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Global emission hotspots of coal power generation

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Coal power generation is a primary cause of greenhouse gas (GHG) and toxic airborne emissions globally. We present a uniquely comprehensive inventory of CO₂, methane, particulate matter, sulfur dioxide, nitrogen oxides, and mercury emissions for 7861 coal generating units including their supply chains.

Total GHG and toxic substance emissions are largest from China, US, India, Germany and Russia (together >64% per pollutant). Overall supply chain contributions are below 19%, but exceed 75% for individual units and pollutants. Methane emissions from underground coal mining offset Chinese coal power plant efficiency advantages in comparison to India. Health impacts, as quantified by regionalized life cycle assessment, are highest in India and parts of eastern and south-eastern Europe due to lack of modern flue gas treatment, and in China due to widespread coal power generation. Deployment of state-of-the-art flue gas treatment, driven by local emission limits, can mitigate health impacts in India and parts of Europe while it is already largely used in China and the US. Phase-out of the 10% most polluting coal power plants (by capacity) would reduce coal power GHG emissions by 16% or human health impacts by 64%, respectively.

Ambient air pollution is the primary environmental threat to human health globally (1). Combustion of fossil fuels, including coal, is the main anthropogenic contributor (2, 3). Coal combustion releases CO₂ as well as fine particulate matter (PM_{2.5}) and its gaseous precursors sulfur dioxide (SO₂) and nitrogen oxides (NO_x) into the air. CO₂ drives climate change, while PM_{2.5} can enter human lungs and cause ischemic heart disease and stroke, chronic obstructive pulmonary disease, lung cancer among adults and acute lower respiratory tract infections among children. Additionally, airborne SO₂ from fuel sulfur can cause acid rain (2). NO_x emissions are toxic and contribute to global warming and terrestrial acidification. Finally, mercury, with its high toxicity, long-term persistency and potential to affect children's brain development, is released to air by coal combustion (4). Once deposited, it may build up in the aquatic food chain and be ingested by humans. Reducing impacts from these emissions is a global priority, but further industrialization, especially in China and India, poses the risk of aggravating the situation instead.

During the transition to renewable, more sustainable alternatives, coal combustion techniques need to improve by targeting the causes of pollution impacts. Existing studies (5–11) however, are unable to identify the most effective mitigation options on plant-level due to lack of spatial resolution, supply chain coverage and regionalized quantification of health impacts (discussed in detail in the supplementary information

(SI)). Thus, there is a clear need for a global model that combines technological standards and supply-chain characteristics of each power plant and allows for the derivation of reliable decision support for the mitigation of human and environmental exposure. To tackle this issue, the present work provides the most detailed global picture of site-specific airborne coal power plant emissions, including mining, coal preparation, trade and transport, mechanistic power plant modeling and flue gas treatment. Emissions are quantified for CO₂, CH₄, PM_{2.5}, PM_{10-2.5}, PM_{>10}, SO₂, NO_x, and three species of mercury (Hg⁰, Hg²⁺, and Hg_p), which have been identified as key pollutants (7, 9). Health and climate change impacts are quantified, technical abatement potential is analyzed in an unprecedented level of detail, and key policy implications are provided.

Global coal supply and emissions. Emissions from coal power plants depend on coal qualities and combustion mechanics. We therefore first connect the coal quality (broken into four ranks) associated with individual coal mines to power plants using a transport model. This model uses plant-specific coal origin data where possible, and otherwise uses coal that minimizes transport costs with respect to national and province-level coal trade data. The coal ranks are comprised of *anthracite*, *bituminous coal*, *subbituminous coal*, and *lignite* (in order of decreasing calorific value). Global patterns of coal transport from mines to power plants for the year 2012 are presented in figure 1.

The main coal mining regions are located in China and the US. However, Indonesia, Australia, Russia, South Africa, and Colombia also produce large amounts of coal. Mining activities contribute significantly to total greenhouse gas (GHG) emissions (quantified with GWP100 (12), 5 %) and NO_x (9 %) emissions of coal power generation (details in the SI). Methane emissions from underground coal mining harm the climate and represent the waste of a resource that could be used for energy supply (13). For countries without proper mine management, these emissions can account for up to 13 % of their total GHG emissions along the coal power value chain (figure 2b). NO_x emissions are caused mostly from the operation of mining equipment and blasting (14) and are in a few extreme cases responsible for up to 68 % of total NO_x emissions from coal power generation. Additional emissions are caused by uncontrolled coal fires but reliable data on these is not available (15) and hence, is not included. Coal washing is practiced in major coal-producing countries like China, Australia, and the US, and for bituminous export coal around the world. This improves coal quality by increasing the calorific value per weight and decreasing impurities like ash, sulfur and mercury content but increases water-borne emissions. Effects of coal washing on power plant emissions are calculated per country, coal rank and destination.

Coal transport is largely due to the international trade of bituminous coal, which is favored due to its low ash content and high calorific value. In contrast, lignite is of low quality and is therefore hardly traded internationally. Global coal

Fig. 1. Globally modeled coal transport amounts from mines to power plants for four transport modes in 2012. Line width is proportional to transported mass. Details show parts of North America (a), Europe (b) and Asia (c) magnified by a factor of 3.7.

transport equals 5800 bn tkm which amounts to 19 % of the global sea and ground transport in 2012 (16). About 66 % is transported via ocean ships, 30 % via train and the rest is transported via truck or barge. Ocean transport is largely driven by the Chinese coal import over long distances from Australia, Indonesia, South Africa and North America (figure 1, detail (c)). Other main coal ocean transport is from Colombia and Russia to Central Europe (figure 1, detail (b)), as well as from Australia, Indonesia and South Africa to Europe, Japan and India.

As China is the largest consumer of coal globally (39 wt.-%), much of the transport happens inside China itself with 80 % by train, 13 % by barge, and 7 % by truck. Chinese coal transport generally involves northern coal mines shipping southward or central and southern mines shipping eastward. The US is second-largest consumer globally (19 wt.-%). The main regions of resulting US rail transport are located in the center of the US and close to the Appalachians (17). The distribution of 232 Mt of subbituminous coal from Wyoming eastwards is responsible for the main coal train transport by tkm globally in 2012 (figure 1, detail (a)).

Transport emissions are mostly from long-distance transport via ocean-going ships and medium-distance transport via train. The ships mostly add to the total emissions of SO₂,

NO_x and PM_{2.5} via their combustion of fuel oils for propulsion. These emissions may not be critical in remote areas, but they can have substantial health and ecological impacts close to the shore. Ocean transport emissions are particularly high along the Chinese coast. On average, ocean transport accounts for 1.7 % of SO₂ and PM_{2.5} emissions, and for 4.2 % of NO_x emissions (figure 2c-e). These emissions could be reduced by the use of higher quality ship fuel. Fuel combustion is also the main cause for high NO_x and PM_{2.5} emissions from rail transport (18) that contribute on average 4.5 % and 2.6 % to coal power life cycle emissions, respectively. As freight is mostly transported with diesel engines (19), an increase of railway electrification may be the most effective emission reduction option. Transport emissions from trucks and barges are only relevant in individual cases. Other emissions from coal handling and storage as well as transport via coal slurry pipelines and belt conveyors are not included in this work due to lack of data. These are expected to have minor impacts on overall results (20).

Globally, coal power plants are the main source for releases of the five priority emission categories GHG, PM_{2.5}, SO₂, NO_x and Hg into the air along the coal supply chain (figure 2). This is due to components of the fuel (for CO₂, PM_{2.5}, SO₂, fuel NO_x, and Hg), but also due to complex reactions during

Fig. 2. (a) Coal mine, transport and coal power plant emission contributions to total emissions of coal power generation, as well as (b-f) contributions of individual coal generating units (y-axis) to pollutant emissions (sorted from largest power plant contribution to smallest). Contributions per unit from (b-f) add up to the sums in (a). GHG emissions given in GWP100.

combustion (again for PM_{2.5} and Hg, as well as thermal and prompt NO_x) and flue gas treatment. Among all pollutants, coal power plants are on average responsible for 81 % or more of the airborne supply chain emissions, and in the case of Hg for nearly 100 % (figure 2e). Mercury emissions at power plants are dominated by elemental mercury (69 %) with smaller shares of ionic (30 %) and particulate mercury (1 %). More measured data on emissions per coal power plant is needed though, in order to reduce the considerable uncertainties associated with modeling emissions, especially in developing regions of the world and along the coal supply chain where emissions are not reported. A discussion of the data limitations, including those associated with mining emissions in particular, is provided in the supplementary information.

Variability of greenhouse gas emissions. Major differences of GHG emissions, including transport and mining, per net generation of electricity are shown in figure 3. The mean cradle-to-gate emission intensities per country are within a narrow range around the generation-weighted global mean of 1.13 kg CO_{2e} kWh_e⁻¹ with a net generation-weighted standard deviation of ±0.06 kg CO_{2e} kWh_e⁻¹. However, the generation-weighted standard deviations among individual generating units is substantially larger than between countries, globally at ±0.14 kg CO_{2e} kWh_e⁻¹ due to the existence of particularly efficient as well as polluting units in most countries. For reduction of GHG emissions per kWh electricity, the latter should be targeted with priority.

Part of the GHG emissions per electricity generation is explained by net electrical efficiencies, which relate net electricity outputs to fuel heat inputs. These are anti-correlated to GHG emissions with a Pearson correlation coefficient *r* of -0.89 as high GHG emission intensities coincide in many cases with low efficiencies. These efficiencies are particularly low for lignite combustion, which is why countries with larger shares of lignite in their fuel mix, like Germany and Russia, perform worse in terms of GHG emission intensities compared to China, India and the US. However, Germany and Russia have a significant share of combined heat and power production from coal. When allocating the GHG emissions between these two co-products based on exergy content, GHG emissions per net electricity generation from coal power plants in Germany and Russia are lower than in China (figure 3). The US coal power plant emissions per net electricity generation are among the lowest in the world but cause the second largest GHG impacts from coal power generation globally after China due to the large amount of US coal power generation.

The overall efficiency of coal combustion is related to plant age. Old coal power plants (constructed before 1970) produce 11 % of net electricity from coal and generally have higher median emission intensities than newer ones due to higher median net electrical efficiencies (supplementary figure 8). For coal power plants constructed between 1970 and 2000, the median net electrical efficiencies are similar (around 31.4 % based on lower heating values). A new trend can be observed for coal power plants constructed since 2000. GHG emission intensities have continuously dropped, largely due to the additions of large, highly efficient super-critical and ultra super-critical coal power plants in China and India. However, less efficient coal power plants are still being deployed in many parts of the world (21, 22).

Individual coal properties play a vital role for emissions.

Bituminous coal has the highest calorific value per carbon content while anthracite, in contrast, has a higher energy content per weight but lower energy content per carbon. Thus, combustion of anthracite emits, on average, 7 % more CO₂ per kWh than bituminous coal. Subbituminous coal and lignite also cause higher CO₂ emissions during combustion than bituminous coal (on average 15 % higher), in part due to their high ash and water content and associated lower boiler efficiencies. The global share of lignite consumption has decreased steadily since the 1990s, thus contributing to reduced GHG emission intensities, but in absolute terms, its consumption remained stable while the consumption of the other coal ranks more than doubled (23).

Another factor in GHG emissions is the type of mining technology involved. When comparing emission intensities of China and India per kWh of electricity, the latter are lower despite lower efficiencies and a similar coal consumption mix. Mining methods in India and China differ substantially: 95 % of Chinese coal is mined underground while 81 % of coal in India stems from opencast mining (24). Underground mining releases higher amounts of methane which is why total Chinese coal bed methane emissions are more than 30 times higher than the Indian ones (25). As a result, mean Chinese mining emissions are 0.08 kg CO_{2e} kWh_e⁻¹ as opposed to 0.02 kg CO_{2e} kWh_e⁻¹ in India. This offsets the Chinese power plant net efficiency advantage of 5 % by a factor of four.

There is a range of measures that can be applied to reduce GHG emissions from coal electricity generation. Efficiency improvements would be one such option, with improvement potentials of up to 15 % at individual plants, but at high costs as shown by (26). Another option may be a (partial) fuel switch of coal power plants to natural gas or biomass where the required infrastructure is locally available. This has been practiced in the US recently due to a cheap local supply of alternate fuels (27, 28). Natural gas combustion may save up to 50 % percent of GHG emissions at the power plants (29), while biomass has even lower net CO₂ emissions. A future option may be carbon capture and storage (CCS), which sequesters CO₂ from flue gas and stores it underground, but there are technical challenges, a lack of financial incentives and an increase in fuel demand, which have to be overcome (30). CCS has side-effects like a potentially toxic amine slip into the environment, and an increase of emissions along the supply chain due to additional coal inputs from efficiency losses, which make re-assessment of environmental and health impacts necessary once this technology reaches maturity.

Non-GHG emissions and health impacts. Unlike the global effects of GHG emissions, the health effects caused by primary PM_{2.5}, SO₂, NO_x and Hg²⁺ are highly dependent on the location of the emission. They result from pollutant inhalation by humans (31, 32) and in the case of Hg²⁺ from ingestion of fish. The local population density surrounding the point of emission is thus one of the dominating parameters that determine absolute health impacts (31). We use regionalized life cycle impact assessment (LCIA) to quantify the health impacts from human exposure. Four regions show the highest health impacts globally: India, eastern China, parts of Europe and the US (figure 4).

PM-related health impacts in this study cause the loss of 8.8 MDALY a⁻¹, which is 11.2 % of the total DALY loss due to ambient PM in 2012 according to (33). The contribution of

Fig. 3. Left axis: Mean CO_2 emission equivalents of coal power plants per net electricity generation including mine and transport emissions without heat allocation. The widths of the bars are proportional to net generation, the colors indicate coal ranks. Black lines indicate GHG intensity distributions of individual coal power plants for major countries. GHG emission intensities above $2 \text{ kg CO}_2\text{e kWh}_e^{-1}$ are not shown. Right axis: The points show the generation-weighted mean electrical power plant net efficiencies per country based on lower heating values (LHVs), and the error bars show the weighted standard deviations of these efficiencies. Dashed lines show GHG emission intensities in case of exergy allocation between electricity and heat. Country abbreviations: RU - Russia, PL - Poland, DE - Germany, CN - China, IN - India, AU - Australia, ZA - South Africa, US - United States, JP - Japan.

SO_2 (6.1 MDALY a^{-1}) stands out in comparison to those of NO_x (1.7 MDALY a^{-1}) and primary $\text{PM}_{2.5}$ (1.0 MDALY a^{-1}). This is a result of the high sulfur content of coal in combination with unabated SO_2 emissions and a large local exposure due to high population densities in major coal-burning countries like India (figure 4). Hg^{2+} human toxicity impacts are calculated to be lower than PM impacts at 0.2 MDALY a^{-1} . It should be noted that impacts of Hg^{2+} are highly uncertain (around three orders of magnitude), as the exposure pathway via the food chain is subject to major model and parameter uncertainty (34). Hg^{2+} impacts occur over a long time frame, while respiratory effects are more immediate, which makes direct comparison difficult. Furthermore, a full global assessment of mercury-related health impacts is not yet included in (1).

Indian coal power plants cause the highest share of global coal combustion health impacts (figure 4b). One of the main reasons is the absence of SO_2 control in India. Similarly, a lack of NO_x control results in high NO_x pollution impacts. Finally, Indian coal is characterized by high mercury content with large regional variability (35), which also leads to increased mercury health impacts. Neither of these substances has been regulated at Indian coal power plants until 2015 (36), with the newly introduced regulation still not having resulted in major changes as of now (2018) due to implementation problems.

The Chinese situation is different. Modern coal power plants have been built in recent years, and policies restrict operation of plants close to major cities (37). Nevertheless, $\text{PM}_{2.5}$, SO_2 and NO_x emission and health risks are high, despite the installation of modern electrostatic precipitators (ESP), wet desulfurization technologies and low NO_x burners,

due to the total amount of power generation from coal and generally high population densities. In particular, the health impacts from Hg^{2+} are larger than in other regions of the world (figure 4d), since raw coal is heavily contaminated with mercury (the median Chinese mercury concentration is 86 % higher than elsewhere).

Europe has locally high population densities and many coal power plants in its eastern and south-eastern parts lack modern flue gas treatment or use low-quality coal. SO_2 emissions especially are responsible for regionally high health impacts in these regions (figure 4b). Hg^{2+} emissions from some power plants in the United Kingdom are high in international comparison and cause elevated health impacts.

The case of the US is less severe since the population density is lower than in the three aforementioned regions. Still, we find above-average health impacts for SO_2 in the eastern region where bituminous coal with high sulfur content is burned. Also, the share of coal in the US electricity mix is high, so while impacts per net electricity generation are lower than in many other parts of the world, the large amount of coal power generation makes the US the country with the fourth-highest coal power-related health impacts. These impacts are 18 times lower than in India.

Outside these regions, there are individual locations where health impacts are particularly high. This is typically associated with a combination of high population densities and poor flue gas treatment for example around Pretoria in South Africa, Moscow in Russia or Jakarta in Indonesia (figure 4a). These locations should be prioritized for emission reductions, as changes at few generating units lead to major health im-

(a) Human health impacts per coal power plant [DALY/year]

(b) Coal power plant health impacts per region

Fig. 4. (a) Human health impacts per coal power plant, and (b) as impact intensity per coal power production for each region and substance (areas of the bars are proportional to total impacts per region). Hg^{2+} impacts have been calculated with a different approach compared to SO_2 , NO_x and $PM_{2.5}$ and are subject to high uncertainty (three orders of magnitude). Country abbreviations: IN - India, RU - Russia, CN - China, ZA - South Africa, DE - Germany, JP - Japan, US - United States.

provements.

Emission control. A general approach for reducing primary particulate matter emissions is through particulate removal with bag houses. These are up to 91 % more effective at filtering PM_{2.5} particles out of the flue gas compared to current ESPs (38, 39). Bag houses are not used widely outside the US, as PM emission limits typically do not target PM_{2.5} specifically but instead PM₁₀ or total PM (37). Health impact reductions from introduction of bag houses would mostly come from India and China, where primary PM_{2.5} health impacts are highest and cost-benefit ratios for PM_{2.5}-removal are particularly favorable (as shown in the SI). Technical obstacles include an increased pressure drop in the flue gas stream and corrosion problems in the case of high shares of acidic components in the flue gas, but general feasibility is proven by 444 coal-fired generating units operating with baghouses in 2012. Upgrading 10 % of global generating capacity to include bag houses could reduce primary PM health impacts by 47 % (0.5 MDALY a⁻¹) with an additional benefit from reduced particulate mercury (Hg_p) releases. The costs for installing and operating bag houses are similar to those for ESPs (supplementary table 35 in the SI).

The health impacts from SO₂ can be reduced with wet limestone injection. These reductions would come mostly from India where impacts are highest as there is no active SO₂ control measure installed for 92 % of the capacity in 2012 and persists until today (2018). The installation of wet limestone injection at the 10 % most polluting coal generating capacity globally could result in SO₂-related PM health impact reductions of 67 % (4 MDALY a⁻¹). Co-benefits include reduced particulate and mercury emissions (38–40). As a consequence, local PM_{2.5} health impacts would be reduced by 21 % (0.2 MDALY a⁻¹) and ionic mercury human toxicity impacts by 11 % (0.02 MDALY a⁻¹).

Selective catalytic reduction (SCR) is an option for NO_x health impact reduction. This technology is already employed by 1053 generating units in 2012. Utilization at all units can reduce NO_x-related PM health impacts by up to 58 % (1 MDALY a⁻¹), while upgrading 10 % of generating capacity would reduce the coal power NO_x health impacts by 38 % (0.6 MDALY a⁻¹).

While elemental mercury is a pollutant of global concern due to its stability in the atmosphere, ionic (Hg²⁺) and particulate (Hg_p) mercury compounds deposit locally with rainfall and accumulate in the food chain by contaminating fish (41). Associated health impacts may be reduced from wet flue gas treatment or particulate removal (40), but the mercury then may enter the waste water and go to the sewage sludge (42) or end up in the retained fly ash. Its fate then depends on the individual use of sludge and fly ash, and is not fully understood. Injection of brominated pulverized activated carbon (PAC) is an option to adsorb elemental mercury from flue gas with a positive side-effect on Hg²⁺ (as well as Hg_p). This is practiced by 160 generating units in 2012 and may further reduce total mercury emissions by up to 39 % while also decreasing Hg²⁺ human toxicity impacts by 4 % (0.007 MDALY a⁻¹). An alternative to upgrading flue gas treatment can be the phase-out of power plants in particularly sensitive areas and replacement with renewable alternatives. Health impacts depend strongly on local conditions at the power plant so the phase-out of few highly polluting units can lead to major health improvements.

To reduce corresponding health impacts by 50 %, the dirtiest 9 % of global generating capacity for primary PM_{2.5}, 4 % for SO₂, 12 % for NO_x and 15 % for Hg²⁺ would have to be phased out, respectively (as shown in detail in the SI).

The deployment of advanced flue gas treatment and the phase-out of old power plants is largely driven by local emission limits and their enforcement (39). In light of the plant-specific differences in health impacts per amount of pollutant release, we suggest to introduce plant-specific or region-specific legislation for pollutants with mostly local impacts. However, detailed site-specific fate and exposure modeling has to be conducted to set adequate emission limits and to avoid burden shifts between substances, locations, environmental compartments and impacts. The present paper allows to prioritize regions and power plants for more detailed assessments from a global perspective.

Materials and Methods

Overview and system boundaries. The emission inventory model for coal power plants in this work consists of three parts. It covers mine emissions, transport emissions and plant emissions. As such, it includes the airborne substance releases of carbon dioxide (CO₂), particulate matter in three size classes (PM_{>10}, PM_{10-2.5}, PM_{2.5}), sulfur dioxide (SO₂), nitrogen oxides (NO_x), three species of mercury (Hg⁰, Hg²⁺ and Hg_p; from plants only, else as total Hg), and methane (CH₄, from mines only) which are responsible for the main environmental impacts of coal power plants according to previous studies (7, 39). Airborne emissions during construction and de-construction of infrastructure are usually below 0.1 % compared to emissions in the operation phase as presented in (7). Thus they were not modeled explicitly in the present study.

The system boundaries applied in this work are visualized in supplementary figure 1.

Model structure. The model first sets up the databases for power plants, mines, transportation grid, coal samples, trade data, and reported emissions. In a second step, power plants and mines were connected by transport routes, and individual coal mixes were retrieved per power plant unit. A coal washing model was applied to account for coal preparation at the mines. Missing power plant data was imported from several databases if available, retrieved from satellite images or estimated and will be explained in more details in the following sections. Then, power plant emissions were imported from external data sources where possible, while for the remaining units, emissions were calculated based on technical power plant specifications, flue gas cleaning and coal properties. Regional mine and transport emissions were retrieved from ecoinvent v3.3 (cut-off system model) (19) with individual improvements on mine emissions to capture regional differences in more detail. Then, human health impacts for PM_{2.5}, SO₂, NO_x and Hg²⁺ emissions were calculated. An overview of the model structure is shown in supplementary figure 2. We developed the model in the programming language R (version 3.4.0) with the R packages listed in supplementary table 19.

Coal mining. A coal mine database was created from USGS minfac database (n=355) (43), Chinese USGS mine data (n=2440) (44, 45), the Australian NPI database (n=114) (46),

the Canadian NPRI database (n=19) (47), European E-PRTR database (n=342) (48), two EIA US coal mine data sources (n=802 and n=460 (49, 50)) and from satellite imagery (n=705) based on the data reported in (51) (supplementary figure 4). Manually collected mine data is presented in supplementary table 12. Additionally, a consistent coal sample database with 9456 coal samples was created from the three USGS databases NaCQI (52), WoCQI (53, 54) and COALQUAL (55) as well as German lignite samples (56) by harmonizing data format, merging databases, converting units and removing duplicates. These also contain coal types for steel-making which could not be excluded as a clear distinction was not possible. Median values of coal constituents that have been obtained are shown in supplementary table 9. Lower heating values (LHVs) for the coal samples were calculated based on the enthalpies of coal constituents as described in the SI. Then, coal ranks were assigned. The four coal ranks are *anthracite*, *bituminous coal*, *subbituminous coal*, and *lignite*. The first two comprise hard coal with high calorific value, the latter two brown coal with low calorific value. This classification is internationally not consistent since exact criteria differ. Wherever needed, the most recent US guideline for classification (57) was used for consistency reasons and because most coal samples used in this study were collected and analyzed by the US Geological Survey (USGS). In case no coal rank could be assigned due to missing data, the originally reported coal rank was used instead. In the cases where US ranking standards were not followed, the ranks were converted with the rank correspondence shown in (58). The ranks of coal samples were also used to assign coal ranks to coal mines with missing rank information based on the closest coal sample contained in the coal sample database within 100 km (great circle distance, applied to 17% of coal mines). Remaining missing coal ranks of mines were filled in with coal ranks of the closest coal power plant within 100 km as coal is often burned locally (also by great circle distance, 6% of coal mines). Still remaining coal mines without coal rank (3%) were assumed to produce bituminous coal as this is the dominant coal type globally (59).

To estimate emissions from mining, regional emission data per kg coal mined was taken from ecoinvent v3.3 (cut-off version) (19), which provides coal mine data sets for Australia, China, Indonesia, Poland, Latin America, North America, Russia, Western EU, South Africa, and the rest of the world for non-lignite coal while lignite is covered with a European and a rest of the world dataset (60). Methane emissions for non-lignite coal were updated based on national coal mining emissions (25) for the years 2010 to 2012 (taking the most recent available one) divided by corresponding production amounts from (61) (assuming these amounts as well as the original ecoinvent numbers referred to the production of washed coal). This was based on the assumption that lignite comes mostly from open pit mining with much lower methane emissions (as shown in (60)). As an exception, Turkey also mines lignite underground, which is why we divided total Turkish methane emissions by total Turkish coal production, and created both a lignite and a non-lignite coal mining dataset for Turkey. Emissions other than methane were taken from the ecoinvent region the countries are part of, so for example the US would have specific US methane emissions and North American PM emissions. Thus, updated and new non-lignite mining datasets could be generated for Argentina, Australia,

Brazil, Canada, Chile, China, Czech Republic, Germany, India, Italy, Malaysia, Mexico, New Zealand, Norway, Peru, Poland, Russia, Spain, Turkey, Ukraine, Great Britain, the United States, and Viet Nam, while also an additional lignite mining dataset was generated for Turkey (details including the calculated methane emissions per amount of coal are presented in supplementary table 18). Only on-site emissions (mining, blasting, diesel burning, and infrastructure emissions) were included and Chinese emissions from coal fires were removed for consistency reasons as they were not included in any of the other datasets. To make methane release comparable in terms of climate change impacts, these emissions were converted into CO₂-equivalents based on IPCC guidelines by multiplying them with a factor for global warming potential of 28 for a 100-year perspective (12).

Coal washing. Coal washing is a set of mechanical and chemical processing steps, that aims at improving coal properties before transportation to the power plants. Specifically, it splits the coal into at least two fractions. The largest fraction has lower ash, sulfur and mercury content and higher calorific value and is sold, while any smaller, less pure fractions have lower economic value and are thus disposed or burned locally. Details on the modeling of coal washing are presented in the SI.

Transportation. The transportation model in this work serves two purposes. It allows the estimation of direct emissions and locations that come from transportation of raw coal from mines to power plants, as well as the estimation of the average blend of coal that goes into each power plant. Thus, a global coal transportation network was created that contains potential routes from mines to power plants via rivers, oceans, rail and road. River transport other than on main rivers were beyond the scope of this study. Belt conveyor transport as well as coal pipeline transport were also not modeled in this study since no sufficient information on its global use was available. Belt conveyors cover short distances up to a few kilometers and thus have little impact on total transport emissions from their energy demand. Fugitive emissions from belt conveyors though, as detailed in (62), could be a factor to include in future studies. Coal pipelines have existed in few regions of the world (for example the *Black Mesa* and the *Consolidation* pipelines in the US) but play no important role anymore. Thus, only data on the global rail network as well as shipping routes and major rivers for coal transport including coal ports was collected (details in the SI). The road network was approximated as also described there.

In a next step, the unlinked way points (mines, power plants and ports) and the individual transportation networks per mode were linked to create the full global coal transport network. The linking algorithm is shown in supplementary table 2. It consists of 21 steps, each of which either creates links between way points or removes previous links where more detailed data is available. A power plant for example may connect directly to a nearby ocean way point but in case there is an ocean harbor nearby, direct connections between power plants and ocean way points are removed and instead the connection is made via the coal port. This procedure ensures connectivity where local coal harbors at power plants are not known but allows for more accurate route estimates if they are known. Distance thresholds of 10 km, 100 km and 1000 km were set as

upper limits to avoid unrealistic connections. One way point could be connected to all way points within these distance thresholds or just to the closest one. This procedure automatically created 4699 road segments, 264 river segments and 358 ocean route segments out of a total 433 405 segments. The transport mode per connection type is also presented in supplementary table 2. After the linking procedure, the complete network was turned into a graph where edges represent the transport pathways and nodes the way points. Edge weights were based on multiplying the edge length calculated from their coordinates as great circle distances with transport cost per mode of transport as reported for truck, rail and barge in (63) and for ocean vessels as mean value of coal ocean transport costs from (64, 65). As a result, costs for truck transport were $0.08417 \text{ USD tkm}^{-1}$, rail transport costs $0.01544 \text{ USD tkm}^{-1}$, barge transport costs $0.00899 \text{ USD tkm}^{-1}$ and ocean transport costs $0.00262 \text{ USD tkm}^{-1}$. These represent Chinese conditions and in case of ocean transport global conditions, but ratios between transport modes were found similar to US conditions in (66, 67). Since these two are the main consumers of coal (China 47.7% and US 10.6% (59)), this seems to be a reasonable assumption.

To then satisfy the coal demands of the coal power plants, all plants were connected to mines based on the available data per plant. That included individual plant-level information on specific US mines (50), individual mine data per plant from (68), coal origin provinces for US power plants (69), domestic, imported or both types of coal per plant from IEA coal power atlas (70), province-level information for Chinese power plants from (71), IEA average coal mixes per country and coal rank for OECD countries (59), own calculations of country mixes per coal rank for non-OECD countries with major global relevance in the coal sector (China, India, Russia, Indonesia, South Africa, Colombia, Vietnam, Kazakhstan, Ukraine, Philippines) from (72, 73) and finally the multi-regional input-output (MRIO) database exiobase 2 for the rest of the world (supplementary figure 6) (74). The shares for non-OECD countries were calculated from the production data per coal rank in 2012 minus the respective exports from (73) and adding the imports per coal rank from (72). Wherever production amounts were zero due to data gaps in (73), we calculated the minimal domestic production from the total coal demand per coal rank we knew from our coal power plant database (for example for anthracite production in China). Results are shown in supplementary tables 26–29. For Chinese coal power plants without plant-specific coal source data, we combined these shares with province-level coal origin data (71) for the domestic coal consumption. Countries without explicit representation in exiobase 2 were assumed to have coal demands representative of the larger region they are located in (Chile for example was treated as *Rest of World America*).

To calculate the cheapest reachable mine per power plant within each supplying mining region, the Dijkstra algorithm (75) was used. Only mines offering the required coal rank (anthracite, bituminous, subbituminous, lignite) were allowed to supply coal to a power plant. For power plants that blended coal, weight shares were based on the average coal supply to that region (again from IEA data (73) for OECD nations, own calculations for large coal-consuming non-OECD nations, and from exiobase for the remaining countries (74)).

Specific regional coal properties (lower heating value, sulfur

content, ash content, water content, mercury content, chloride content) were assigned to each coal mine. For anthracite, bituminous and subbituminous coal, these were the median values per province, for lignite the values of the closest coal sample. This distinction was made because low value lignite is typically only supplied to the closest power plants within reach due to transportation costs. For the other coal types, transportation takes place over larger distances, so the share of transportation costs in comparison to total fuel costs is larger and so a more suitable coal might get chosen with little additional cost. Thus, we calculated average coal mixes per province as we considered these more representative. In case there were data gaps for individual properties, these were filled from country-level data of the same coal rank, and if that was not possible also from global data of the same rank. Global average properties were calculated as production-weighted mean values (72–74).

Based on the shares of coal input from different regions and coal ranks, the weighted mean coal properties per kilogram of coal at the plant were calculated. Then, the total coal consumption in kg was estimated with the average lower heating value to satisfy the heat input requirement per plant. The total coal consumption was then split to all routes connecting the mines according to the coal amount shares calculated before. Mine capacity restrictions were not taken into account as data on these were not available globally.

An appropriate ecoinvent v3.3 data set was assigned to each transport route segment based on transport mode and location (selected data sets and regions listed in supplementary table 32; ship transport was global, for barge as well as truck it was specific to Europe and the *Rest of the World*, and for trains it was specific to Austria, Belgium, China, France, Germany, Italy, Switzerland, US, *Rest of Europe* and the *Rest of the World*) (19). It was then possible to quantify the direct emissions along the individual transport routes from the transported coal mass per segment and its length. These emissions do not include indirect emissions such as from manufacture of vehicles, maintenance, or electricity demands as these are not emitted along the transport route.

Power plants. Operational coal power plant units in 2012 were taken from the Platts WEPP database (76) while corresponding locations were obtained from emission registers and related databases (47, 48, 77) or from (78) and (79) (supplementary figure 3). Estimation of airborne emissions per plant unit then followed a two-tiered approach: Preferably, airborne emissions were used from publicly reported data as this is considered to yield the most accurate representation of actual emissions. These were available to varying degrees for China (80–82), India (83), the countries in Europe (48), the US (84), Australia (46), Canada (47), South Africa (85) and Japan (86). Only if data was missing, models for the individual pollutant releases were used. These were based on material balances and, in part, on empirical equations as described in detail in the SI. Flue gas treatment data to quantify emission abatement and resulting emissions per generating unit was obtained from several data sources (76, 80–82) and from satellite imagery.

Quantification of health impacts. Regional characterization factors (CFs) for PM health impacts of SO_2 , NO_x and $\text{PM}_{2.5}$ emissions have been obtained from the life cycle impact assessment (LCIA) method of (87) (choice is explained in the SI).

This LCIA method provides characterization factors mostly on a country level. The coordinates of point sources have been used to identify the CFs with CF maps made available by (88). The global average CF was assigned to all coordinates outside the geographic boundaries in (88) (applies to 254 coal generating units accounting for 5.2% of global coal power generation capacity). Emissions were multiplied with the geographically corresponding CFs to quantify health impacts. Alternative approaches for SO₂, NO_x and PM_{2.5} health impacts from (31) and for PM_{2.5} impacts from (89, 90) have additionally been applied and compared in the SI. For Hg²⁺ health impacts, the characterization factors from (91) have been used based on the classification of urban and rural regions described in the SI.

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Author contributions. CO, SP and SH designed the research; CO collected the data and did the modeling with contributions from CR; CO and SP interpreted the results; CO wrote the paper with inputs from SP, CR and SH.

Competing interests. We declare no competing interests.

Materials and correspondence. Detailed descriptions of existing research, extended methods, additional results, discussion of policy relevance, limitations of this work, uncertainties, and recommendations for future work are available in the SI. The emission dataset per power plant and the source code for the model is available at <http://dx.doi.org/10.17632/dm3rjb9ymc.1> (92). For specific questions or data requests, please contact the authors directly at oberschelp@ifu.baug.ethz.ch.

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