convert to euros first and then adjust for inflation because we assume that the produced goods are tradable [53].

Table 1. Description of monetization perspectives.

Perspective	Rationale	Details	Missing impact categories	Selected studies	Geographical scope
damage costs	costs equal to economic damages	In cases where direct economic valuation is not possible, methods like stated or revealed preferences are used to estimate damages as willingness-to-pay.	metal/mineral resources	Environmental prices [47–49]	Europe
prevention costs	costs equal to marginal cost for reducing the impact	Measures to reduce the given impact are ordered in a marginal abatement cost curve. A level of abatement that is deemed to reduce the impact to sustainable levels is taken to derive the prevention cost.	ionizing radiation	Eco-costs 2023 [50, 51]	mixed
budget constraint	costs derived from reduction of potential per capita economic production	sets the monetary value of quality-adjusted life year (QALY) equal to potential economic production per capita. Ecosystem damages are valued relative to this value using an exchange rate[a], and material use impacts by market prices.	water use	Stepwise method [52]	Europe
low-end estimate	stick to costs that are consistently reported in literature	the 0.2 quantile of the full costs samples		derived in this paper	mixed

[a] In the case of the Stepwise method: 'full protection of an ecosystem of 21 ha (210,000 m2) for one year has the same value as an extra life year at full health for one person.'

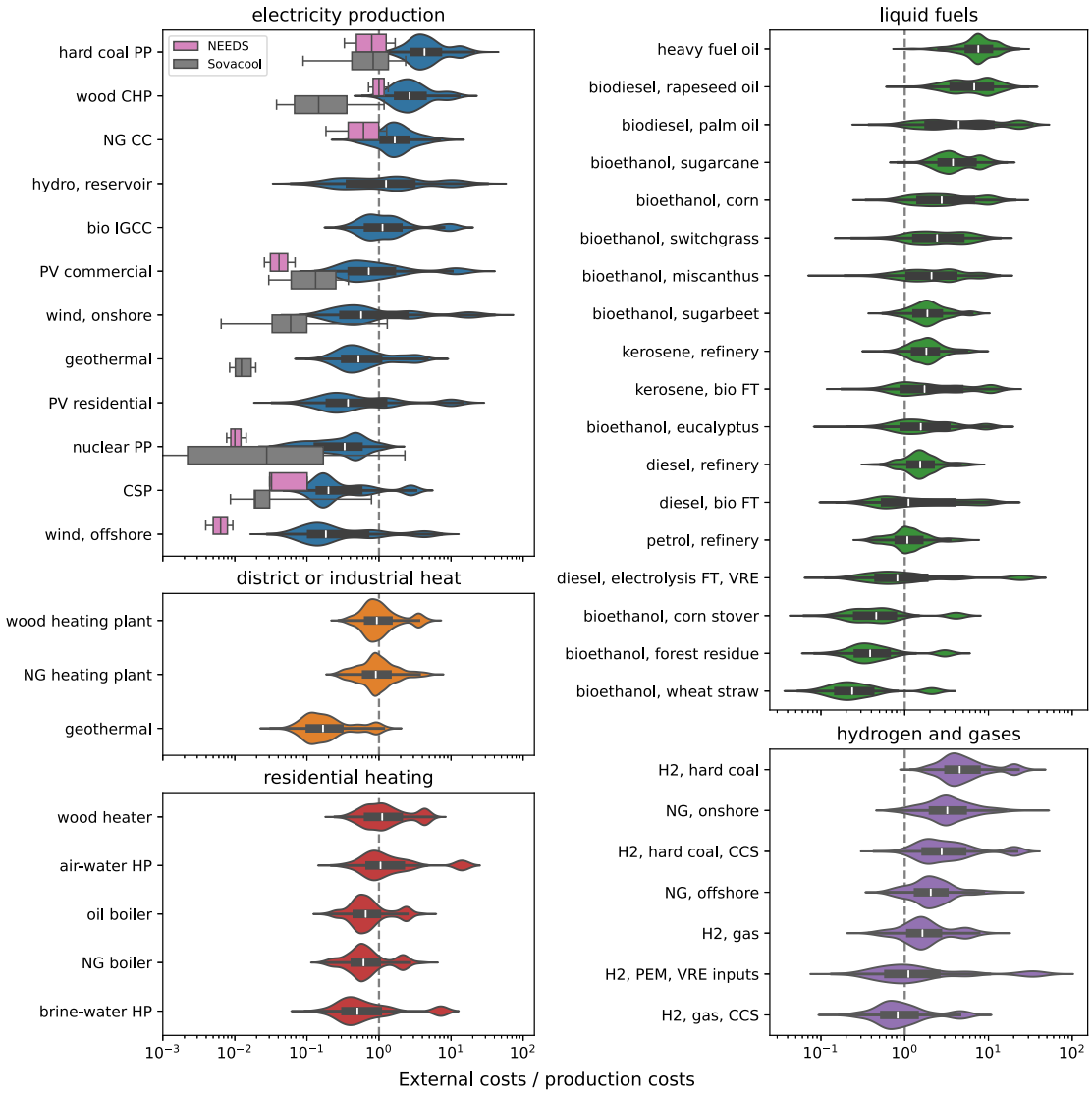

Figure 3. Comparing environmental and production costs. Our external environmental cost estimates are comparable with or even exceed production costs for many technologies, and are generally higher than previous estimates. Plotted are distributions of ratios between current (year 2022) EECs and production costs. The vertical dashed line indicates parity between environmental and production costs; to the right of that line, environmental costs exceed production costs. We consider global average EECs for 2020 in the 'NPi' scenario. Distributions of EECs are derived from the entire collection of monetization factors, and distributions of productions costs from the selection of sources and the ranges therein. Both external and production costs are adjusted for inflation to reflect 2022 costs. For electricity production, we show the comparisons of external costs from other studies, also expressed as ratios of external costs to production costs (same distributions as for our cost estimates). Boxplots show quartiles and whiskers representing 1.5 times the interquartile range. Abbreviations: **PP** power plant; **CHP** combined heat and power; **NG** natural gas; **CC** combined cycle; **IGCC** integrated gasification combined cycle; **PV** photovoltaic; **CSP** concentrated solar power; **HP** heat pump; **FT** Fischer-Tropsch; **VRE** variable renewable energy; **H2** hydrogen; **CCS** carbon capture and storage; **PEM** proton exchange membrane.

3. Results

3.1. Comparing external costs with production costs

Figure 3 provides an overview of current (2022) EECs across various energy carriers and conversion pathways, relative to production costs derived from a variety of bottom-up sources [54–63] (see table 2). Despite considerable uncertainty, environmental costs are in most cases as significant as direct production costs, or even exceed them. Particularly high ratios of EECs to production costs are observed for hard coal electricity (4.3 (3.1 - 6.6))[6]), biodiesel from rapeseed oil (6.8 (3.8–10)), bioethanol from corn (2.8 (1.5–6.2)) and heavy fuel oil (7.6 (5.8–10)). In contrast, certain technologies exhibit much lower EECs compared to production costs, including nuclear power plants (0.33 (0.14–0.52)), concentrated solar power (0.20 (0.15–0.52)), offshore wind (0.18 (0.11–0.64)), geothermal heat plants (0.17 (0.11–0.28)), and residue-based bioethanol (0.39 (0.28–0.61)).

[6] Unless otherwise stated, we present numerical results with uncertainty in the format median (first quartile - third quartile)

Table 2. Sources of production costs.

Sector	Sources	Notes
electricity production	Lazard [54], IRENA [55]	
biofuels	Advanced biofuels, Dimitriou *et al* [56], Domingues *et al* [57]	Biomass costs from Domingues *et al* combined with other costs from Advanced biofuels report
refinery products	US EIA[a]	
synfuels	Ueckert *et al* [58]	
heating	Hansen *et al* [59]	
hydrogen and gases	Global Hydrogen review [60], Lazard [61], IEA WEO 2019 [64]	

[a] https://eia.gov/dnav/pet/pet_pri_refoth_dcu_nus_a.htm

Furthermore, our median EEC estimates are often more than an order of magnitude higher than those reported in previous studies [14, 20], highlighting a substantial revision in the understanding of environmental costs owing to methodological advancements in impact assessment, broader applicability of monetization across impact categories, and potentially an increase of societal cost estimates over time (see section 4 for more details, and Extended Data figure 1 for a more granular comparison to earlier studies).

3.2. Ranking technologies in terms of external costs

For electricity and heat production technologies (see figure 4, first three columns), fossil-based energy technologies, particularly coal power plants, exhibit the highest external costs across all monetization perspectives. Coal combustion, especially lignite combustion, is the most expensive technology, with external costs ranging from 400 EUR/GJ (damage cost estimate) to 60 EUR/GJ (low-end estimate). Other combustion-based technologies follow, such as natural gas and biomass power plants. In contrast, renewable and nuclear energy technologies have significantly lower external costs, with low-end estimates for nuclear and wind energy of less than 1 EUR/GJ.

Liquid biofuels might bear higher external costs than conventional oil-based liquid fuels due to costs associated with freshwater and land use (figure 4, 4th column). External costs of synthetic fuels match those of refinery liquids when biomass or hydrogen from renewable-based electrolysis is used as feedstock. However, the latter has notably high costs caused by human health-impacting extractive activities needed to supply metals (e.g. copper) to renewable energy plants and distribution networks.

In the hydrogen and gas sector (figure 4, 5th column), natural gas and gas-based hydrogen (especially with CCS) show the lowest EECs followed by biomass-based hydrogen. Coal-based and green hydrogen, however, incur higher costs. In section 3.4, we provide a more detailed analysis of how the ranking of external environmental costs for hydrogen production technologies (and other technologies) evolves over time and across scenarios.

3.3. External costs across various monetization approaches

A closer examination of the different monetization approaches reveals that while the relative ranking of technologies by total costs is quite robust, both the absolute total costs and the shares of impact categories depend more strongly on the monetization approach (figure 4). Across all sectors, damage costs are typically the highest, followed by prevention costs, budget constraint and the low-end estimate.

Damage costs are dominated by human toxicity, which typically accounts for 52 percent of the total costs (the median of all shown technologies). Climate change and particulate matter also contribute substantially, as well as land and water use in the case of liquid biofuels.

For the prevention cost perspective, climate change contributes a large share in all sectors. Other significant contributions arise from acidification and eutrophication (especially for fossil technologies), ecotoxicity (for most technologies), particulate matter formation (from fossil fuels), land use (liquid biofuels), and photochemical ozone formation (heavy fuel oil). Interestingly, human toxicity is valued very weakly in the prevention costs-based approach.

Costs calculated using the budget constraint perspective are similar to those calculated from the prevention costs perspective regarding total size and shares but with a lower contribution from acidification and eutrophication. Again, externalities caused by human toxicity are given more weight than those caused by ecotoxicity.

The low-end estimate consists mostly of climate change, particulate matter, and land use, with similar shares between them as in prevention and budget constraint costs, but scaled down by a factor of about 2.

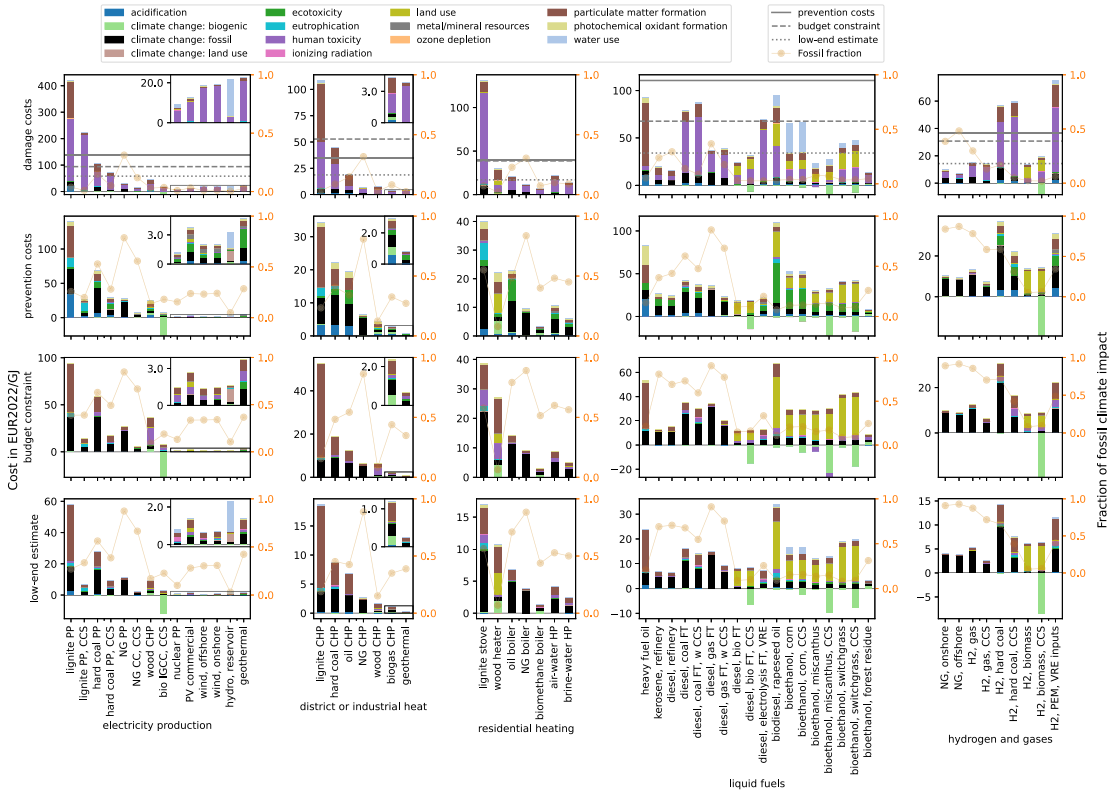

Figure 4. External costs in 2030 from different monetization perspectives. Total costs and how much each impact category contributes varies strongly by perspective, but technology ranking within sectors remain robust. Rows of subplots represent the four monetization perspectives: damage costs, production costs, budget constraint, and low-end estimate. Columns of subplots represent the five sectors of energy provision. The EECs shown here are for the 'NPi' scenario in 2030 and represent a production-weighted average over all dataset locations in the prospective LCA database (regional variation for some technologies are shown in Extended Data figure 2). Insets show the low-cost technologies in electricity production and district or industrial heat in greater detail. Horizontal lines in the damage costs row indicate the maximum costs for the other perspectives to facilitate the comparison of general scales of total costs across monetization perspectives. Abbreviations: **PP** power plant; **CCS** carbon capture and storage; **NG** natural gas; **CC** combined cycle; **PV** photovoltaic; **CHP** combined heat and power; **HP** heat pump; **FT** Fischer-Tropsch; **VRE** variable renewable energy; **H2** hydrogen.

In our analysis, ozone depletion, ionizing radiation, and metal/mineral resource depletion do not feature strongly in any perspective; however, their costs might also be underestimated due to the limitations of the current LCIA methodology (see section 4).

The fraction of total non-negative costs due to fossil climate change impacts (which is plotted on the secondary axis in figure 4) varies with both technology and monetization perspectives. Natural gas-based technologies have the highest fraction, with almost all costs due to climate change in the low-end estimate. Other fossil technologies cause other impacts, lowering the fossil fractions to about 50% in the three cost perspectives other than the damage costs. From these three perspectives, fossil climate impacts in the upstreams of renewable electricity generation and bio-based liquid fuels still amount to around a quarter of total EECs.

3.4. Scenario trends of external costs

We primarily discussed current cost estimates, but our methodology also allows for the projection of future costs by incorporating prospective life-cycle inventories. For many established (fossil) technologies, these inventories remain relatively stable over time, as illustrated in the first column of figure 5 and in Extended Data figure 3. In contrast, electricity-based technologies, e.g., synthetic diesel or heat pumps, exhibit consistent downward trends in cost (shown here are low-end estimate costs; see Extended Data figures 4 and 5 for analogous plots for the 'damage costs' and 'prevention costs' perspectives). The primary driver of these cost changes is the decarbonization of the electricity mixes in the IAM scenarios, which also yield co-benefits such as reduced air pollution, eutrophication, and toxicity impacts. Internalization will provide further incentives to reduce costs over time. In the case of BIGGC, we observe that climate policy scenarios can also increase costs owing to the increased demand for biomass. This, in turn, can drive the use of more land-intensive, purpose-grown biomass, contributing to rising costs.

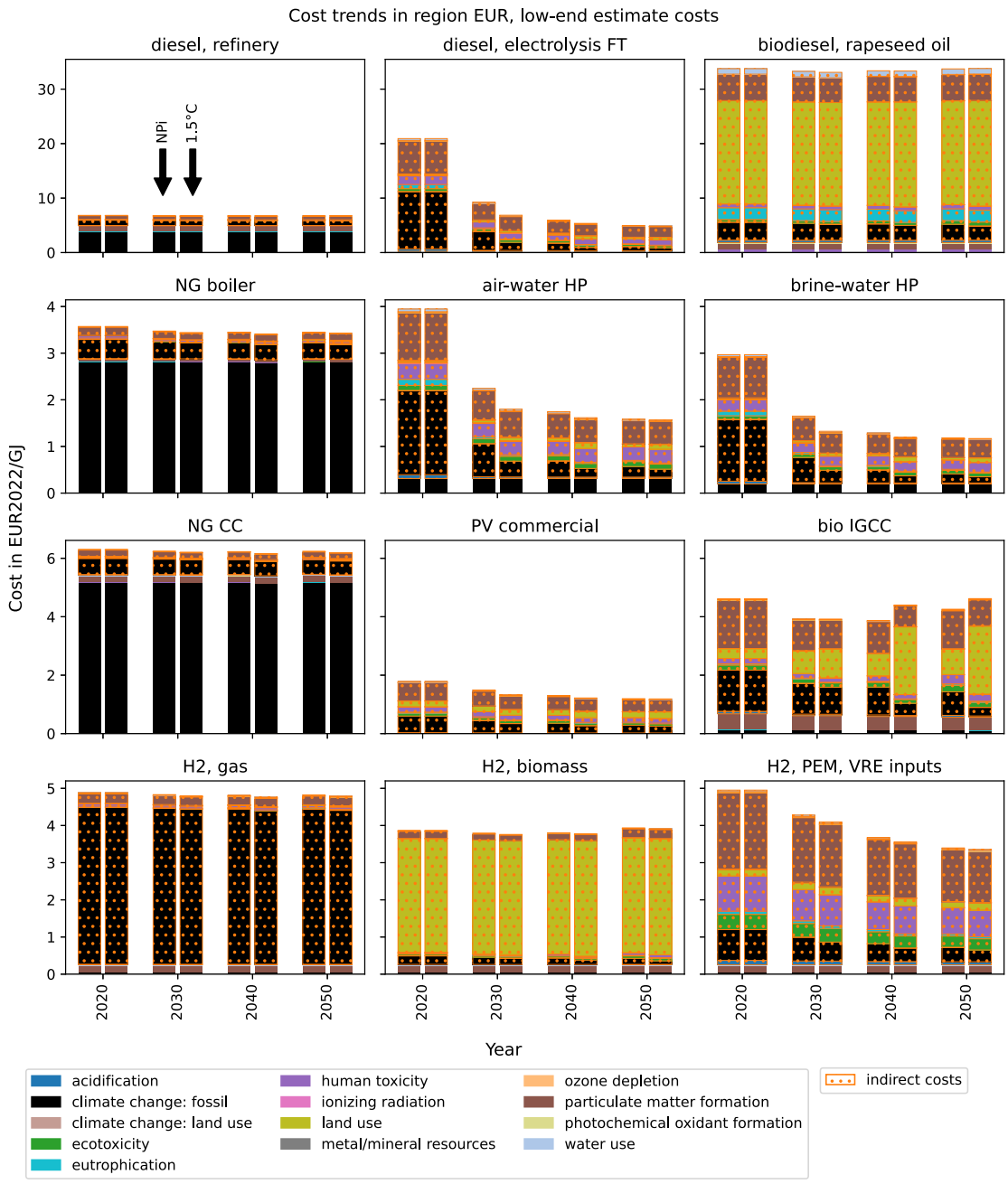

Figure 5. Future projections of external costs under alternative decarbonization scenarios. Costs for established fossil-based technologies remain fairly constant, but decrease for renewable-based over time with increasing climate policy ambition. The EECs for both the 'NPi' and the '1.5°C' scenarios, from 2020 to 2050, are shown using the 'low-end estimate' monetization perspective. Each impact category and color appears twice per bar, with orange dotted segments representing indirect upstream costs. Each row compares a conventional fossil-based technology with alternatives in the sectors of liquid fuels (first row), residential heat (second row), electricity production (third row), and hydrogen production (fourth row). Abbreviations: **FT** Fischer-Tropsch; **NG** natural gas; **HP** heat pump; **CC** combined cycle; **PV** photovoltaic; **IGCC** integrated gasification combined cycle; **H2** hydrogen; **PEM** proton exchange membrane; **VRE** variable renewable energy.

3.5. Costs of battery storage

The EECs of battery storage in both 'CONT' and 'TC' scenarios decreases over time predominantly due to battery efficiency increases (figure 6, first row). The large contributions of toxicity impacts result from treatment of waste produced in the mining of the metals necessary for the battery production. Notably, the TC scenario has slightly higher environmental burdens; with the exception of the metals/minerals category which the 'TC' scenario is optimized for. These costs per battery capacity translate to relative cost increases of up 25% for solar PV and 5% for onshore wind and hydrogen production in the median cost perspective (see figure 6, second row); the other perspectives yield similar results (see Extended Data figure 6). Relative costs increase are mostly driven by the technology-specific trend of storage requirements reflected in the capacity ratio *CP*. They are faster in the 1.5°C scenario due to faster rollout of renewables, and stronger for solar PV.

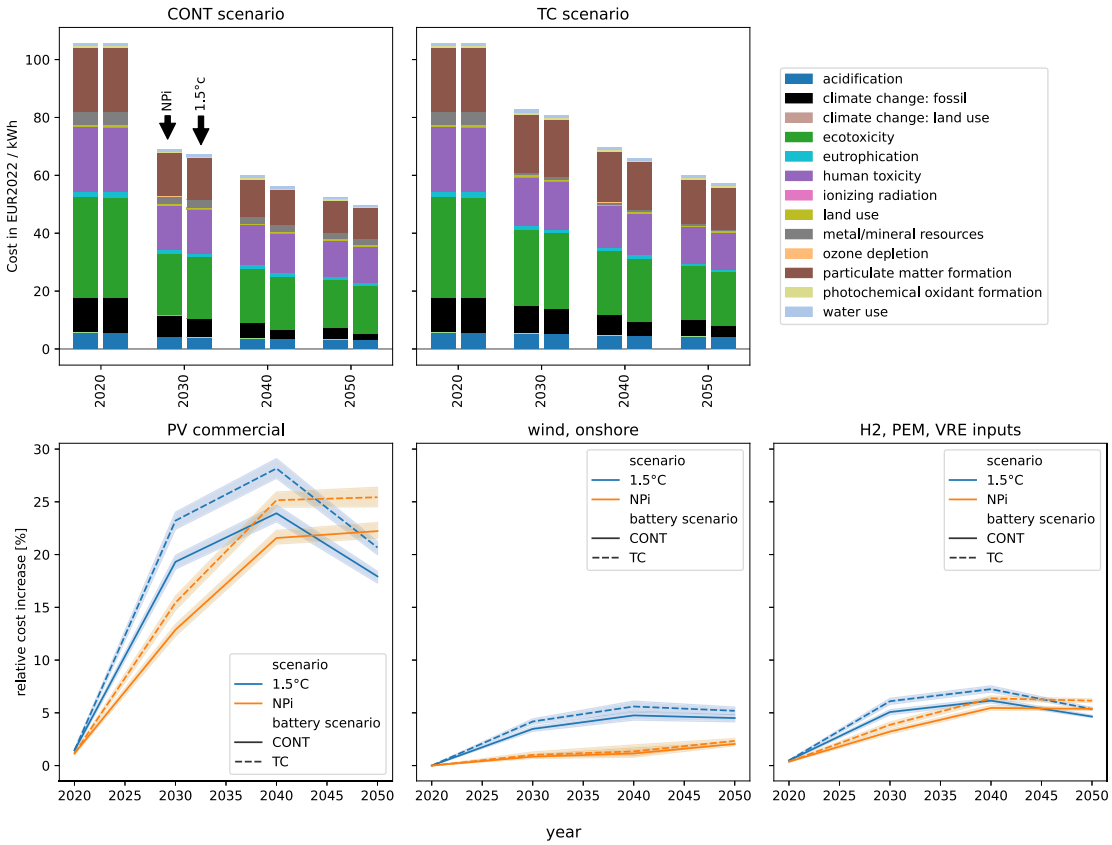

Figure 6. Costs of battery storage Costs per battery storage capacity decreases moderately over time. The cost mark-ups for VRE electricity generation and hydrogen production are most substantial for solar PV. The first row shows external environmental cost trends (by year and scenario) per kWh of battery storage comparing the two battery scenarios. The second row shows resulting cost-markups for solar PV, onshore wind, and electrolytic hydrogen production. All costs shown are using the 'median cost' perspective. All battery capacity datasets are global; regional differentiation in EECs of PV / wind / hydrogen as well as variation in the capacity ratio, capacity factor, and energy-to-power ratio (see section 2.6) produce the 5th-95th percentile confidence intervals in the second row of plots showing the cost mark-ups. Abbreviations: **PV** photovoltaic; **H2** hydrogen; **PEM** proton exchange membrane; **VRE** variable renewable energy.

3.6. Comparing monetization uncertainty to technological uncertainty

The monetization perspective is the most relevant from the list of factors used in our sensitivity analysis. In the ternary plot of sensitivity (normalized variances) shown in figure 7, more than 50% of technologies fall into the upper triangle, which indicates that the variance caused by monetization uncertainty is at least four times larger than the variance from either technosphere or biosphere uncertainty. This is usually due to extreme monetization factors to which the squared metric of variance reacts strongly. The other 50% lies on either side of the triangle, representing either a strong technosphere or biosphere sensitivity as shown in figure 7, subplots a and b. The examples of distributions on the left of figure 7 also show a more detailed qualitative picture beyond variance. While technosphere and biosphere variations generate mostly continuous unimodal distributions of EECs, monetization variation commonly creates multimodal distributions (subplots c and d) due to distinct monetization factors that are particularly far apart.

4. Discussion

Previous studies [14, 20] have highlighted the high external costs of energy relative to production costs; our findings confirm the technological ranking in terms of relative external costs but often significantly exceed prior external environmental cost estimates. This might be due to several developments: First, impact pathway modeling has advanced and thus covers more pathways and substances. Second, in our approach we cover all available impact categories, in contrast to the existing literature [14, 20]. Lastly, monetary values of non-market goods (human health, ecosystem quality) might have increased over time faster than prices have inflated. Such a longitudinal study has, to our knowledge, not been conducted, but would be a significant addition to the understanding of external costs.

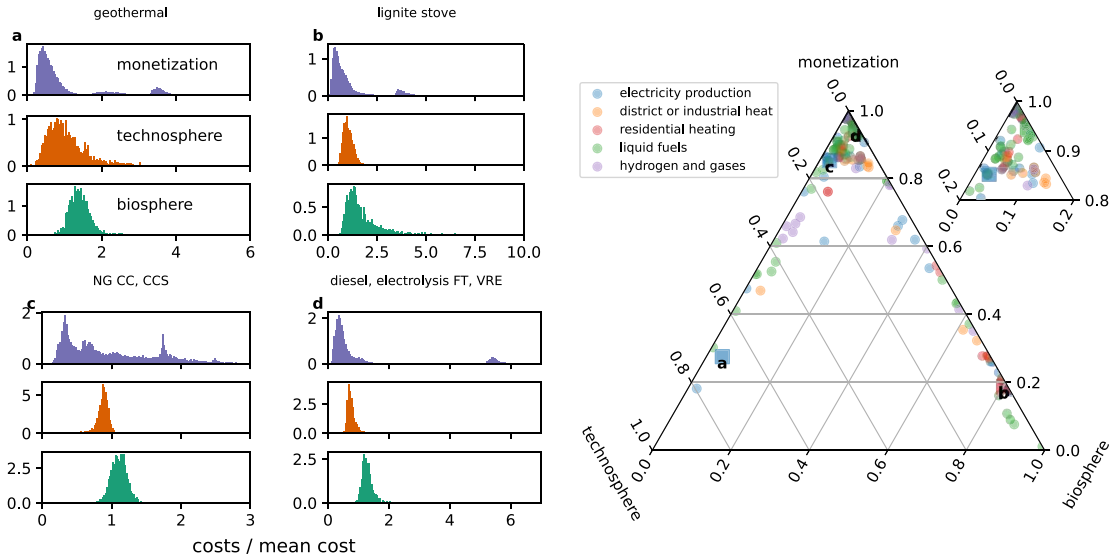

Figure 7. Sensitivity analysis. Variability in monetization factors leads to multimodal cost distributions, and dominates the sensitivity compared to uncertain techno- and biosphere exchanges. (**a**) - (**d**) Distribution of EECs resulting from separate variations of the monetization, technosphere, and biosphere. (**e**) Sensitivity scores (normalized variances of each distribution) for all technologies. The example technologies from the left are marked with bigger squares. Abbreviations: **CC** combined cycle; **CCS** carbon capture and storage; **FT** Fischer-Tropsch; **VRE** variable renewable energy.

We find that technology rankings are quite robust against the choice of monetization approach, but total cost and shares of impact categories vary drastically. This underscores the importance of employing multiple monetary valuation approaches to better understand external costs.

In our analysis, coal power bears the highest external costs among electricity generation technologies, adding to the literature documenting its devastating environmental impacts [65–67]. The external costs of renewable and nuclear power is much lower —aligning well with previous studies and reviews [14, 18, 20, 22, 67]. We additionally incorporate external costs of battery storage for variable renewables. The mark-ups are significant for solar PV (~ 25 %), but it still remains a technology with low external costs. The climate benefit of liquid biofuels compared to conventional fossil liquids is outweighed by land-use-mediated ecosystem damages, a trade-off pointed out in earlier studies [68]. By estimating all impacts in a standard monetary unit, we can quantitatively express co-benefits and trade-offs and rank environmental impacts in terms of their societal costs. Such rankings are valuable for future research as they inform the prioritization of more comprehensive assessments of the impact categories. Overall, we observe a strong correlation between air pollution and climate change impacts. On top of land use costs, costs related to toxicity also feature strongly for many technologies, with fossil fuel (especially coal) combustion [67] and treatment of mining-related wastes as the main sources. We also compared the variation introduced through monetization with that from technological uncertainty for the first time and found the former far more relevant. Future studies should also attempt to add uncertainty from the impact assessment step into this comparison.

Using monetary valuation presupposes that we can put price tags on human health and ecosystem services; it is essential to note that this is a normative standpoint [69]. Different monetization methods, while being very helpful in reducing the dimensionality of impacts and making internalization into economic models feasible, add another level of variation and further value judgments into the analysis. We address this aspect by using several monetizing perspectives, stating which results are perspective-specific, and communicating the underlying assumptions of the perspectives as best as possible. To remain valid for transparent decision-making, future monetary valuation research needs to be more explicit about their value assumptions (see also [24]). The valuation of some impact categories (e.g., land use) has been much less studied than others (e.g., air pollution), and using several monetization perspectives (as opposed to a single central estimate) only partially addresses this research gap. Regional differentiation is another gap for monetization factors, life-cycle inventories, and impact assessment factors. In this study, global average LCIA characterization factors were used. Regionalized characterization methods exist [70], but LCA software and data standards have not kept up with these developments [71]. Further, the characterization factors used here are not prospective, but including future trends will have crucial effects on assessing impact categories such as water scarcity [72] or biodiversity, as climate change might in the future surpass land use as the driver of biodiversity loss [73, 74]. For our results, that might imply a shift in the climate change-biodiversity trade-off, making liquid biofuels more desirable. Thirdly, some impact categories are poorly understood, even on the biophysical level (before monetization). For example,

biodiversity impacts mainly account for species diversity [75], and marine impacts are only starting to be integrated [76]. Lastly, impact pathways typically include some degree of non-linearity [77], e.g., through the interaction of several impacts. Yet, LCIA reduces them to a linear relationship between environmental flows and impacts. The same is true for the monetization step. The prospective LCA methodology covers significant technological developments. Still, some advancements relevant to energy supply are out of scope, like potentially cleaner fertilizer production or more environmentally sound mining practices in the future.

Monetization methods use different discount rates to evaluate costs of impacts occuring in the future. Discount rates have been the basis of long-standing discussions in the academic literature, in the context of climate change mitigation famously in the debate between Stern and Nordhaus [7]. As Arendt points out [24], choices of discount rates in monetization methods are often not transparently documented. Therefore, any differences in monetization perspectives that we observe might in part be due to (intransparent) discount rates assumptions that we inherit from the underlying monetization methods. Future research on external environmental costs would benefit greatly from making the choice of discount rate explicit, consistent across impact categories, and showing the effect of varying discount rates. Additionally, most climate impacts are econometric estimates based on historic data; the existence of tipping points in our climate system might actually mean that climate change impacts are underestimated [78].

We consider only external *environmental* costs; to assess the sustainability of energy technologies more broadly, impacts on the social dimension also need to be considered. Additionally, an integration of external environmental cost assessments and the existing literature on subsidy schemes in for renewable energy systems, such as feed-in tariffs or R&D subsidies [82, 83], would provide a more complete picture on the techno-economics of sustainable energy systems.

In our calculation of additional cost of battery storage, we derive the necessary parameters from REMIND or take exogeneous assumptions. For a more detailed analysis, one could integrate a more granular power system with dispatch. Also more generally, this would help to understand the influence and external costs of storage and transmission infrastructure (also for hydrogen), thus extending the system boundaries beyond where we set them in this study.

5. Conclusion

We conclude with the following remarks and policy implications.

- We perceive the plurality of monetization perspectives not as a hindrance to sound policy advice but as a factor enabling more informed and transparent decisions [84]. Comprehensively quantified external costs give justice to impacts that otherwise run the risk of being mainly ignored —and, in light of our comparison of external and production costs, these impacts weigh too heavily to remain ignored. Policies must be holistic to avoid other bottlenecks or ecological crises when attempting to mitigate the climate crisis.
- In the power and heat sectors, we see that transitioning from fossil fuels to renewables does not only critical for decarbonization, but also for reducing externalities across environmental impact categories. Storage requirements might increase environmental burdens for solar PV, but likely only in competition to the other low-cost techs. Phasing out coal is most crucial.
- Achieving lower external costs in the other sectors (liquids, gases, and hydrogen) is more challenging. The climate benefits of bio- or synfuels might need to be paid for dearly by ecosystem impacts through land use and exposure to toxic emissions. Good environmental performance also hinges on the availability of the 'right' feedstocks to avoid fossil lock-in. This might enhance the case for policy instruments such as green lead markets, and underscores the priority of cleaning up electricity supply.
- In line with previous studies [32, 85], our results on key transformative energy technologies (figure 5) indicate a shift from direct to upstream impacts. Incentives for greening supply chains will be of great importance alongside securing the material needs of an energy system in transition and preventing the build-up of neo-colonial structures. These questions about the distribution of impacts, costs, and profits necessitate a refined geographical and socio-economic analysis.

[7] In his report, Stern took a prescriptive approach, and arguing that any non-zero pure rate of time preference is unethical arrived at a discount rate of 1.5% [79]. Nordhaus in turn challenged that assumption and took a descriptive approach of defining the discount rate to be the rate of return on capital investments, yielding discount rates of at least 4% [80]. The debate is far from resolved; in his philosophical treatment [81], Heilmann argues that the choice of discount rates remains 'at heart an ethical problem'. The developers of the 'Environmental prices' method [48] that we use to construct the damage cost perspective in this paper choose a discount rate of 3% for climate change impacts —falling right in between the values proposed by Stern and Nordhaus.

- Finally, external costs are present and high (compared to production costs) for *all* technologies. This stresses the importance of reducing overall energy demand, e.g. through R&D investments to increase energy efficiencies, improving demand-side management, or policies enabling individual and societal life-style changes.

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Declarations

Supplementary information

This article has 4 associated supplementary tables, provided as *.csv* files, as well as 6 supplmentary figures, provided as a *pdf*.

Data availability statement

The data cannot be made publicly available upon publication because they are owned by a third party and the terms of use prevent public distribution. The data that support the findings of this study are available upon reasonable request from the authors.

Competing interests

The authors declare no competing interests.

Code availability

The Zenodo repository https://doi.org/10.5281/zenodo.14525954 also includes all code for running the analysis and creating the figures.

Author contribution

D.B. and G.L. devised the research question and then conceived the study with consultation of R.A., R.S., and C. B.. R.S. further developed the *premise* tool for prospective LCA. R.A. and D.B. jointly developed the approach for monetary valuation. D.B. wrote the analysis code and created the figures. D.B. wrote the manuscript with contributions from all authors.

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