The impacts and challenges of water use of electric power production in China

Jin, Y.

The impacts and challenges of water use of electric power production in China

Yi Jin

金艺

The impacts and challenges of water use of electric power production in China

PhD Thesis at Leiden University, The Netherlands

The research described in this thesis was conducted at the Institute of Environmental Sciences (CML), Leiden University, the Netherlands.

ISBN: 9789051912999

Cover: Yi Jin

Layout: Yi Jin

Printing: GVO printers & designers - www.proefschriften.nl

The impacts and challenges of water use of electric power production in China

Proefschrift

ter verkrijging van
de graad van doctor aan de Universiteit Leiden,
op gezag van rector magnificus prof.dr.ir. H. Bijl,
volgens besluit van het college voor promoties
te verdedigen op dinsdag 21 juni 2022
klokke 11:15 uur

door

Yi Jin
geboren te Suzhou city, Jiangsu province, China
in 1992

Promotor:

Prof. dr. A. Tukker

Co-promotores:

Dr. P. Behrens

Dr. L. Scherer

Promotiecommissie:

Prof. dr. E. van der Voet

Prof. dr. H. X. Lin

Prof. dr. A. Ramírez (Technische Universiteit Delft)

Dr. S. Pfister (ETH Zurich)

Dr. R. Wang

Table of Contents

Chapter 1

General Introduction

1.1 Energy security challenges worldwide

Energy is essential to human society and energy security is vital in maintaining many services, from health to education. Energy demand has been growing rapidly around the world over recent decades, due to rapid industrialization and population growth. Meeting this demand is a crucial but challenging task in many regions [1]. Energy infrastructure is inadequate in some developing countries and 11% of the global population still lacks access to a reliable electricity supply [2]. This issue was recognised by Sustainable Development Goal 7 (SDG 7), which calls for universal access to affordable, reliable, and modern energy services by 2030[3].

However, improving the supply and security of energy is challenging for a number of reasons. With the global energy system still largely reliant on fossil fuels, there is a direct conflict with the ambitions to limit climate change[4]. Moreover, an energy system based on fossil fuels also creates other environmental problems, such as biodiversity threats due to extraction of fossil fuel resources; emission of toxic substances, such as PM2.5, during use; and water stress due to both the use of water for cooling in power plants and the impact of hydropower plants on water systems [5, 6]. In recent decades, many studies have focused on the air pollution caused by energy production, while lately the conflict between energy and water resources is also raising concerns around the world [7, 8]. Given that both energy and water are key resources for human development, understanding their nexus is an important step toward achieving both SDG7 and several other, interconnected SDGs. Yet this nexus between energy and water resources is still insufficiently understood, especially for key, highly populous regions, such as China.

1.2 Energy-water nexus

In the past, resource governance mostly focused on single resource categories, such as water, land, or energy, but policymakers and researchers are now increasingly aware of the interdependencies of resources and the need to manage resources from a nexus perspective. The 'nexus' concept was formulated in response to siloed thinking, and emphasizes the examination of critical linkages across resources [9]. The nexus between water and energy is one of the most critical [10]. Analyses of problems and solutions for current and future energy-water nexus challenges are of significant interest to policy and research [11-14]. The energy-water nexus can be investigated in two directions, that is 'energy for water' and 'water for energy'. The former focuses on energy inputs needed at various elements of the water system, including extraction from lakes, rivers, and aquifers, desalination, and water treatment [15]. The latter focuses on the water required at every stage of the energy cycle, from the extraction and processing of fossil fuels to the generation of electricity [16]. Compared with the energy constraints on water, the water constraints on energy are more prominent and have raised many concerns. Water shortages have already impacted national energy systems quite regularly in recent years, forcing reductions in energy supply [17]. This situation is likely to become worse in the future, due to growing national water consumption and climate change [18-20]. In view of this important challenge, the focus of this thesis is the water requirement for energy.

The energy system comprises various energy types, with both primary and secondary energy sources. Primary energy sources are available in many forms, including nuclear energy, fossil energy – such as oil, coal, and natural gas – and renewable sources, such as wind, solar, geothermal, and hydropower. These primary sources can be converted to electricity, a secondary energy source. Electricity is becoming increasingly important, since the ambitions to reduce greenhouse gas emissions are prompting the world to shift to a much more electrified energy system, for instance by replacing the direct use of fossil fuels for mobility and home heating with electricity-based solutions, such as electric vehicles and heat pumps [21]. At the same time, the electric power sector has become the largest water user among all energy types [22, 23]. Water is an essential requirement for operating the global power plant fleet and, as mentioned above, this has knock-on implications for energy security[5, 16, 24, 25]. Power plants face water stress and even reductions in usable capacity if their water withdrawal requirements cannot be met [26]. For example, numerous power plants in Europe had to be throttled back in the summers between 2015-2018 due to water shortages [17]. Understanding the water-electricity nexus is therefore essential to the sustainable development of water and energy systems.

It is worth noting that the impact of using water for electricity production does not end at water resources but often extends to other systems. Water resources are a major carrier of ecosystems. The impact of the increase in human water use on ecosystems is already a deep concern in many regions [27]. The water use of electric power results in freshwater ecosystem impacts caused by its water consumption and pollution [28, 29], which have often been neglected. Assessing the associated ecosystem impacts of the water use of electricity production is an important step toward ecosystem protection.

1.3 Cross-regional transfers of the impacts of electricity production on water resources

The picture is complicated by electricity transmission, which can separate the users from the producers of energy products and services. Depending on the extent of transmission, electricity produced at power plants can be transported over long distances for eventual use by consumers. National and regional grid systems balance the supply and demand and enable the development of power plants that are remote from centers of energy use. Electricity transmission is becoming increasingly important for global energy security [5, 30, 31], but this also creates a situation where the production-related impacts occur in a different location than the electricity use.

This ‘telecoupling’ of impacts via transmission has attracted much attention. Via the electricity grid, electricity users consume ‘virtual’ water, i.e., water used in electricity production; this is also referred to as ‘virtual water transfer via power transmission’ [32, 33]. The virtual water concept, first introduced by Allan (1993), is the water required for the production of goods and services along their supply chains [34, 35]. Water-scarce regions can import electricity to satisfy their domestic consumption instead of producing it locally and can thus conserve their domestic water resources [36], whereas electricity-exporting regions see the water available in their region providing services for other regions. Understanding virtual water use in relation to power transmission is highly relevant for water management and should be assessed in grid planning.

1.4 China’s water-electricity nexus

Globally, electricity generation grew by 1.3% in 2019 – around half its 10-year average. Growth was weak or negative in most regions and mainly occurred in China, which increased electricity generation by 340 TWh (4.7%), accounting for 95% of net global growth (360 TWh) [37, 38]. In China, the electric power sector has become the second-largest water user, after irrigation [39]. Water availability is therefore a key component of China’s electricity production [16, 24]. Of the total human water withdrawal, a lower percentage is used for electricity in China than in other countries, but China is one of the world's most water-stressed regions. That is, water availability per capita is classified as close to the international warning level of water stress (1700 m^3 per capita) [40], and the situation is likely to be further exacerbated by economic development and urbanization [41]. Additionally, studies have shown that there is a severe geographic mismatch between available water resources and thermal power plant locations across China [42]. Thus, there is an urgent need to understand the current and future conflicts between China's power production and water resources.

As the world's largest carbon dioxide (CO_2) emitter [43], China has set a carbon-neutral target of 2060 [44]. Many scenarios suggest that this will be met with substantial carbon capture and storage (CCS) [45, 46], potentially making the carbon available for use in the power sector (which emitted 4.2 $GtCO_2$ in 2019, comprising 41% of China's emissions) [45, 47, 48]. Most power systems modelling research has been aimed at optimising the energy mix for CO_2 reduction or optimising cost for various low-carbon goals [49], while the water impacts and challenges of water use for power production have not been incorporated into power system planning.

It is also of specific interest to examine the impacts on biodiversity related to the water-electricity nexus, as biodiversity conservation is attracting attention in China, where the new *Biodiversity Conservation in China* regulation [50] was implemented in 2021, with an emphasis on water biodiversity protection.

China is not only a large electricity producer but also has a large-scale power transmission infrastructure. It exchanges very little electricity internationally, but the scale of interprovincial power transmission within China is large and increasing (it increased 220% between 2008 and 2019 [51]). Interprovincial electricity transmission increased more rapidly than electricity generation, with 19.7% of the total electricity transmitted across provinces in 2019, mainly from the west to the east [48], although water scarcity is generally more severe in the west than in the east. Electricity-importing provinces are outsourcing water stress and water biodiversity impacts via transmission to other provinces, which may aggravate the water issues for some exporters. In assessments, it is increasingly important to quantify the virtual water embedded in transmission systems to provide insights for mitigating water stress across the country.

Although water and electricity are closely connected in China, the water-electricity nexus has not been fully incorporated into the country's water and electricity planning, due to a lack of information on this nexus. There is a need to depict a more detailed and complete map of the current water-electricity nexus and also its changes in the future.

1.5 Research questions

As discussed above, electric power production requires water resources and many power plants around the world are facing water stress. The situation is especially problematic in China due to the high water stress in most subnational regions and a geographic mismatch between water resources and the large-scale, water-intensive power production. The main research question posed in this thesis is: *What are the impacts and challenges of water use of electric power production in China?*

Investigating this question requires several studies at different stages, and prompts several further subquestions, which are listed below.

First, there are many types of electricity technologies, which have different requirements for water resources. These requirements can differ across countries due to different geographic conditions. It is necessary to understand the water requirements of various technologies across different regions.

SQ1. *What are the water requirements of different electricity technologies and what is the availability of regionally specific data?*

To answer this question, a literature review of the existing research is needed. This also requires an assessment of the data availability and results of data on water use of power production. Here there is a focus on what country-specific data are available, because different technologies require different amounts of water.

SQ2. *How much water is required for power production in China and how much water is virtually transferred via power transmission?*

To answer this question, an inventory of power production, transmission, and China-specific water intensity is used. On the basis of the answers, we can move on to the next stage, i.e., to assess the impacts of power production on the water system.

SQ3. *What are the impacts of power production on freshwater biodiversity in China?*

To answer this question, both freshwater consumption and freshwater thermal pollution of power production need to be quantified. Freshwater biodiversity loss will be estimated, to assess the extent of the impacts. The above three stages show the impacts of power production on water resources and the related biodiversity; on the other hand, power production faces challenges from water resources if its water requirements cannot be met. This issue is studied in the next stage.

SQ4. *What are the changes in water stress and the consequent impacts on power production in the future, and how might future carbon capture and storage (CCS) requirements exacerbate water issues in China?*

To answer this question, we combine a hydrological model and a thermoelectric power model. We assess the water stress faced by thermal power in the future and the impact that CCS as a solution for net-zero carbon emissions could have on water issues faced by the power sector.

1.6 Guide to this thesis

This thesis consists of 6 chapters (Figure 1.1). This first chapter gives a general introduction, describing the motivation, research questions, and outline of the thesis.

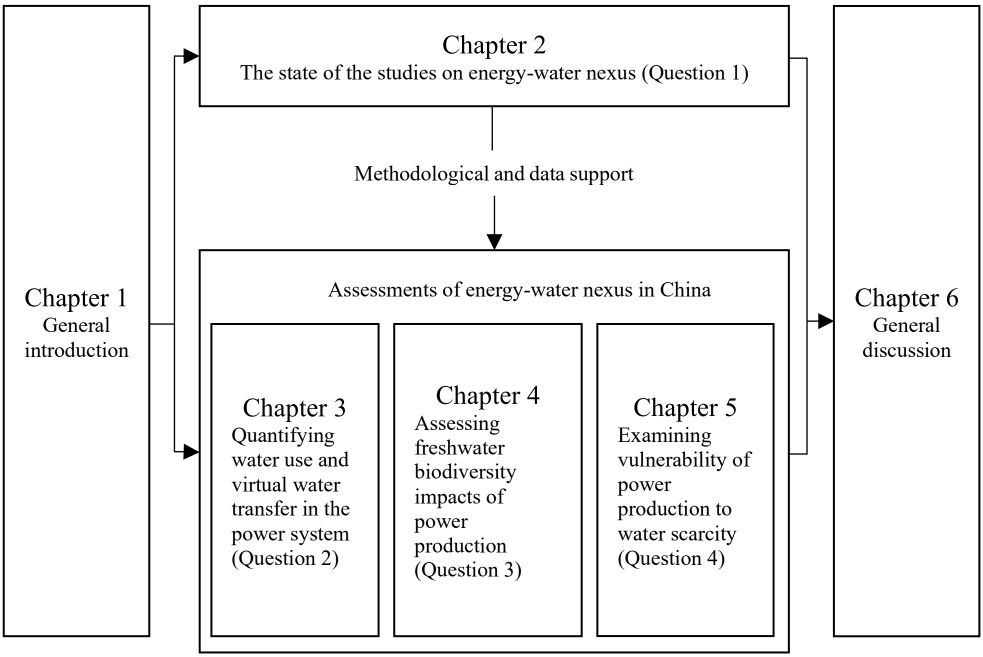


Figure 1.1 Outline of this thesis.

Chapter 2 addresses the first research question. It presents the available data on water use of power production at different life cycle stages, gathered by conducting a global meta-analysis. The differences in water use estimates within and across power types are reported, and the key drivers behind them are analyzed. We then analyze the uncertainties in assessments from the perspective of both methodological theories and the data inventory that we compiled. Finally, gaps in knowledge are identified to guide future studies.

Chapter 3 addresses the second research question. It quantifies the water use of power production and virtual water transfer via power transmission across China, using information on numerous renewable and non-renewable power-generating units and interprovincial power transmission. On this basis, it investigates the spatial and seasonal variations in water use and virtual water, and the impacts of power production and transmission on provincial water stress.

Chapter 4 addresses the third research question. It assesses the freshwater biodiversity loss caused by China's power production and the embodied biodiversity loss in power transmission. In this process, the characterization factors of water use impacts on freshwater biodiversity are developed for China. Further, based on the results, we analyze the decoupling relationship between biodiversity loss and electricity and the driving factors of the changing biodiversity loss.

Chapter 5 addresses the fourth research question. It examines the vulnerability of power production to water scarcity in China by developing a hydrology-electricity modelling framework, which quantifies the impacts of the changes in water availability and climate change mitigation actions of power plants (i.e., CCS) on power production. It also tests a set of adaptation options that have the potential to mitigate the vulnerabilities of power production.

Chapter 6 presents a synthesis of the answers to the research questions given in the preceding chapters, followed by a general discussion and outlook for future work.

Chapter 2

Water use of electricity technologies: A global meta-analysis

This chapter has been published as: Jin, Y., Behrens, P., Tukker, A., Scherer, L. Water use of electricity technologies: A global meta-analysis. *Renewable and Sustainable Energy Reviews*. 2019,115,109391.

Abstract: Understanding the water use of power production is an important step to both a sustainable energy transition and an improved understanding of water conservation measures. However, there are large differences across the literature that currently present barriers to decision making. Here, the compiled inventory of the blue water use of power production from existing studies allowed to uncover the characteristics of water use, and to investigate current uncertainties. The results show that photovoltaics, wind power, and run-of-the-river hydropower consume relatively little water, whereas reservoir hydropower and woody and herbaceous biomass can have an extremely large water footprint. The water consumption of power production can differ greatly across countries due to different geographic conditions. Only a few studies provided the values for the influencing factors of water use, such as the capacity factor. Values that are reported came mainly from assumptions and other literature rather than direct measurement. Omitting a life cycle stage may lead to significant underestimations. Water scarcity is attracting more attention, but the few existing results are not useable for a regional comparison due to data gaps and inconsistent measurements. In the future, a clear and detailed definition of the water footprint and system boundary of power production is essential to improving comparisons and energy systems modelling.

2.1 Introduction

Electric power production is a major driver of water stress worldwide [5, 6]. This situation is likely to be exacerbated due to growing energy demands and climatic change [18, 19, 52, 53]. In recent decades, technically plausible energy transition pathways have been designed to meet climate goals, but a concurrent analysis of the implications for water resources is mostly lacking. In some scenarios, emission mitigation benefits drive increased pressure on water resources [54, 55]. For instance, many climate stabilization scenarios rely on bioenergy with carbon capture and storage (BECCS) as a negative-emission technology, but it is a very water-intensive option [56, 57]. Rising water stress is of increasing concern to both renewable [24, 58] and non-renewable power production [22, 23]. Further energy system planning would greatly benefit from the incorporation of water stress perspectives and there are increasing efforts to include water resources as significant components in energy transition modeling [11-14]. The existing scientific literature provides a variety of water use estimates for various energy technologies and life cycle stages. However, many of these estimates differ widely or are even conflicting, giving an unclear picture of the energy-water nexus.

The use of water in the electricity system can be assessed using multiple metrics. The most common measure is the volumetric water footprint. It includes direct (i.e. water use for cooling at the point of generation) and indirect water use (i.e. upstream water use in the supply chain of fuels or equipment). It is defined by the volume of freshwater used by a consumer or producer over the entire supply chain [59, 60]. In recent years this concept has been extended to impact-oriented water footprints that assess not just the volume of water use but the potential environmental impacts [61]. The impact-oriented approach additionally considers regionalized impact indicators as part of traditional impact assessment frameworks [62]. Although both methods have been applied to studies on the water use of power production, most existing studies consider only the volume of water use of power production, which is therefore the main focus of our study.

Previous reviews on the water use of power production have focused on the United States (U.S.) [63-65]. Global assessments [66, 67] often rely on data from the U.S. and assume that generation in other countries has similar water use characteristics. A global overview of the differences in water use of power production is currently lacking. Water use covering the life cycle of power production have been used for estimating water use at the global [66-68] and country level [69-72]. For power production, the life cycle of water use can be split into fuel cycle, plant operation, and plant infrastructure stages. Analyses typically focus on the operational stage, distinguishing the water use by different cooling technologies and energy types. However, there are other important factors driving water use including fuel type, power plant type, and environmental conditions.

Although there must be uncertainties in the water use of power production, these are often not estimated in studies generally. This is often due to a lack of information on how to assess these uncertainties. This systematic literature review serves to investigate the above knowledge gaps by tearing apart the differences between previous studies, and presenting a picture of the current state of knowledge.

2.2 Methodology and data

Estimates from the literature were gathered following the PRISMA guidelines [73]. The meta-analysis focuses on the variations in water use estimates across technologies and locations, and the completeness of data reported across papers. In terms of the type of water uses, this study focuses on blue water (i.e. the use of surface or groundwater, such as irrigation water for biomass). In the framework of volumetric water footprints, blue, green (soil moisture), and grey water (hypothetical volume needed to dilute pollutants) are often added as if they were equivalent. In contrast in the life cycle assessment (LCA) community, green water use and water pollution are assessed through separate impact categories due to their fundamental differences [74], and are beyond the scope of this study. The gathered data represent two types of blue water use: withdrawal and consumption, with more emphasis on the latter. The former reflects the volume of water diverted from a water source for use, while the latter refers to the volume of withdrawn water not returned to the source due to evaporation, transpiration or incorporation into products [63, 75-77].

The database search was conducted in April 2019 using Web of Science and ScienceDirect without applying a time restriction. Search terms related to water footprints were used: *water footprint, water use, water consumption, water withdrawal, water demand, water requirement*, in combination with other terms representing both renewable and non-renewable power production: *renewable, non-renewable, fossil fuel, coal, oil, natural gas, shale gas, nuclear, hydropower, biomass, biofuel, geothermal, wind, solar, photovoltaic* and *electricity*. The full list of terms and their relevant variations, together with the numbers of results for each stage of screening, are shown in Supplementary information.

This search yielded 910 publications, which were filtered depending on whether the following inclusion criteria were met: (1) the value of the water use during the entire life cycle or a specific life cycle stage was reported; (2) the type of water use (consumption or withdrawal) could be distinguished; and (3) the information on the energy type was provided. Snowball sampling was also used. The final sample included 93 publications. (see Figure S7.1.1 for the full selection processes)

Data were extracted from publications either directly from tables, or from figures using WebPlotDigitizer, version 4.1. Common categories of analysis included: the type of energy (e.g. natural gas), energy sub-type (e.g. shale gas), type of water use (i.e. consumption or withdrawal), and the life cycle stage (e.g. fuel cycle). Extracted information on other factors included the country of assessment (e.g. Canada), cooling type (e.g. dry cooling), generator technology (e.g. combined cycle), conversion efficiency, capacity factor, lifetime, and environmental conditions (e.g. solar irradiation). The full dataset and influencing variables are shown in Supplementary information, respectively.

Due to data limitation and inconsistency for impact-oriented water footprints (namely water scarcity footprints), these are discussed separately (Section 2.4.2). Generally, studies estimated blue water use based on the values of the influencing factors, such as the conversion efficiency. However, the effects of such factors on water use lack quantitative assessment. In this study, correlation analysis and linear regression are used to investigate the relationships between key factors and water use of power production. As for linear regression, this study investigates the relations between operational water consumption and its influencing variables (cooling type and conversion efficiency) for five power types (coal, natural gas, oil, nuclear and biomass).

2.3 Results

2.3.1 Overall results

Blue water consumption and withdrawal for the total life cycle were reported in 32 studies (34% of sample, see Figure 2.1 for consumption and Figure S7.1.2 for withdrawal). As expected, there is a large range in water uses across energy types. For instance, the median life cycle water consumption for biomass is 8.5×10^4 L/MWh, one to three orders of magnitude larger than other types. Generally, biomass can be classified into four groups, including wood and woody biomass, herbaceous biomass, aquatic biomass, and animal and human waste biomass [78]. Previous studies on the water use of biomass power focused on the first two above-mentioned groups, as they are the main feedstock of biomass power. Hence, the latter two biomass types are not included in this study. The extreme estimate represented the large requirement for irrigation of herbaceous perennials in the arid Southwestern U.S. [64]. Although the water consumption for wind power is widely thought to be negligible [23, 68, 79-81] and it is characterized by the lowest median water consumption, it can still reach 700 L/MWh if direct and indirect material inputs for wind power are included using hybrid LCA (see detailed discussion in section 2.4.1) [82]. Similarly for photovoltaics (PV), the outliers of life cycle water consumption were caused by using a hybrid method [82, 83]. For geothermal energy, the only outlier resulted from the large belowground water consumption for an enhanced geothermal system (EGS), in which case 10% belowground water loss during operation was assumed [84]. However, as the belowground water consumption is for maintaining the reservoir, the water does not need to be of high quality. If the water used for belowground operation was not freshwater, then its life cycle water consumption would decrease dramatically from 7037 L/MWh to just 185 L/MWh.

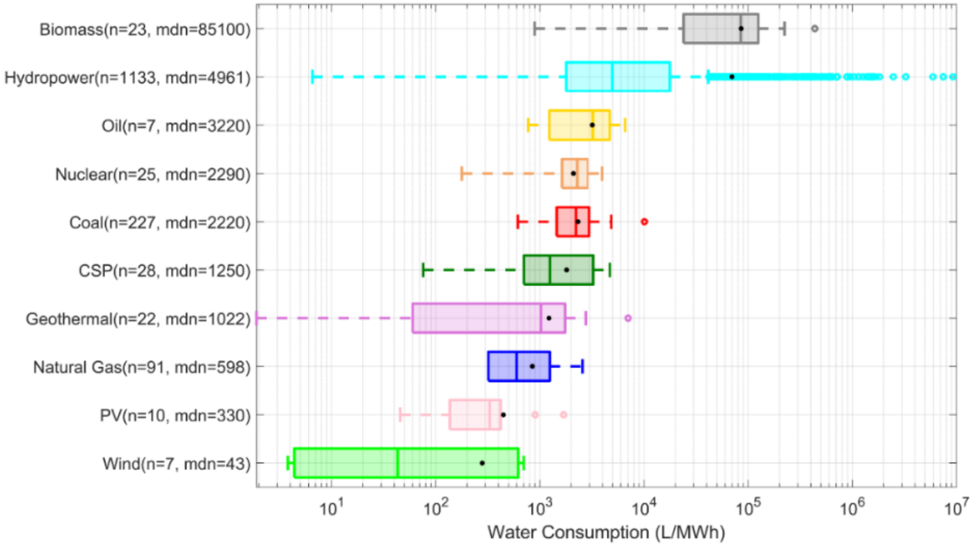


Figure 2.1 Blue water consumption over the life cycle across energy generation types. Water consumption is visualized on a log scale. The annotation mdn gives the median value of water consumption for each fuel type. Circles represent the outliers, while the dots represent the mean for each power type.

Another point to note is the generally high variability in water consumption across power plants of the same type. Coal power plants has relatively low variabilities in life cycle water consumption, whereas hydropower has a marked variability, with a coefficient of variation of 634% (Table S7.1.2). Local estimates are especially important for biomass, oil power and hydropower. For water withdrawal, the ranking of energy technologies based on the median or average values remains the same, except for natural gas which ranks higher than geothermal energy in terms of water withdrawal (Figure S7.1.2). The range of water withdrawal is generally much wider than of consumption due to large withdrawal differences between once-through cooling and other cooling types. Water consumed during once-through cooling is generally negligible (around 1%).

2.3.1.1 Water use of fuel supply

The water uses of fuel supply reported in this section only apply to fuels for electricity generation, that is: coal, natural gas, oil, nuclear, and biomass, as shown in Figure S7.1.3. The water uses here refer to the blue water used for fuel supply, i.e. extraction (for biomass, it refers to crop cultivation), processing and transport. For biomass power, the key stage of life cycle water consumption is the fuel cycle due to the considerable water input in crop cultivation [85, 86]. Herbaceous and woody biomasses are separately examined in terms of the fuel cycle, as they have different water demands for growth. The median water consumption for herbaceous biomass (7.6×10^4 L/MWh) is much larger than that of other fuel types by more than two orders of magnitude, but still much smaller than that of woody biomass (8.3×10^5 L/MWh). Within biomass, water consumption varies greatly, from 7200 L/MWh [87] to 2.8×10^7 L/MWh [86]. An exception excluded in this figure is [88] where hybrid poplar was assumed to be rain-fed (i.e., no irrigation water is used). In terms of fuel cycle, natural gas has the lowest median water consumption (128 L/MWh), lower than that of nuclear (156 L/MWh) and coal (231 L/MWh). Within natural gas, there are three fuel sub-types: conventional gas, coal-bed methane and shale gas, with median consumptions of 60 L/MWh, 70 L/MWh, and 222 L/MWh respectively. Water use in fracturing rock for shale gas explains the large volume (65% of the median variation), with the remaining from indirect water use in the supply chain (33% of the median variation) [65, 89]. Oil is a large water consumer at the fuel cycle, with median water consumption of 891 L/MWh for conventional and 1658 L/MWh for unconventional oil (oil sand and oil shale) [66, 76]. Studies generally assume that water withdrawal in the fuel cycle is equal to water consumption in the fuel cycle [65, 76, 90-93].

2.3.1.2 Operational water use

The water uses here refer to the blue water used in the operational process of power plants. Studies typically focus on cooling systems, as it accounts for most of the operational water use. Hereinafter, cooling water consumption refers to the blue water evaporated during operation for cooling purposes. Water consumption is reported first. Hydropower is the largest water consumer during the operational phase (median = 5.1×10^4 L/MWh), one to three orders of magnitude larger than that of other types (Figure S7.1.4). Large differences exist within hydropower, ranging from 0 [94, 95] to 1.2×10^8 L/MWh [24]. Most studies estimated water consumption based on the gross water evaporation from reservoirs, which changes as a function of the reservoir surface area. According to [94] and [95], the gross water consumption was regarded as zero for those hydropower stations running without reservoirs (i.e. run-of-river plants). Similarly, for plants running with reservoirs, evapotranspiration was assumed to occur from the same area prior to the establishment of the reservoir [29]. Taking this into account, some studies calculated the net water consumption by subtracting the evapotranspiration before the dam construction from the gross water evapotranspiration [24, 76, 96-98]. These studies show that net water consumption is on average 54% of the gross water consumption. However, because reservoirs offer multiple purposes, such as water supply, flood control, and navigation, some studies suggest that for a fair comparison with other energy types the impacts of the reservoir should be allocated among its multiple purposes [24].

For coal, extremely large operational water consumption was generally caused by closed-loop plants with low conversion efficiency or with carbon capture equipment [64, 65, 99, 100]. Once-through cooled units may also have high water consumption rates, driven by low electricity output and large incoming flows of cooling water in unique locations [99, 101].

In terms of water withdrawal, nuclear is a large water withdrawer, with a median value of 2.67×10^4 L/MWh (see Figure S7.1.5). Compared to other thermal power plants, nuclear plants generate steam at lower temperatures and pressure for operational safety, and consequently, are less thermally efficient and withdraw more cooling water per unit of electricity [102]. The median value for oil is much larger than for coal and gas because studies on oil mainly focused on wet cooling technology, especially once-through cooling [56, 103]. PV plants may withdraw a considerable amount of water for mirror washing, but in practice, PV panels are seldom washed by operators [65].

Figure 2.2 and Figure S7.1.6 present detailed values for water consumption and water withdrawal for different cooling technologies, indicating that water uses of operation show greater agreement when grouped according to cooling types as opposed to power types. For coal, natural gas, oil, nuclear and biomass, power plants with closed-loop cooling technology are the largest water consumers, while plants with once-through cooling technology are leading water withdrawers. For concentrating solar power (CSP) and geothermal, water withdrawal was widely assumed to be equal to water consumption at the operational stage [56, 64, 65, 75, 104-106].

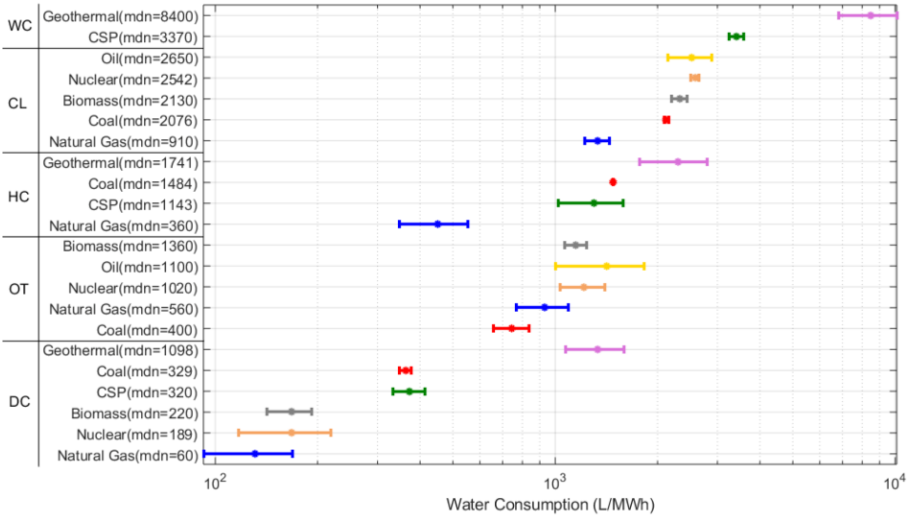


Figure 2.2 Blue water consumption of operation distinguished by power type and cooling type. The dots represent mean water consumption, while the line segments represent the standard error of mean. The annotation mdn gives the median value. Hydropower, wind, and PV do not have cooling needs and are not included. The two-letter codes are as follows: WC wet cooling, CL closed-loop cooling, HC hybrid cooling (combining wet and dry cooling), OT once-through cooling, and DC dry cooling. Colors map to fuel type for the estimate.

2.3.1.3 Water use of plant infrastructure

The water uses of plant infrastructure refer to the blue water used to manufacture each material input of power plants, with the indirect blue water embodied in material inputs also included. The water use of plant infrastructure was often neglected due to its small proportion in the total life cycle for most power types. However, this does not apply to all types. As shown in Figure S7.1.7 and Figure S7.1.8, large amounts of water consumption and withdrawal are required for the plant infrastructure of CSP. PV can consume more water than CSP, reaching up to 794 L/MWh if the PV material is crystalline silicon [65]. Wu et al. [107] indicated that coal thermal power plant requires significantly more water for infrastructure than natural gas combined cycle plant. According to our study, coal thermal plant's infrastructure is the largest water user among all fuel-powered thermal plants. Nuclear power has the lowest water consumption per unit of power production due to the high electricity output over generally longer lifetimes.

2.3.1.4 Water use of carbon capture and storage

Carbon capture and storage (CCS) heavily influences the water use of thermal power plants [108-112]. The water uses of CCS refer to the additional blue water used due to the addition of the CCS system. All estimates related to CCS adopted by natural gas and biomass power were available for combined cycle cooling only. Due to the additional water requirement for CCS equipment and the loss in operation efficiency [65], plants using CCS generally consume more water than the counterparts without CCS (Figure 2.3). Biomass also faces the challenge of large water uses for the feedstock, while for BECCS, water use is increased further by the CCS additions. Direct air capture (DAC) is an emerging technique that capture the carbon dioxide from the ambient air, and may potentially provide negative emissions. Yet, it may have large water requirements due to the evaporative loss of the DAC unit based on amine technology [57, 113]. DAC water requirements may change as the technology scales but further research is necessary.

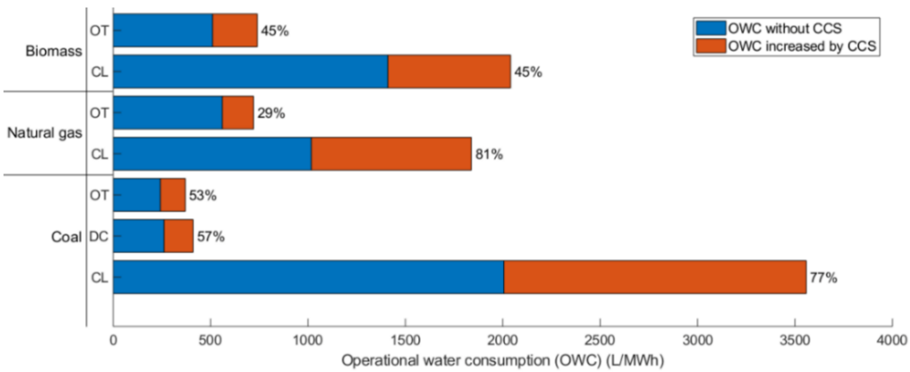


Figure 2.3 Additional blue water consumption due to the addition of CCS for different fuels and cooling types. The numbers on the right of each bar indicate the percentages for CCS compared to operational water consumption (OWC) without CCS. Only the literature which reports both the OWC with and without CCS are included.

2.3.2 Country-specific water use of power production

The number of studies per region and per power type are shown in Figure 2.4. Hydropower and coal are widely studied across many regions, whereas geothermal lacks research in all regions except the U.S. The water consumption and withdrawal of each life cycle stage for the different countries studied previously are shown in Tables S2.4-S2.8. China and the U.S. are the predominant focus, and their specific water uses of power production are presented in Tables S2.9-S2.14 with the consideration of both cooling type and generating technology (e.g. combined cycle or steam turbine).

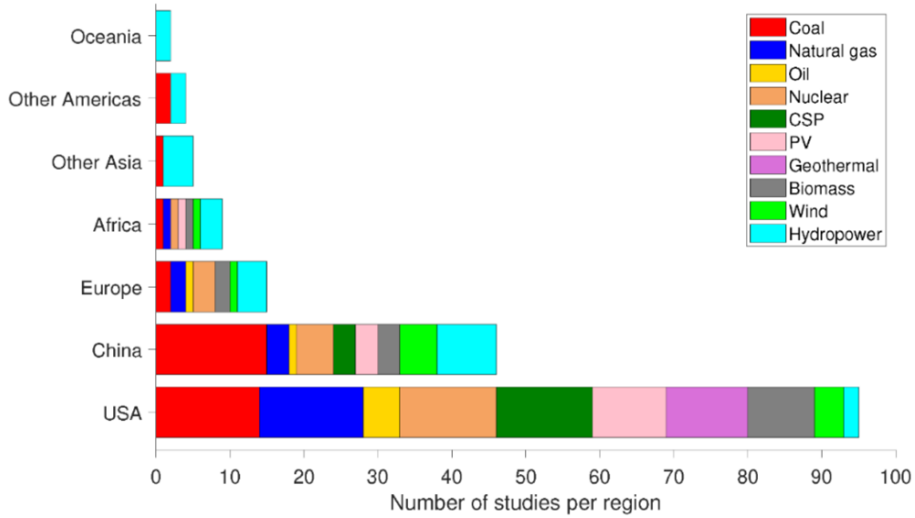


Figure 2.4 Number of studies per energy source per region. Many studies investigated more than one energy source and region, and can therefore occur multiple times.

Studies on shale gas in China focused on the shale-rich Sichuan basin [114, 115], whereas U.S. shale plays are distributed more widely [116, 117]. The more complicated shale formations and water-intensive fracturing techniques in China led to higher water consumption for shale gas extraction and power production as compared to the U.S. [118]. For nuclear power, the water use of the fuel cycle depends on the type of mining activities and enrichment [65, 93]. There are three types of mining activities: in-situ leaching (ISL), surface mines, and underground mines. There are also two types of enrichment: diffusion and centrifugal enrichment. In France, Uranium used was mainly from underground mines, and processed through the diffusion enrichment, both activities generally consume less water than counterparts in the U.S.. Poinssot et al. (2014) indicated that the use of ISL techniques in France could consume a larger amount of water than underground mines [93]. However, the water consumption of underground mines varies greatly. For countries where underground mines consume more than 87 L/MWh, the maximum of ISL techniques, the application of ISL instead becomes a way to save water [65].

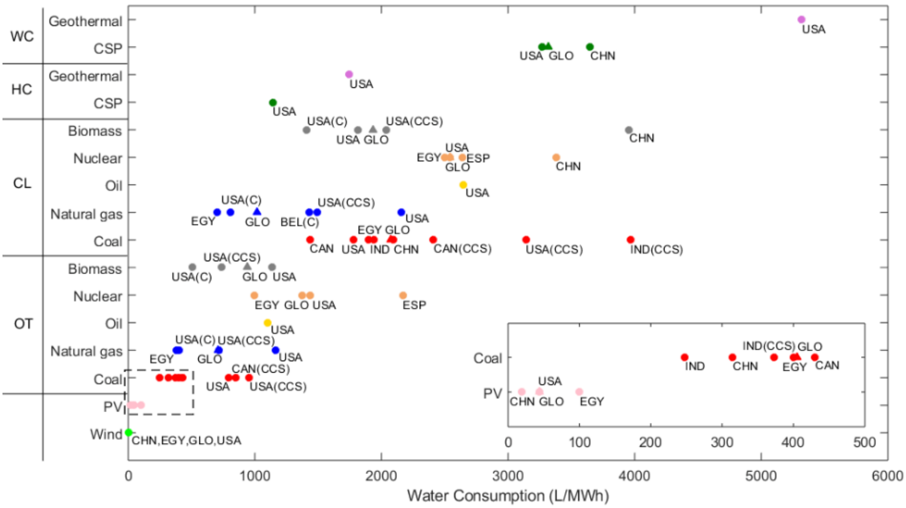


Figure 2.5 Operational blue water consumption for each power type and country. Countries are indicated by ISO3 codes [126]. GLO denotes the globally median value and is represented as a triangle. For clarity, the contents of PV and coal power in the dashed box are enlarged and shown inset in the bottom right. Codes denote WC wet cooling, HC hybrid cooling (combining wet and dry cooling), CL closed-loop cooling, and OT once-through cooling. "C" in parentheses denotes combined cycle power plants. "CCS" in parentheses denotes power plants using carbon capture and storage technology. The median operational water consumption of wind power is zero for all countries herein. The operational water consumption of hydropower is not included in this figure due to its wide range, but is available in Table S7.1.5.

For both CSP and nuclear, China consumes more water than the U.S. (Figure 2.5). Direct normal irradiation (DNI) determines CSP operating efficiency [119]. As such 99% of U.S. CSP capacity is in three states: California, Arizona, and Nevada [120]. According to [172], DNI of these U.S. regions ranges from 2400 to 2940 kWh/m^2/year. In China, CSP plants are mainly in northwest and Wu et al. [121] indicated high operational water consumption of a CSP plant in Gansu province. According to the World Solar Atlas [122], this area receives less DNI (a maximum: 2193 kWh/m^2/year) than in the U.S., contributing to a lower operating efficiency and a higher water use.

Geographic conditions also influence the water uses of nuclear power [123]. In the U.S., operational water consumption for nuclear power plants with closed-loop cooling could be more than 3000 L/MWh [63-65, 99, 105, 124], with a minimum of 1408 L/MWh. Even within China, differences in the nuclear power water requirement could reach 24% due to climate differences between northern and southern regions [125]. Coal power plants with closed-loop cooling consume more water in India and China than in Canada and the U.S. Conversely once-through cooling consumes less water in India and China than in Canada and the U.S.

2.3.3 Key factors influencing the water use of power production

Studies generally presented estimates of water uses without simultaneously presenting their influencing factors. Compiling the key factors from the literature and analyzing their effects on water uses allows for a better understanding of the drivers behind different water uses.

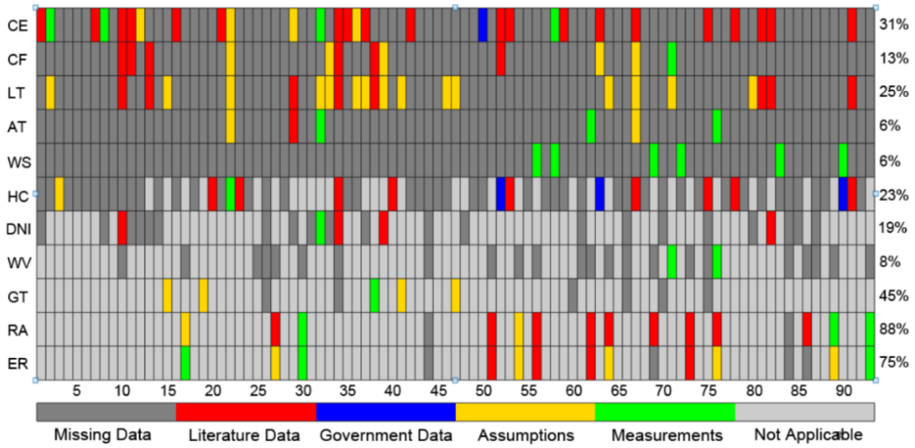


Figure 2.6 Factors reported across the literature. The x axis presents each study, numbered from 1 to 93 chronologically from left to right (these studies are listed in Supplementary information). The y axis (left) presents influencing factors: CE (conversion efficiency), CF (capacity factor), LT (lifetime), AT (ambient temperature), WS (water scarcity), HC (heat content of fuel), DNI (direct normal irradiation), WV (Wind velocity), GT (geofluid temperature), RA (reservoir area), ER (evaporation rate). The y axis (right) presents the percentage of reporting in the literature overall (the ratio of reporting to the total applicable studies). Colors denote the data sources of factors in the literature. "Measurements" means that values were directly measured, or came from primary data. "Not Applicable" means that the factor is not relevant to the study i.e. reservoir area only applies to studies including

hydropower.

Beyond cooling and power type there are other important factors driving footprints. These can include the resource quality (e.g. heat content of fuel), power plant specifications (e.g. conversion efficiency, capacity factor, lifetime) [127-130] and environmental conditions (e.g. ambient temperature [131], direct normal irradiation [104], wind velocity [132], geofluid temperature [106], reservoir area [133], evaporation rate [96]). Influencing factors were collected from the literature for each life cycle stage of power production. As shown in Figure 2.6, there are many data gaps across studies. Further, most values came from assumptions and other literature rather than direct measurements. There is no evidence of increased reporting of these factors over time. Water scarcity is an exception, having received more attention recently, even though the numbers of studies are still limited. Ambient temperature is typically a determinant of the operational water use due to its influence on production efficiency, cooling efficiency, evaporation rate, etc. However, it is not generally reported. The reported conversion efficiency, capacity factor, plant lifetime, and heat content of fuel are shown in Tables S2.15-S2.18.

Of the common factors, conversion efficiency is most frequently cited in the literature (31% of studies). It has been identified as a key driver of operational water consumption for most power types, especially those with cooling water demands [134]. Here a regression model is used to calculate the impact of each factor on water consumption with cooling type and conversion efficiency included. Since the operational water consumption of five power types (coal, natural gas, oil, nuclear, and biomass power) closely agree when grouped by cooling type (Figure 2.2), these five power types are considered in the model without distinctions. The model established is shown in Supplementary information, and the results are presented in Table S7.1.20. The impact of conversion efficiency on operational water consumption varies across cooling types. On average, -36.8, -16.2, and -10.3 L/MWh of operational water can be saved with every 1% increase in conversion efficiency for closed-loop cooling, once-through cooling, and dry cooling, respectively. Compared to improving conversion efficiency, adopting dry cooling technology is a direct approach for conserving water but generally increases investment costs [135, 136] and lowers plant efficiency [137, 138]. Wet cooling can bring synergistic benefits, e.g. energy conservation and emission reduction [139].

There are additional factors for each power type. The heat content of fuel is important for coal, natural gas, and biomass. Generally, biomass has a lower heat content than natural gas and coal. The operational water consumption of hydropower varies greatly depending on two factors: the evaporation rate and the surface area of reservoir [24, 133, 140, 141]. The evaporation rate in different locations could range from 486 mm yr^{-1} to 3059 mm yr^{-1} [142]. The reservoir area also varies over time due to changes in water volume throughout the year [133, 142, 143]. However, these are usually only estimated annually due to data limitations in monitoring area over the year. This may change as better remote sensing methods become available; already some studies have estimates at a monthly resolution [144]. An analysis of variance is conducted to look at the contributions of both evaporation rate and surface area on the operational water consumption of hydropower. Results show that the evaporation rate typically plays a more important role in determining the operational water consumption (Table S7.1.21). For geothermal energy, the plant type (flash cycle, binary cycle) typically determines the water requirement. For flash power plants, a lot of freshwater can be saved, as most cooling water is provided by the geothermal fluid that flashes to steam and during the generation process condenses to form high quality water that can be used for cooling [106].

Finally, the water source (i.e. freshwater water and sea water) for plant cooling differs across regions. For example, many nuclear power plants in Spain and the U.S. use water from rivers and lakes for cooling, whereas China has all presently operable nuclear power plants in coastal areas with seawater as cooling medium to save freshwater [145, 146]. The deployment of power plants and cooling water sources make big differences to the blue water use of power production.

2.4 Discussion

2.4.1 Uncertainties in water use assessment

Uncertainties derive from the methodological choice, the system boundary cut-off, and the water source.

Methodological choices: The two main methods are process-based LCA (PLCA) and hybrid LCA (a method linking PLCA and input-output analysis (IOA)). PLCA is a bottom-up approach based on production processes [83, 147], whereas IOA is a top-down approach [148, 149]. Hybrid LCA was developed based on PLCA and IOA to combine their strengths and reduce weaknesses from data quality, system boundary, difficulty of application, etc. [150-152]. In recent decades, hybrid LCAs have increasingly been employed in energy-related environmental footprint analyses [148, 153-155] and water footprint analyses (as mentioned above). The application of both methods to carbon footprinting for wind power indicated that emissions for hybrid LCA was more than double that for PLCA [156]. Equivalent differences by method in water use estimates are shown in Table S7.1.22. Hybrid LCA leads to larger water use estimates for most power types, especially wind and PV. The additional input from economic sectors not covered by process analysis was the major contributor to the differences. Though it remains an open question of which method should be recommended for life cycle inventory compilation, since both are in line with the ISO standard [157], the differences between PLCA and hybrid need to be appreciated. Firstly, using a pure PLCA approach may lead to significant underestimation because power production relies indirectly on large amounts of inputs from various sectors, especially heavy industries (steel, metal and cement) [82, 83, 91, 158] and agriculture (e.g. wood used for fuel extraction and construction; agricultural products used for chemical production) [22, 89], which are generally water-intensive [89, 159, 160]. Second, using IO-based hybrid LCA presents a challenge in sector disaggregation. Power production is typically a homogenous sector in IO tables [161, 162], even though each power type has a distinctive water use. Efforts are needed to isolate the targeted power type from the power production sector [161]. Third, IO tables are normally released later than process-based data [157]. This may be an issue for emerging power production technologies.

For some energy technologies, there are specific methodologies that influence water use estimates. For hydropower, the main issue is the lack of methodological consistency in allocating water consumption among multiple purposes [67, 96, 163]. Many allocation methods were used to separate the water consumption of electricity from the total reservoir evaporation by using an allocation factor. The allocated water consumption of electricity may be much lower than the reservoir evaporation or remain unchanged [24, 141, 164], depending on the allocation factors as shown in Table S7.1.23. The temporal resolution of models can also lead to different estimates due to the seasonal fluctuations in the reservoir water level.

Choices for boundary cut-off: Although some studies cover the entire life cycle of power production, operational water uses are a focus across the literature. The water uses of the fuel cycle and plant infrastructure are often omitted. Omitting the water uses of a certain stage can lead to a bias that varies across power types (Figure 2.7). Over the total life cycle of water consumption, the share of water consumption from the plant infrastructure varies greatly, especially for renewable energy with the exception of bio-power. Likewise, the fuel cycle of coal, natural gas, oil, and nuclear, ranges from 2 to 79%, largely depending on the cooling water consumption of power plants. Within the fuel cycle, the water consumption of each sub-stage is presented for coal and natural gas (Figure S7.1.10). For coal, the transport by pipeline, especially the slurry pipeline, is a highly water-consuming choice. For natural gas, processing and pipeline transport are large water consumers. The sum of their median water consumption is approximately half of the median water consumption of fracturing, thus narrows the gap between conventional gas and shale gas for total fuel-cycle water consumption.

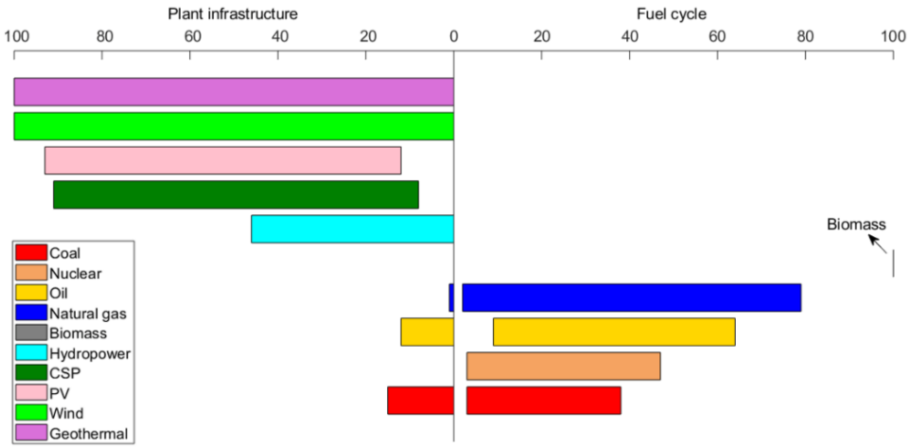


Figure 2.7 Proportions of the blue water consumption of fuel cycle and plant infrastructure in the total life cycle. Values are shown in Table S7.1.24.

Water source: For biomass and geothermal, water sources considered in the assessment make a difference. Irrigation (blue water) and soil moisture from precipitation (green water) are main sources of water required for biomass growth. The former accounted for 0-60% of the total water consumption [58, 87, 165]. It is essential to identify the feedstock type, as water demand varies across types [166]. Geothermal fluid and freshwater are two sources of operational water use of geothermal power production but geothermal fluid is not typically considered as freshwater consumption because it is not sourced from a body of freshwater [63, 106]. Either or both types of water resources are used in operation, depending on the practical situation [84, 106], which leads to variations in estimates. Nuclear power plants in coastal areas typically use seawater for cooling [100], which is irrelevant to blue water use.

2.4.2 Water scarcity related to power production

The large amount of water abstracted for power production might exacerbate local water scarcity [132, 167-169]. This depends on two factors: the life cycle water use of power production, and the water scarcity in the region [170]. A certain amount of water use in water-poor regions typically has larger impacts on other local water users than in water-rich regions. To alleviate the risk of water use in water-scarce regions, the regional water scarcity needs to be considered in addition to volumetric water use [74]. However, there is no consistent measurement to reflect the impact of power production on water scarcity. Two main approaches have been used, which are both related to water scarcity indices. One measures the energy-related water scarcity index by dividing the water use of power production either by total water availability [95, 168, 171], or by the remaining water availability after subtracting non-power production uses in a water basin [53]. Another approach uses the water scarcity footprint, which is calculated by multiplying the water use of power production with a regional water scarcity index [22, 24, 169, 172, 173]. Thus, the water scarcity footprint includes the information of both the volumetric water footprint (i.e. the water use inventory) and regional water scarcity.

Apart from the different approaches to assess water scarcity, the differences in the scaling of the water scarcity index used across studies also imply that their water scarcity measurements are not comparable [173]. Hence, it is suggested to still report water use besides the water scarcity footprint. According to [174], the value of the water scarcity indicator exceeds 100% when the blue water use is higher than 20% of the natural runoff within a region (i.e. the water availability), whereas in [175] and [176, 177], the scaling of 0.01 to 1 and 0.1 to 100 were used for the values of the water scarcity indicator, respectively. In addition, it is worth noting that the terminology “water scarcity footprint” is used by the LCA community. A similar concept was proposed by the Water Footprint Network and named water footprint impact index [59]. The impacts of power production on local water scarcity have raised increasing concerns. However, the existing studies on water scarcity footprinting seldom provided cross-regional strategies for mitigating water scarcity. More efforts are needed to figure out synergies between water and energy management (i.e. water allocation and energy deployment).

2.4.3 Water in energy system modelling

Energy systems typically consider three aspects: reliability, affordability, and sustainability [21]. Integrating water use into energy system modelling is important from both reliability and sustainability perspectives. The two key factors influencing reliability are capacity factor and installed capacity, both of which dictate the reliability of plants when water availability changes. However, relevant information is seldom provided in studies on water use, making it difficult to link water and energy system modelling.

For example, natural gas plants are used in many different modes on different electricity grids. While some studies assume a capacity factor of natural gas of 80% [92] or 85% [65, 91], natural gas plants can often act as 'peakers', that is they only operate during high demand. According to EIA [178], the capacity factor of natural gas thermal power plants in the U.S. in 2017 was 6.7% and 10.4% for combustion turbines and steam turbines, respectively; and even for the combined cycle plants it was only 51.2%. Clearly, this also has a temporal aspect since plants may be peaking under high cooling loads, which may be at the same time as low water availability or thermal constraints.

Additionally, installed capacity is often underreported in studies (except those on hydropower), as is cooling type. Both variables are needed for a realistic and complete energy system model. For instance, if the water use of energy systems is optimized without data on the capacity of each cooling technology, results may suggest that plants convert to dry cooling (a water-saving technology), despite the fact that this may not be suitable in regions where air temperature is high and results in low production efficiencies [179]. Providing the capacity of each operating technology is essential to give a baseline when looking for trade-offs between water saving and energy production.

The studies on the water sustainability of energy systems rely on the information of water availability. Hydrological models are often used to show water availability at regional or basin level. Although there are many global hydrological models [180], they are highly uncertain and need to move to a finer spatial resolution to address more targeted water scarcity at plant levels. In addition, the upper limit of water use in regulations is another index that has been used as a reference to water availability [171]. In practice, the water availability of energy systems is sometimes more restricted by water use regulations [111, 171] other than the water scarcity limitation shown in hydrological models, to ensure long-term water security [181-183]. To figure out whether water will be a barrier for energy systems, a better understanding of both the natural water scarcity and water use regulations is needed.

Besides water, there is also a need to consider other critical resources that are required as an input into energy systems and influence its sustainability. To do so, the exergy concept has recently been applied to evaluate the environmental impacts [184, 185] or economic performance [186, 187] of energy systems. Exergy accounting enables studies to reveal the resource depletion and measure all impacts in homogeneous units [188].

2.5 Conclusions and future prospects

This study gathered available data of water uses of power production at different life cycle stages. Differences and uncertainties in water use estimates were analyzed for each power type. The following conclusions are reached:

Renewable energy may be water-saving or water intensive: PV, wind power, and run-of-river hydropower consume relatively little water; CSP and geothermal consume intermediate volumes of water; whereas woody and herbaceous biomass and reservoir hydropower may possess an extremely high volumetric water footprint. Non-renewable energy falls within the two higher water use classes, except for natural gas between the two lower water use classes of renewable energy. The deployment location of power production largely affects countries' water use of energy systems due to different climate conditions and water resources, as well as the impacts caused by it due to water scarcity; however, the latter are rarely considered. For thermal power plants, the operational water consumption increases up to 81% (natural gas) due to the addition of CCS.

Inconsistent system boundaries may cause uncertainties in water use estimates across studies. For example, the fuel cycle of biomass, nuclear power and natural gas is worth more consideration in the future. Besides clarifying the water use type (consumption vs. withdrawal), clarifying the water sources also helps reduce uncertainties in water use estimates for biomass (precipitation vs. irrigation), geothermal (geofluid vs. freshwater) and nuclear power (seawater vs. freshwater). Emphasis for future studies should be to increase transparency and report key influencing factors, such as conversion efficiency, capacity factor, lifetime, ambient temperature, and depending on applicability also the heat content of the fuel, direct normal irradiation, wind velocity, geofluid temperature, evaporation rate, and reservoir area. Finally, the inclusion of water scarcity in energy system optimisation models is essential for mapping the energy transition.

Chapter 3

The energy-water nexus of China’s interprovincial and seasonal electric power transmission

This chapter has been published as: Jin, Y., Behrens, P., Tukker, A., Scherer, L. The energy-water nexus of China’s interprovincial and seasonal electric power transmission. *Applied Energy*. 2021,286,116493.

Abstract: Modern energy systems use large amounts of water for electric power production. This has important impacts on future water management and energy system planning decisions. In this study, we quantify the physical water use of power production and virtual water transfer via power transmission between Chinese provinces using the information on 5408 electricity-generating units and interprovincial power transmission. We show that China's power production withdrew 62.7 billion m^3 of freshwater in 2017, of which 13 billion m^3 was consumed (i.e. not returned to the original water basin but lost via evaporation, etc.). A large volume of freshwater was virtually traded through the transmission system. Overall, 6.2 billion m^3 of freshwater withdrawals and 2.1 billion m^3 of water consumption was traded. Nationally, power transmission reduced freshwater withdrawal but increased consumption in China because, compared to the east, the west generally has a larger water consumption factor but a lower withdrawal factor. Water stress was more equally distributed across provinces through power transmission. We find large seasonal variations in inter-regional virtual water consumption transfer, with an August peak. While the Yangtze River basin and downstream of the Yellow River basin have abundant water relative to other basins, the many power plants located along the two rivers aggravate local water stress. These dynamics will become increasingly important for policymakers and energy planners as China undergoes climatic changes and a rapid energy transition.

3.1 Introduction

Global electricity demand grew 4% in 2018 and is already a major driver for water stress worldwide [25]. Meanwhile, climate change and water shortages have increased the sensitivity of power systems to water availability [189], raising both research and policy concerns [53]. In 2017, 26% of global electric power was produced by China [37], with thermal and hydropower the main contributors. Both technologies depend on water resources and satisfying this requirement is a key component of energy security [16, 24]. Specifically, China accounts for 29% and 28% of the world's thermal power and hydropower production, respectively [38]. An assessment of the water use of the two energy technologies in China is important for understanding the global energy-water nexus. Across China's border regions, especially in the South and Southwest, power production may have impacts on transboundary water resources, for instance across the Lancang Mekong river basin [190], depending on what choices are made for electricity generation and cooling as these regions develop further. The operations of hydropower dams in the mainstem on the Lancang Mekong Basin have effects on the downstream water flows, generally reducing the flow during the wet season and increasing it during the dry season. Another example of transboundary water resources originating in China is the Brahmaputra River, which directly flows through three countries: China, India, and Bangladesh, and is an important water source for the domestic and agricultural practices in these countries [191, 192]. Since the river flows through some highly disputed areas, the potential for riparian conflicts of interest over water resources development is significant [193]. In China's recently released (November 2020) China's national development plan, more water resources of the Brahmaputra River will be used for hydropower generation [194]. The type of hydropower plant (i.e. run-of-river or running with reservoirs [16]) will determine the impacts of power generation on the water availability of downstream users inside and outside of China. As the world's largest carbon dioxide (CO_2) emitter [43], China promises to make efforts to be carbon neutral before 2060 [44]. This is likely to entail the large-scale use of carbon capture and storage (CCS) which requires additional water resources [195]. This may be used both in the power sector (which comprises 50% of China's emissions) and in the development of negative emission technologies [46, 196]. For the regions that lack sufficient water resources to meet the additional water demands of CCS, the adoption of CCS can exacerbate the vulnerability of power plants to water scarcity. A spatially explicit mapping of power plants' water use is essential to the trade-offs between CO_2 emissions reduction and water scarcity mitigation for China and the world.

Water use is typically split into two forms: withdrawal and consumption [197]. The former reflects the volume of water diverted from a water source for use, all or part of which may be returned (but generally at a lower quality), while the latter refers to the volume of water not returned to the water body due to evaporation, transpiration or incorporation into products, i.e. water consumed always reduces the remaining water quantity [198, 199]. The cooling requirement of thermal power production needs a significant amount of water [16, 171, 179]. There are three common cooling types: 1) once-through cooling, requiring large amounts of water withdrawal and directly returning most of that water to its source; 2) closed-loop (wet tower) cooling in which some of the water is consumed through evaporation (it withdraws less but consumes more than once-through cooling); and 3) air cooling using air-cooled condensers for steam cooling, which avoids evaporative water losses [200]. Power plants with closed-loop cooling technology are large water consumers, while plants with once-through cooling technology are leading water withdrawers [90, 99, 103, 109, 111]. Previous studies showed that 57.6 billion m^3 of freshwater was withdrawn for thermal power production in China in 2015 [23], approximately 9.4% of the national total water withdrawal in that year [201]. Estimates for the annual freshwater consumption of thermal power production in China range from 3.8 to 5.7 billion m^3 [23, 168, 202], largely depending on the assumed water consumption factors. Hydropower was not considered in these studies but consumes large amounts of water through reservoir evaporation [16]. Hydropower can be water-intensive depending on the reservoir area and local evaporation rate [24, 140]. Previous research indicates a wide range for hydropower water use, from negative values (due to reservoir water storage increasing availability downstream) to more than 115000 m^3 MWh^{-1} [203]. In China specifically, Liu et al. showed hydropower water use ranges of 13 to 15244 m^3 MWh^{-1} [204]. Zhu et al. [205] and Liao et al. [202] estimated that 11.5 to 14.6 billion m^3 of freshwater was consumed for hydropower production in China in 2010. However, they used water consumption factors at subnational or provincial levels, neglecting differences in evaporation across individual reservoirs.

A further complication to understanding power-related water dependence is that different power plants may use different types of water resources. Four types of water can be discerned: surface water, groundwater, reclaimed water, and seawater. Depending on the technology installed, thermal power plants have the potential to use all four water resources, whereas hydropower uses surface water only. Without distinguishing these water uses, studies omit important factors for estimating freshwater security. For instance, in contrast to Zhu et al. [205] where the freshwater consumption of thermal power was not reported separately (and which found 10.2 billion m^3 of water consumed in China in 2010), Liao et al. [202] focused on freshwater and found that around 3.8 billion m^3 was consumed in that year.

Water is used for power production and then transmitted, virtually, across the power transmission network. Chini et al. studied the virtual water flows of the US electric grid, finding freshwater transfer of 11.2 billion m^3 in 2016 [5]. In China, around 13.7% of total national electricity was transported inter-provincially via transmission in

2011 [172], growing to 16% in 2017 [51]. This implies an increasing amount of water withdrawal and consumption which is serving other provinces as where withdrawal or consumption takes place. The volume of virtual water transfer via transmission has already raised concerns in China [118, 206]. He et al. assessed China's virtual water transfer at the subnational level, finding that virtual water transfer accounted for 9% of the total water consumption for electricity generation in 2016 [207]. Zhang et al. showed that the volume of virtual water transfer increased by a factor of 1.5 between 2006 and 2016 (however, hydropower was not considered) [208]. Zhang et al. used national average water intensities and showed that virtual water in electricity transmission increased five-fold between 2005 and 2014 [209]. Zhu et al. [210] analyzed virtual water transfers of the West-East Electricity Transmission project in China and found that 2.4 billion m^3 of virtual water was transmitted eastward in 2017. Previous studies have shown 1.4 billion m^3 of water consumption transfer through electricity transmission among China's six subnational regions in 2012 [211], and 6.3 billion m^3 of withdrawal transfer between China's provinces in 2014. Wang et al. [206] looked at the impacts of electricity transmissions on water scarcity at the river basin level rather than provinces, and calculated changes in the water-stressed population. Our work differs from these previous studies as follows. We explicitly differentiate between withdrawal and consumption, and types of cooling water used (in which particularly the differentiation between fresh and seawater is essential). Further, previous studies did not examine the seasonal variations in water transfer.

Virtual water transfer via power exports could reduce overall water withdrawal and consumption for power generation in China if the exporting region has higher water productivity or availability than the importing region [118]. Conversely, in the opposite situation power transmission may have negative impacts on water use. These dynamics are rarely quantified. An identical amount of water consumption may result in different impacts on exporting or importing regions with different levels of water stress [212]. Zhang et al. [213] tried to link power-related water transfer and regional water stress for China, but this study had limitations that it was based on water use factors from studies on the US and did not differentiate between types of water resources. Zhang et al. [214] quantified the impact of the spatial distribution of power generation on water consumption at a provincial level in China, indicating that transferring part of power generation tasks away from water-deficient areas could have significant impacts on the mitigation of water scarcity. However, more detailed technical causes of water use and transfer could not be deeply analyzed since it was not conducted at the plant level.

We compiled a state-of-the-art database of over 5,000 power-generating units and a model of the Chinese power transmission network. We investigated the water use of power plants and virtual water transfer by power transmission. Water consumption and water withdrawal are differentiated for a better understanding of water use. Water types (surface water, groundwater, reclaimed water, and seawater) are also distinguished in the assessment. The extensive inventory of plant information allows for a detailed analysis of the drivers of spatio-temporal variations in water use and transfer. Besides, we presented the impact of power production and transmission on provincial water stress using a metric of 'power-related water use to availability' (UTA_p), which is the ratio of power-related water use to regional water availability. Meanwhile, a counterfactual scenario was set to assess the counterfactual UTA_p that would be at stake if the province would fully generate its own power, and make a comparison to the actual UTA_p related to power consumption in a province. The difference between actual and counterfactual UTA_p represents the contribution of power transmission in terms of increasing or ameliorating water stress [35]. This work represents a significant improvement to the understanding of the energy-water nexus across China at multiple spatial scales and water resource levels. This work also provides a template for similar analyses in other nations.

3.2 Materials and Methods

3.2.1 Materials

Power: we include coal, gas, biomass, geothermal, nuclear and hydropower. Thermal plant information included: plant name, installed capacity, the beginning year of operation, unit type, location, operation status, cooling system, and monthly electricity generation. Hydropower information included: plant name, installed capacity, year of operation start, location, operation status, reservoir area, and electricity generation. Data were sourced from the Global Coal Plant Tracker [38], World Electric Power Plants Database [215], GRanD v1.3 [216] and Liu et al. [204]. For the cooling system of thermal power plants, we used Google satellite imagery cross-checked with information from the China Electricity Council [217]. We collected the installed capacity of each plant and used the monthly provincial capacity factor to estimate the electricity generation of each plant. The provincial capacity factor was calculated by dividing the provincial power production by provincial installed capacity. Both provincial power production and installed capacity are provincially available data, obtained from the Chinese National Bureau of Statistics [218] and China electric power yearbook [219] respectively. The compiled database covered 96%, 75%, 50% and 23% of the national installed capacity for coal, nuclear, hydropower and gas power plants, respectively. In total, 5408 power production units were included. The information of these units was used to assess the provincial power-related water use factors and total water use.

Transmission: The power transmission data in 2017 were obtained from the China Electricity Council [51]. These data are mostly reported in the form of province-to-province transmission. A small amount of transmission data are from provinces to the subnational grid. We disaggregate them into the province-to-province transmissions based on actual electricity transmission lines [172, 220]. Monthly provincial power transfers were obtained from the Professional Knowledge Service System for Energy [221]. The provinces are shown in Figure S7.2.1a.

Water consumption and withdrawal: We use China-specific water withdrawal and consumption factors for power plants. Specifically, water use factors of most coal power units were obtained from the National Energy Efficiency Benchmarking Competition [217]. We also obtained data for other power units from the inventory compiled in our previous study [16]. Finally, some once-through cooling water withdrawals were obtained from Zhang et al. [23], who used the monitoring data of withdrawals for some plants with once-through cooling systems in the Yangtze River basin (The nine river basins are shown in Figure S7.2.1b). A once-through cooling system is a technically simple system, which requires large amounts of water withdrawal and directly returns most of that water to its source. In a closed-loop (wet tower) cooling system, water goes through a cooling tower where some of the water is consumed through evaporation. Closed-loop cooling generally withdraws less but consumes more water than once-through cooling. An air cooling system uses air-cooled condensers for steam cooling and can avoid evaporative water losses [200]. The water type was obtained from the China Electricity Council [217] and the Power Industry Statistical Information System [222]. For hydropower, water use was determined by the evaporation factor and reservoir area. The reservoir evaporation factor for each month was extracted from the Noah Land Surface Model [223]. Water consumption and withdrawal within basins were obtained from the World Resources Institute Aqueduct database [224] and adjusted for the year 2017 using China's water use data from the National Bureau of Statistics [201]. The provincial available water, comprising both surface water and groundwater supply, was obtained from the Water Resources Bulletin [225]. Supplementary information discusses in more detail how we build our database with these data sources above and presents more specific information on each power plant. Two assumptions were made for water use assessment: that the water use factors of thermal power plants and the reservoir area of hydropower plants were assumed to be constant throughout the year.

3.2.2 Methods

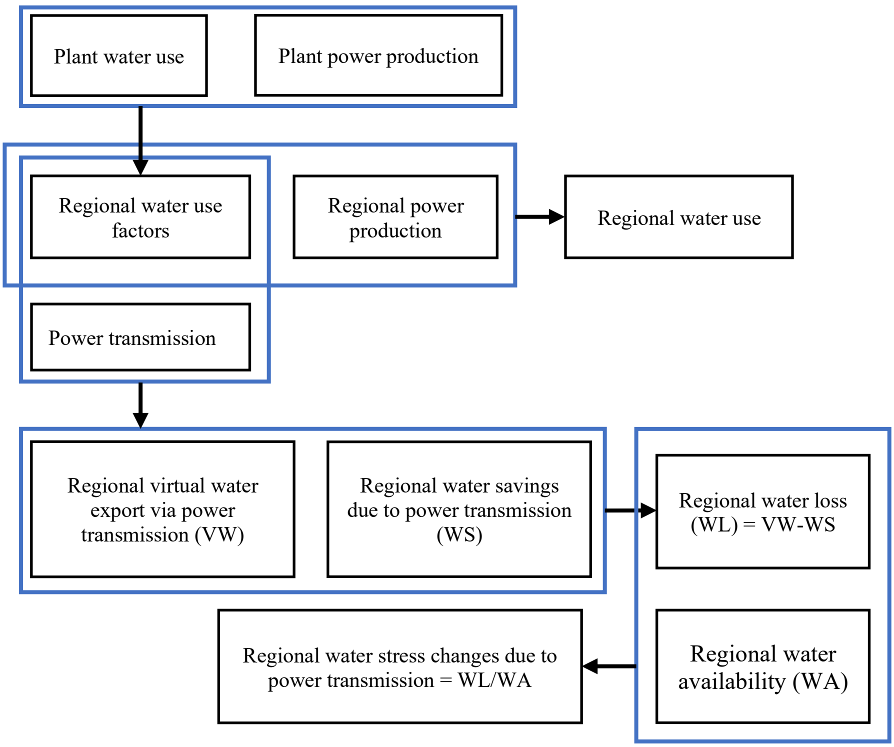


Figure 3.1 Flow chart of the calculation process.

The modeling schematic is shown in Figure 3.1. We outline each step in detail below but provide a brief overview here. First, we use individual plant data to estimate regional water use factors. We then combine this with data on regional power production and regional power transmission, and we can assess regional water use and virtual water transfers. Finally, we examine the impacts of the electric power system on water stress.

3.2.2.1 Calculating the water use of power production

A bottom-up approach was used, discerning six power production technologies: coal, gas, biomass, geothermal, nuclear and hydropower. Wind and photovoltaic power technology were not included because they consume negligible water. Four types of water resources were considered: surface water, groundwater, reclaimed water and seawater. Water use was specified as water consumption and water withdrawal. The water use of power production can be estimated in several steps.

1. The first step is to estimate water use at the plant level, beginning with thermal power:

$$WU_i = F_i \cdot PP_i \tag{1}$$

In which, WU_i denotes the water use of power plant i (m^3); F_i denotes the water use factor of power plant i (m^3/MWh); PP_i denotes the power production of power plant i (MWh). The water use of hydropower plants is calculated as follows:

$$WU_i = E_i \cdot A_i \cdot \eta_i \cdot 1000 \tag{2}$$

In which, E_i denotes the evaporation factor at the site of hydropower plant i (mm/month); A_i denotes the reservoir area of hydropower plant i (km^2); η_i is a dimensionless parameter to allocate the water use of a reservoir to hydropower plant i, determined by the economic values of the different purposes of the reservoir [204]. The water withdrawal of hydropower is assumed to be equal to water consumption, i.e. the surface water evaporated from the reservoir.

2. The power production for each plant is calculated as:

$$PP_i = \frac{RPP_{m,r,e}}{CP_{r,e}} \cdot CP_{r,e,i} \tag{3}$$

Where $RPP_{m,r,e}$ denotes the provincial power production using technology e in region r in month m (MWh); $CP_{r,e}$ denotes the provincial installed capacity using technology e in region r (MW); $CP_{r,e,i}$ denotes the installed capacity of plant i with technology e in region r (MW). Equation (3) implies an assumption that in each province the power production for each plant is proportional to the installed capacity of this plant. This assumption is made based on the small differences in the capacity factor across plants according to the information (described in Table S7.2.1) on the capacity factor of 1111 electricity-generating units in 28 provinces.

3. The third step is to calculate water use at the regional level:

$$WU_{m,r,e.s,u} = \sum_i WU_{m,r,e,s,u,i} \quad (4)$$

Where $WU_{m,r,e,s,u}$ denotes the regional water use of power production from the power plants in the database we complied (m^3) in month m, region r, for power production technology e; water resource type s; water use type, i.e. water consumption vs. water withdrawal is given by u. The power production of plants in the database is aggregated to the regional level using:

$$PP_{m,r,e} = \sum_i PP_{m,r,e,i} \quad (5)$$

In which, $PP_{m,r,e}$ denotes the power production using technology e in region r in month m from the power plants in the database we complied (MWh); e is categorized into three types: hydropower, nuclear, and other thermal power (not six types due to data limitations on the availability of power production).

Then, the regional water use factor of each power-generating technology (WUF) can be obtained by dividing the total water use of plants by their total power production in the region.

$$WUF_{m,r,e,s,u} = \frac{WU_{m,r,e,s,u}}{PP_{m,r,e}} \quad (6)$$

Where, $WUF_{m,r,e,s,u}$ denotes the quantity of water of type s consumed/withdrawn per unit of power production from technology e in region r in month m (m^3/MWh). The regional water use (RWU) of power production can now be obtained:

$$RWU_{y,r,s,u} = \sum_m RWU_{m,r,s,u} = \sum_e WUF_{m,r,e,s,u} \cdot RPP_{m,r,e} \quad (7)$$

Where, $RWU_{y,r,s,u}$ gives the quantity of water of type s consumed/withdrawn for power production in region r in the year 2017 (m^3); $RWU_{m,r,s,u}$ gives the quantity of water of type s consumed/withdrawn for power production in region r in month m (m^3). The regional water use factor of power production can be obtained for assessing the virtual water transfer via power transmission:

$$RWUF_{m,r,s,u} = \sum_e WUF_{m,r,e,s,u} \cdot \frac{RPP_{m,r,e}}{RPP_{m,r}} \quad (8)$$

Here, $RWUF_{m,r,s,u}$ denotes the quantity of water of type s consumed/withdrawn per unit of power production in region r in month m (m^3/MWh); $RPP_{m,r}$ denotes the total power production in region r in month m (MWh).

$$RWUF_{y,r,s,u} = \sum_m WUF_{m,r,e,s,u} \cdot \frac{RPP_{m,r}}{RPP_{y,r}} \quad (9)$$

Where $RWUF_{y,r,s,u}$ denotes the quantity of water of type s consumed/withdrawn per unit of power production in region r in the year 2017 (m^3); $RPP_{y,r}$ denotes the total power production in region r in the year 2017 (MWh).

3.2.2.2 Calculating virtual water transfer via power transmission

The virtual water transfer across regions can be estimated by:

$$VW_{i,j,s,u} = RWUF_{y,i,s,u} \cdot PT_{i,j} \quad (10)$$

Where $VW_{i,j,s,u}$ denotes the virtual water exported from region i to j through power transmission (m^3); $RWUF_{y,i,s,u}$ denotes the quantity of water of type s consumed/withdrawn per unit of power production in region i in the year 2017 (m^3/MWh); $PT_{i,j}$ denotes the yearly power transmission from region i to j (MWh). Power importing regions save water, which can be estimated by using a counterfactual where a region does not import power but satisfies the local demand by producing power itself using the expression:

$$WS_{j,s,u} = RWUF_{y,j,s,u} \cdot \sum_i PT_{i,j} \quad (11)$$

Where $WS_{j,s,u}$ denotes the water-saving in region j by importing power (m^3). The impact of power transmission on regional water use can now be estimated:

$$WL_{j,s,u} = \sum_i VW_{j,i,s,u} - WS_{j,s,u} \quad (12)$$

In which $WL_{j,s,u}$ denotes the water loss in region j due to power transmission (m^3). A negative value of $WL_{j,s,u}$ indicates that in region j the water-saving achieved by importing power is larger than the water export by exporting power, thus region j saves water through power transmission; a positive value of $WL_{j,s,u}$ indicates that there is water loss in region j.

3.2.2.3 Assessing the impact of the power system on water stress

Power production is a large water user, but its impact on water use differs across regions due to the differences in power-generating technologies. Also, regional water use already differs across the country. The proportion of the power-related water use to the total water use is estimated at the basin level using:

$$WP_{b,u} = \frac{\sum_i WU_{b,u,i}}{WU_{b,u}} \quad (14)$$

Where $WP_{b,u}$ denotes the proportion of the power-related water use to the total water use in basin b; $WU_{b,u,i}$ denotes the water use of power plant i within basin b (m^3); $WU_{b,u}$ denotes the total water use of basin b (m^3). To assess the changes in water stress caused by power transmission, we use the above counterfactual method. The indicator use-to-availability (UTA_p), which is the ratio of power-related water use to regionally available water resources [168], was used to assess the impact of power transmission on regional water resources. Specifically, both water consumption and water withdrawal were considered, i.e. we calculated both CTA_p (consumption-to-availability) and WTA_p (withdrawal-to-availability) using:

$$PCTA_p = \frac{RWU}{WA} \tag{15}$$

$$CCTA_p = \frac{RWU-WL}{WA} \tag{16}$$

$$DCTA_p = CCTA_p - PCTA_p \tag{17}$$

Where $PCTA_p$ denotes the CTA_p driven by the present power system; $CCTA_p$ denotes the counterfactual CTA_p without power transmission; $DCTA_p$ denotes the CTA_p decrease induced by power transmission, a positive value means provincial CTA_p is reduced via power transmission; RWU denotes the provincial water consumption for power production; WL denotes the water consumption increased by power transmission (eq.12); WA denotes the provincial water availability. WTA_p is calculated analogously.

3.3 Results

3.3.1 Water use characteristics at the plant and national level

In 2017, 14.6 billion m^3 of water resources were consumed for power production in China, comprising of 12.8 billion m^3 surface water, 0.23 billion m^3 groundwater, 0.27 billion m^3 reclaimed water and 1.25 billion m^3 seawater. Hydropower was responsible for 68% of the surface water consumed by power generation. Power plants using groundwater and reclaimed water are generally situated in northern China (Figure 3.2a). In northwestern China, both direct and indirect air-cooling systems are commonly used to save water. There are many power plants with closed-loop cooling systems located downstream of the Yellow River basin, which increases local water consumption. All nuclear power plants in China are located along the coastline and use seawater for cooling. Coal power plants comprise 4 billion m^3 of freshwater consumption in 2017.

With respect to total water withdrawals (as opposed to consumption), power production in China withdrew 62.7 billion m^3 of freshwater, which amounts to approximately 10% of the national total for all sectors in 2017. In this study, we define freshwater as surface water and groundwater [225]. Compared to other regions, water withdrawal in southeastern China is much larger due to the preponderance of once-through cooling systems at generation units, cooling systems that require larger water withdrawals. Many of these once-through plants are in the Yangtze river basin (Figure 3.2b) and aggravate local water competition (Figure S7.2.2). There are heterogeneities in freshwater use of power production across plants. 80% of power production withdrew just 10% of the total water for power in China whereas the remaining 20% withdrew 90%. Large water withdrawers are hydropower with large reservoirs and thermal power plants with once-through cooling systems.

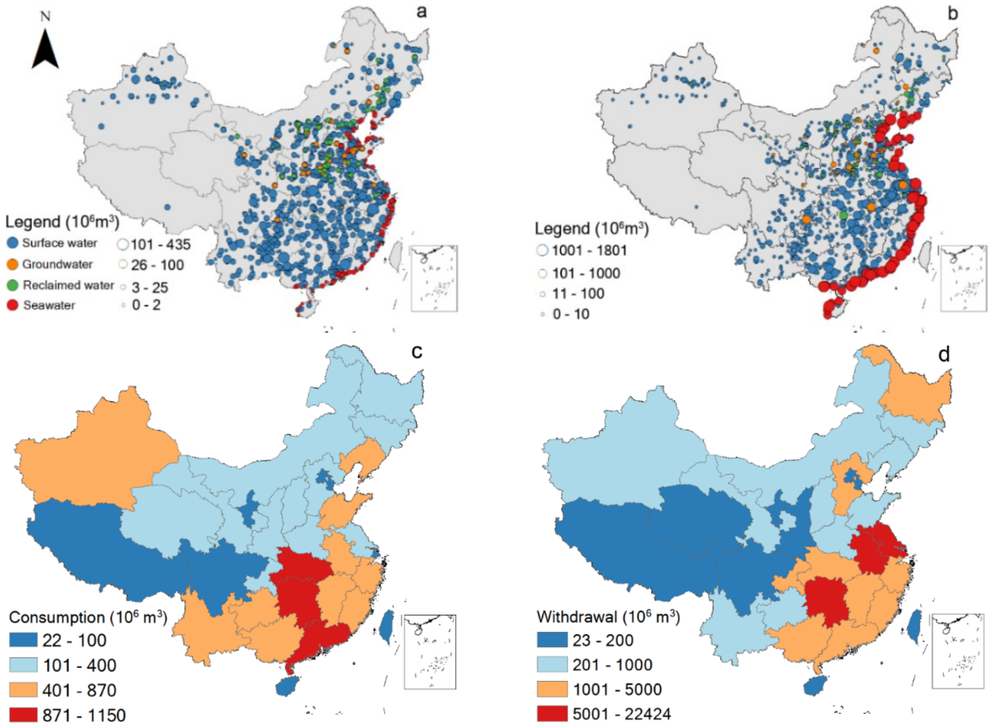


Figure 3.2 Water consumption (a, c) and water withdrawal (b, d) of power at the plant and province level in China in 2017. For more details of plants' location, water use and power output, see Supplementary information.

Power-related water use differs widely across provinces (Figures 3.2c-d). The top three freshwater consumers are Hunan (1.2 billion m^3), Hubei (0.9 billion m^3), and Guangdong (0.9 billion m^3). Hunan province is the largest freshwater consumer due to the high water consumption factor of hydropower. The top three freshwater withdrawers are Jiangsu (22.4 billion m^3), Shanghai (5.8 billion m^3), and Anhui (5.4 billion m^3). Jiangsu sees the largest freshwater withdrawal due to its large-scale power production and wide use of once-through cooling systems. In terms of monthly figures, July sees the largest consumption and withdrawal in Hunan (147 million m^3) and Jiangsu (2261 million m^3), respectively. Groundwater is mainly consumed in water-scarce regions: Hebei (65 million m^3), Shandong (62 million m^3), and Inner Mongolia (25 million m^3). Reclaimed water is mainly used in the northern regions: Liaoning (65 million m^3), Hebei (36 million m^3), and Beijing (35 million m^3). Seawater is used in coastal provinces such as Jiangsu, Guangdong, and Zhejiang. From the perspective of power type, hydropower dominates surface water consumption (9 billion m^3), while thermal power dominates surface water withdrawal (52 billion m^3). The descriptive statistics of electricity-generating units by region are shown in Table S7.2.2.

3.3.2 Power transmission transfers freshwater across provinces

Provinces export virtual water by exporting power and import virtual water by importing power (Figure 3.3). As the largest electricity importer, Guangdong sees a virtual inflow of 0.58 billion m^3 in water consumption, mainly by importing hydropower from Yunnan. Large volumes of water withdrawals are exported from Anhui, where once-through cooling systems are used for thermal power, to Jiangsu (0.55 billion m^3) and Zhejiang (0.7 billion m^3) provinces. Liaoning and Guangdong export large amounts of virtual seawater used for nuclear power plants. Focusing on freshwater, power transmission accounts for total virtual water withdrawal and consumption of 6.2 and 2.1 billion m^3, respectively. Compared with the counterfactual scenario with no interprovincial power transmission, power transmission reduced national freshwater withdrawal by 10.1 billion m^3 but increased consumption by 0.21 billion m^3 in 2017 (Table S7.2.3). These counterintuitive results are caused by the differences in electricity technologies and cooling systems between western and eastern regions. As shown in Figs. 4a-b, in Ningxia, Sichuan and Yunnan provinces, power production is much larger than local power demand and more than 35% of power-related water use is exported to other provinces via power export. The proportion is less than 5% for 11 provinces, most of which are in the developed Southeastern and Northern regions with high power demand.

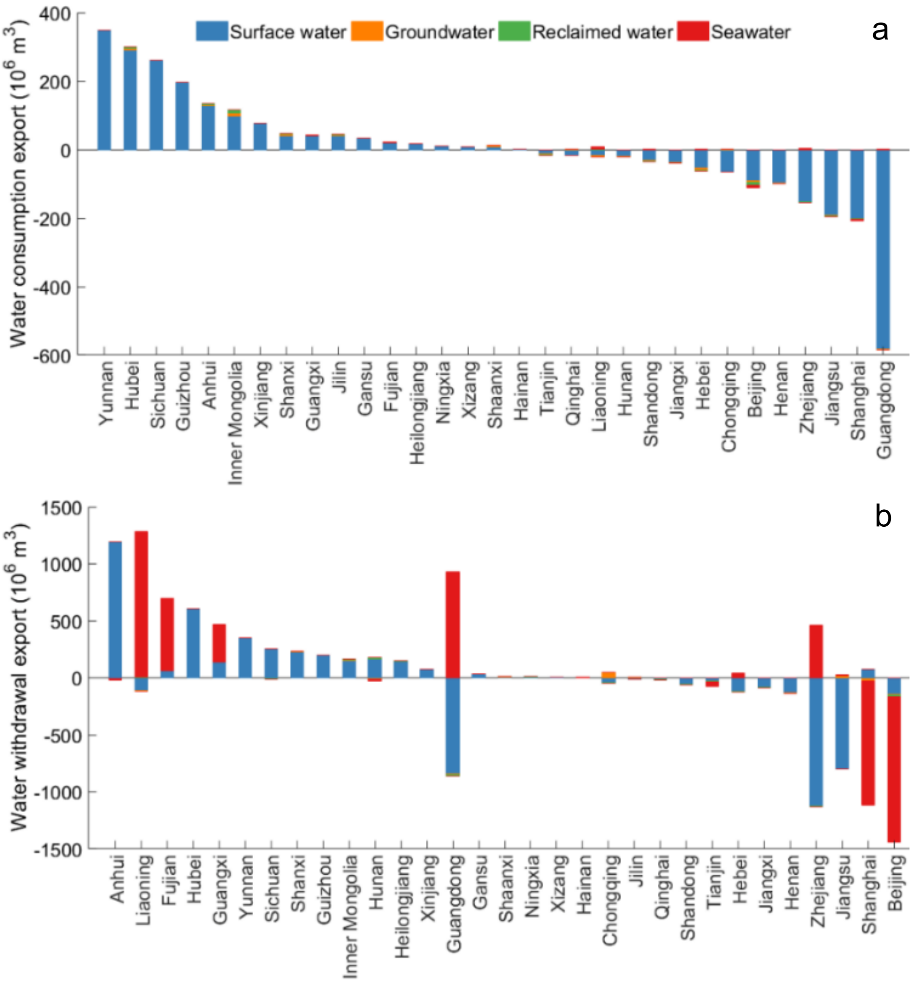


Figure 3.3 Net exports of water consumption (a) and water withdrawal (b) across regions in China in 2017. The positive values designate net water export, while the negative values designate net water import.

Large seasonal variations exist in inter-regional water transfer (Figures 3.4**)**. Specifically, both consumption and withdrawal transfer peaked in August. There are two causes: first, large amounts of electricity were transferred from southwest to south and east in summer due to the high power demand of cooling, especially in economically developed regions, such as Guangdong, Shanghai, Jiangsu, Zhejiang, and Beijing. In the peak month of August, 19% of the national electric power is transferred across provinces, 5% larger than in February. Second, higher temperatures during the summer cause higher evaporation, leading to higher water exports from hydropower plants in the southwest and central provinces. Specifically, the median provincial water consumption factor of hydropower varies significantly throughout the year, with 3,355 and 21,133 m^3/GWh in February and August respectively. In the peak month, the largest exporter and importer of water consumption are Yunnan (43 million m^3) and Guangdong (76 million m^3), while the largest exporter and importer of water withdrawal are Anhui (126 million m^3) and Jiangsu (147 million m^3). The water export of the central region, which does not exhibit the typical peak in August, is determined by Hubei province. There is a trough in the water export of the central region in August because of the decrease in power export in Hubei province from 10.7 TWh in July to 8.4 TWh in August.

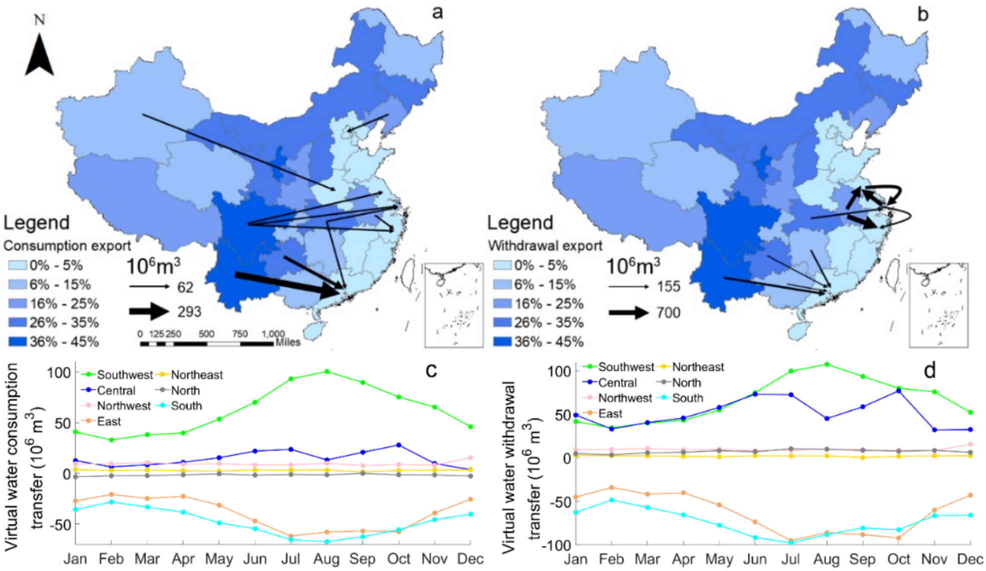


Figure 3.4 Interprovincial transfer of freshwater consumption (a) and freshwater withdrawal (b), the colour shows the proportions of provincial water consumption and withdrawal of power production that are exported respectively; Monthly net transfer of freshwater consumption (c) and freshwater withdrawal (d) via power transmission in China in 2017. A positive value means virtual water export in a region is larger than import. Provinces are classified into seven regions as in Cai et al. [226]. The transfers of all water types are provided in Supplementary information.

Nationally, seasonal virtual water consumption transfer is dominated by hydropower, while withdrawal transfer is dominated by thermal power (Figure S7.2.3). Yunnan, Sichuan and Hubei are large virtual water exporters via hydropower, exporting between 264-329 million m^3 each. The top 3 transmission corridors of hydropower-related virtual water consumption transfer are Yunnan to Guangdong (277 million m^3), Guizhou to Guangdong (99 million m^3), and Sichuan to Zhejiang (79 million m^3) (Figure S7.2.4). Inner Mongolia, Anhui and Guizhou are large virtual water exporters via thermal power, exporting between 71-94 million m^3 each. The top 3 transmission corridors for thermal power-related virtual water consumption transfer are Guizhou to Guangdong (55 million m^3), Anhui to Zhejiang (43 million m^3), and Anhui to Jiangsu (34 million m^3) (Figure S7.2.5). There are considerable flows of virtual water withdrawal among Anhui, Jiangsu, Shanghai, and Zhejiang provinces, mainly due to the use of once-through cooling systems for thermal power. Since these provinces are within the eastern region, their withdrawal transfers do not involve the other six regions. The east and south are net water importers throughout the year.

3.3.3 The impacts of the power system on water stress

Power-related virtual water transfer through transmission networks changes provincial water stress for both freshwater consumption (Figure 3.5a) and withdrawal (Figure 3.5b). We define the water stress in terms of the consumption-to-availability (CTA) ratio and the withdrawal-to-availability ratio (WTA). Power transmission exacerbates issues when the CTA or WTA is larger than in the counterfactual scenario. Overall, water stress was more equally distributed through power transmission. The relative standard deviations of provincial CTA (from 130% to 127%) and WTA (from 186% to 136%) decreased through power transmission. Yunnan province sees the largest increase in CTA with 2.2%, mainly due to hydropower exports. In terms of WTA, power transmission appears to be an effective way to help reduce pressures in regions such as Shanghai, Jiangsu, and Chongqing, where WTA reduces 46%, 7%, and 5% respectively. If Shanghai were to satisfy its power demand itself, the freshwater demand for power production would exceed the total current water supply, unless it were to shift to cooling systems with a lower water intensity or that use more seawater. Anhui province contributed to the water stress alleviation in the above regions, with a WTA increase of 4.2%.

The environmental impact of power production depends on both provincial water consumption and water scarcity. Provinces are classified into four categories according to provincial water consumption and water stress index (WSI) (Figure S7.2.6). The provincial WSI is assessed in this study according to Scherer et al. [227] using annual withdrawal data, and a WSI of 0.5 defines the threshold between medium and high water scarcity [228, 229]. For water consumption, the median value (373 million m^3) is used as the line between medium and high water consumption. There is large heterogeneity in water scarcity across provinces. Shandong, Xinjiang, Liaoning supply large amounts of freshwater for power production even though they face severe water scarcity. Hunan and Hubei have low water stress despite large amounts of power-related freshwater consumption.

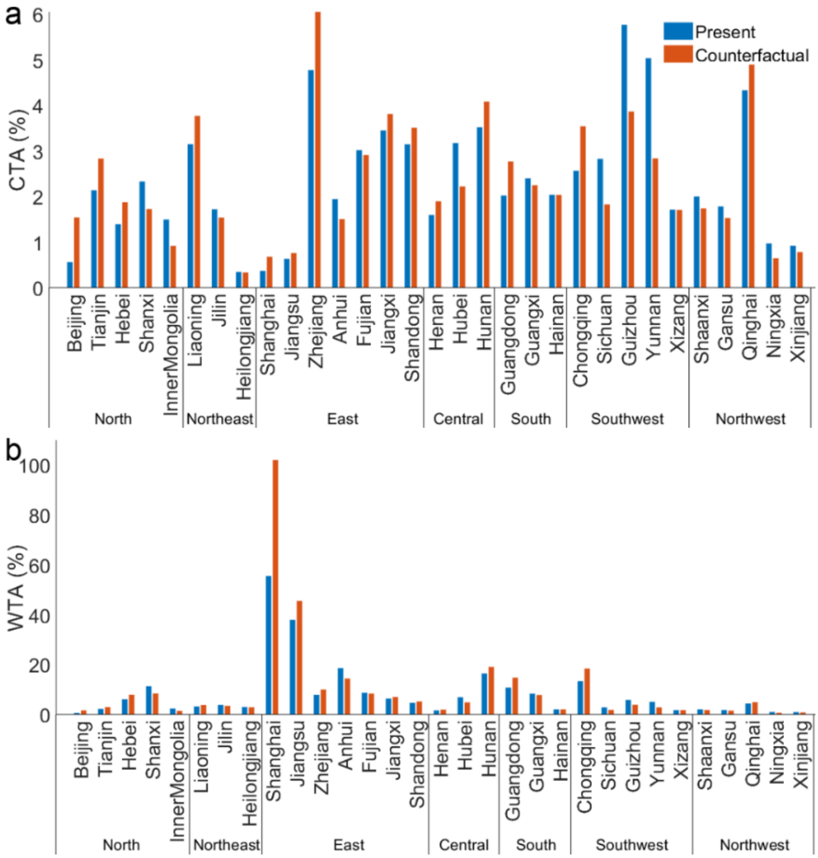


Figure 3.5 The changes in provincial CTA (a) and WTA (b) caused by interprovincial power transmission in 2017.

3.4 Discussion

3.4.1 Trends in water use and transfers of virtual water through the power system

Several recent trends are particularly important for the electric power system and its water use in China. First, the volume of freshwater withdrawal for thermoelectric cooling has decreased since 2011 due to increasing numbers of air-cooled and seawater cooled units [23, 230]. As once-through cooling systems are being phased out and replaced with more efficient systems in terms of water withdrawal [231], the total withdrawal for power production is expected to continually decrease. Meanwhile, since air cooling systems are increasingly used for new plants and there is no significant increase of closed-loop cooling plants, thermal power plants will very likely also reduce water consumption per unit of electricity production in the future. In any case, water consumption is mainly driven by hydropower plants and is hard to reduce given that annual reservoir areas do not change greatly. While this study focuses on annual water scarcity, there may be large seasonal variations, and the impact of hydropower on water scarcity is often alleviated by storing water in the wet season and releasing it in the dry season [24]. Second, although thermal power and hydropower still dominate power production, renewables and their low water-intensity are expanding quickly, especially wind power and photovoltaic (PV), with increases in capacity of 12.4% and 34% in 2018, respectively [232]. Due to reduced costs and decreased power curtailments, the expansion of wind and PV in western and northern China is expected to reduce local water export and alleviate local water stress [233]. Nuclear power is also growing, but will not put pressure on freshwater resources as only seawater is used.

Third, despite the rapid expansion of low water-intensity technologies, there are still large uncertainties in total water use in the future due to the growth of China's power demand. Coal power and hydropower generation increased by 5.3% and 3.2% in 2018 respectively [232]. Also, increasing hydropower production is crowding out thermal power in many provinces, especially Yunnan, Zhejiang, Fujian, Hunan, and Guangdong [234], which would increase water consumption. Fourth, as part of air pollution mitigation, China began the construction of twelve long-distance power transmission lines in 2014 [235]. These lines transmit inland electricity eastwards to coastal areas [236]. Ten lines carry mainly hydropower and coal electricity, with the other two carrying both coal and wind power [118]. Among the twelve lines, there are eight ultra-high voltage, all completed at the end of 2017 [219]. Nationally, these lines are contributing to an increase in interprovincial water transfer. The power transmission of the transmission corridor from Yunnan to Guangdong has increased rapidly (Figure S7.2.7) and virtual water transfer is expected to continue increasing. The transmission corridor between Guizhou and Guangdong is the largest for both hydropower- and thermal power-related virtual water transfer because of its large amount of power production and transmission of both energy types.

Last, in recent years China has been paying more attention to groundwater resources and has banned its use in new power plants and units in the northern water-deficient areas since 2004 [231]. Combining our results with Liao et al. [42], we see that although groundwater is still consumed for power production in the Huang-Huai-Hai area of northern China, the volume consumed is significantly decreasing. In the coming years reclaimed and surface water will substitute groundwater in many areas of northern China due to stricter regulations on water use and the completion of the 'South-to-North Water Diversion' project. Currently, the freshwater requirement is high in some coastal regions such as Jiangsu, Shanghai, and Guangdong. However, seawater use is expected to increase in these regions with the development of seawater treatment projects encouraged by the government [217, 237]. Although hydropower is the largest water consumer among all energy technologies previous studies and government reports usually neglect the technology.

3.4.2 Comparison with previous studies

Our estimates show that in 2017 the freshwater consumption for the cooling of thermal power was 4.3 billion m^3, falling within the range of published values [23, 168, 202]. This study also examines the detailed technical causes of water use. 54% of the coal-power units are equipped with closed-loop cooling systems, resulting in a high level of water consumption. As for freshwater withdrawal, Zhang et al. [23, 168] found 68 billion m^3 in 2010 and 57 billion m^3 in 2015 for thermal power. Our estimates suggest that this was further reduced to 52 billion m^3 by 2017. The capacity factor is a key variable in water use assessment but has been seldom indicated in previous studies. We show that both the median and average capacity factor of coal power units are 70% in China in 2017. Zhu et al. [205] and Liao et al. [202] estimated the water consumption of China's hydropower production, indicating water consumption between 11.5 and 14.6 billion m^3 in 2010. Both studies were conducted at the regional level using national or provincial averaged water consumption factors for assessments but did not consider the differences in the reservoir area and evaporation rate across regions. This study is based on plant-specific data, including evaporation factors. The results show the large differences in the water consumption factors of hydropower across provinces, demonstrating the importance of the high spatial detail. Liao et al. [118] indicated that power transmission could save 20.1 billion m^3 of water withdrawals nationally in 2014 due to differences in water productivity in exporting and importing provinces, but water resources were not specified. Our study distinguishes water types and we show that power transmission saved 33 billion m^3 of water withdrawal in 2017, but 69% is seawater, which would not reduce regional freshwater stress. The technical details show that in the western power-exporting provinces, such as Guizhou, Sichuan and Yunnan, hydropower and the thermal power plants with closed-loop cooling systems are commonly adopted, whereas the eastern power-importing provinces, such as Shanghai, Jiangsu and Anhui, have more plants with once-through cooling systems. Compared to the east, the west generally has a larger water consumption factor but a lower withdrawal factor. Power generation that consumes large amounts of water is often transmitted from west to east [206], consequently reducing water withdrawal but increasing water consumption nationally. Zhang et al. [172] estimated virtual water transfers through interprovincial power transmission, finding 0.6 billion m^3 of water consumption was transferred in 2011. However, water consumption of hydropower was not included, which is crucial in some province-relationships, e.g. from Yunnan province to Guangdong province. Interprovincial power transmission was 1.6 times higher in 2017 than in 2011 [51, 172], which is also a contributor to the increase of water transfer. Besides, previous studies were on a yearly resolution, while our results show the variations in monthly water transfer.

Thermal power production accounts for 45% of total water withdrawal in the US [238] and 43% of total freshwater withdrawal in Europe [239]. This is explained by the wide use of once-through cooling systems using freshwater across the US and Europe [240]. In contrast, China's total power production is responsible for only 10% of national freshwater withdrawals. Though power plants with once-through cooling systems in China account for 21% of all plants, only 48% of them use freshwater. Furthermore, the relatively high water withdrawal in the US and Europe may be a result of strict temperature regulations since power plants have to withdraw more water for heat discharge in order to keep the cooling water temperature under limits [241, 242], whereas

China has only vague guidelines on water temperature [243]. It is important to note that low withdrawal does not mean low consumption. China withdraws much less freshwater for annual thermal power production (54 billion m^3) than the US (230 billion m^3) but sees higher consumption (4.3 billion m^3 compared to 4 billion m^3 in the US) [238]. Previous research showed that seasonal variation has a significant influence on power demand for many countries (e.g. India, Algeria, and Germany) [244-246]. This influence can result in variations in power-related water use and virtual water transfer. Moreover, the seasonal variations in virtual water transfer differ across regions. Electricity and virtual water transfer peak in the winter (due to heating demands) in Europe [30], whereas they peak in the summer in China (due to cooling demands). For countries like China and Brazil, wind and solar power are concentrated in some subnational regions [247, 248]. Improving the interconnection of electricity transmission across these regions not only reduces energy curtailment but can also conserve water resources.

3.4.3 Limitations and implications

Although we examined a large database, we were unable to include all power plants. Since 50% of hydropower plants were covered and the average water use factor of them was applied for other unknown plants, there would be uncertainty in the total water use estimate for hydropower. However, since we separately calculated the water use of thermal power and hydropower, the results did not have a bias in the water use of hydropower in the power mix. In China, coal, hydropower and nuclear power dominate power production, while gas power plants account for less than 5% of the total in 2017 (and oil power accounts for only 0.05% so it is not included due to data limitation) [219, 249]. For future global research, efforts are needed to compile a more complete database of power plants. We estimated the power production of individual plants based on installed capacity per plant and the monthly provincial power production data. Precise information on the actual power production at the plant level would allow for more detailed insights. The water use factor of thermal plants was assumed to be constant throughout the year yet plants often have higher water requirements in summer than in winter due to lower thermodynamic efficiencies [16, 241]. The reservoir area of hydropower plants was also assumed to be constant throughout the year. Both assumptions would lead to an underestimation of the seasonal variations in power-related water use. Our database covers 80% of the total installed capacity of thermal, nuclear, and hydropower in China. Considering the water demand for power and the current coverage of capacity, efforts are needed to include more gas and hydropower plants in the future. For a better understanding of power-related water use, we suggest power data at a higher spatio-temporal resolution should be provided by power suppliers, such as capacity factor and monthly power production at the plant level, and more detailed province-to-province power transmissions.

The electric power system poses threats to water stress in some regions. There are several ways to reduce the dependence of power system on freshwater: first, we can reduce the water demand of power by developing photovoltaics and wind power, using air cooling systems, replacing coal with gas, and improving the capacity factor of hydropower; second, more seawater and reclaimed water should be used for cooling. Third, we can improve power transmission from low water-scarce to high water-scarce regions. By improving the interconnectivity of electricity grids, capacity in high water-scarce regions can be downsized and less affected by freshwater scarcity. A quantitative analysis is out of scope for this paper. Our study provides an international perspective in terms of the application of methods and results. First, the methods can be applied to other nations if sufficient data on power production and water use are available. Specifically, in this study, the electric power system is examined from two perspectives: power production and power transmission. The method of assessing water use for power production we present here can be applied to other nations if plant data are available, particularly the cooling type of thermal power plants and the reservoir area of hydropower plants. For nations where plants' water use is not available, the methods of studying the impacts of power transmission on water stress can still be applied by using national water use factors, though it would entail increased uncertainties. Second, our results for China, as one of the major energy users worldwide, can be used as an important part of a database of global energy-related water use and thus support further analyses of global water use and transfer. Third, the general implications of our study also apply to other countries, e.g. related to the risk to increase water stress through power transmission, the trade-offs between water withdrawal and consumption changes, and the differences between technologies.

3.5 Conclusions

This study assessed the water use of power production in China for numerous renewable and non-renewable power-generating units, from the perspective of both water consumption and withdrawal. Water sources (surface water, groundwater, reclaimed water, and seawater) are distinguished in the assessment. Based on the results, we also examined the seasonal shift in water stress caused by power transmission. The following conclusions are drawn:

China's power production withdrew 62.7 billion m^3 of freshwater in 2017, of which 13 billion m^3 was consumed. There are large heterogeneities in the water use of power production across plants. Hydropower plants with large reservoirs are large freshwater consumers whereas thermal power plants with once-through cooling systems are large freshwater withdrawers.

Water stress was more equally distributed across provinces through power transmission. Nationally, power transmission reduced freshwater withdrawal but increased consumption in China because, compared to the east, the west generally has a larger water consumption factor but a lower withdrawal factor. Power-related water transfer varied greatly throughout the year, with an August peak due to the high cooling demands in the east and high reservoir evaporation in the southwest.

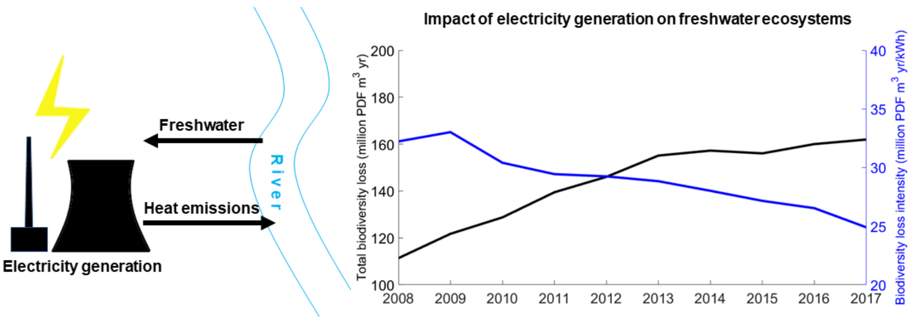
Chapter 4

Biodiversity loss from freshwater use for China's electricity generation

This chapter has been published as: Jin, Y., Behrens, P., Tukker, A., Scherer, L. Biodiversity loss from freshwater use for China's electricity generation. *Environmental Science & Technology*. 2022, 56, 5, 3277-3287.

Abstract: Electricity generation has two major, under-investigated impacts on freshwater biodiversity due to its water use: the consumption of freshwater and thermal emissions to freshwater. Here we analyze the spatio-temporal freshwater biodiversity impacts of China's electric power system and the driving factors for these impacts. We show that between 2008 and 2017, the freshwater consumption of electricity generation peaked in 2013 (13.6 Gm^3). Meanwhile the freshwater consumption factor of China's electricity generation decreased from 3.2 to 2.0 L/kWh. However, due to increasing thermal emissions the biodiversity loss via freshwater use increased from 1.1×10^8 in 2008 to 1.6×10^8 PDF m^3 yr. The overall biodiversity loss per unit of electricity generation decreased from 3.2×10^{-5} to 2.5×10^{-5} PDF m^3 yr/kWh. Biodiversity loss from thermal pollution is 60% higher than that driven by water consumption. Electricity transmission results in the shifting of biodiversity impacts across regions. The results show that 15% of total biodiversity loss was embedded in transmission networks. In terms of electrical power system drivers of biodiversity loss, the total generation was the main driving factor of the increase in loss (rather than shifts in generation type for example). Our results indicate the necessity of assessing the biodiversity impacts of electricity generation and incorporating them into energy system planning.

TOC Graphic



4.1 Introduction

While carbon emissions are a key environmental focus of electricity generation analyses, its biodiversity impacts have been largely overlooked [55, 250-253]. Biodiversity is a critical indicator of ecosystem health and provides many ecosystem services to society [254]. Human activities are causing an accelerating biodiversity loss at rates 100 to 1,000 times pre-human levels [255]. Current losses in biodiversity are considered critical and could threaten earth system functioning and its adaptive capacity [256]. Simultaneously, global electricity generation is growing quickly, dominated by thermal power (77% of the total) and hydropower (16%) in 2018 [37]. Linking electricity generation with biodiversity impacts can help deepen the understanding of biodiversity conservation and the energy transition.

Current electrical power systems require large amounts of freshwater in the thermodynamic conversion of heat to work or the water held in hydropower reservoirs. These processes can result in both consumption of water or the warming of the water in the environment (termed thermal emissions) [16, 61, 257]. Both freshwater consumption and thermal emissions have impacts on biodiversity [258, 259] (water consumption refers to the volume of water not returned to the water body due to evaporation, transpiration, or incorporation into products [16]). Research has shown that thermal and hydropower generators are major water consumers. Emerging renewables such as wind power and photovoltaic (PV) consume negligible water during operation [8].

Different power-generating technologies use water in different ways. Thermal power plants withdraw water for cooling [260, 261] and some of the water is consumed through evaporation [171, 179]. Liao et al. [202] and Zhang et al. [23] assessed the freshwater consumption of China's thermal power production and found freshwater consumption of 3.8 and 5.7 Gm^3 in 2010 and 2015 respectively. While hydropower is an important renewable energy source, it can consume a lot of water via evaporation from the reservoir surface [262, 263]. Estimates of water use for hydropower range widely [24]. For China, Liu et al. showed the water intensity of hydropower plants ranges from 13 to 15244 m^3 MWh^{-1} [204]. Zhu et al. [205] and Liao et al. [202] found that 11.5 to 14.6 Gm^3 of freshwater was consumed for China's hydropower production in 2010.

Despite the large water requirements of electricity generation, the aquatic biodiversity impacts of electricity generation have received little attention. Dorber et al., in the few examples of such an assessment, quantified the water consumption of Norwegian hydropower reservoirs and found that the impacts on fish species vary over six orders of magnitude [29]. Biodiversity impacts of electricity generation in China are of specific interest, as the two most biodiversity-threatening generation types, thermal and hydropower, together comprise 87% of national electricity generation (as of 2019 [264]). However, their water consumption-related biodiversity impacts have not been quantified in previous research.

In addition to water consumption, the heat transferred into cooling water from power plants and then returned to the water source also has biodiversity impacts [265, 266]. For thermal power, there are two common wet cooling types: 1) once-through cooling, requiring large amounts of water withdrawal and directly returning most of that water to its source; and 2) closed-loop cooling in which some of the water is consumed through evaporation [200]. Freshwater heat pollution is predominately from once-through cooling systems, which involves the direct rejection of the heat back into the water body [61, 267]. The temperature of discharged water from plants is higher than the natural river temperature and harmful to aquatic systems [268, 269]. In closed-loop cooling, almost all the heat absorbed during the steam cycle is removed via evaporation and dissipated into the atmosphere. The heat contained in the periodic cooling tower blowdown is negligible compared to the heat released in once-through cooling emissions [270]. Raptis et al. assessed the biodiversity loss caused by freshwater thermal pollution and showed the varying impact of electricity generation between countries [61]. Pfister and Suh assessed the impact of thermal pollution on freshwater ecosystems in the US, finding that the ecosystem impact for the different US electricity grids can differ by an order of magnitude [271]. Cheng et al. simulated the impacts of thermal pollution from power plants on the aquatic ecosystem, indicating that fishes can be heavily affected [272]. Hydropower stations also increase the temperature of the rejected water, but to a lesser degree than thermal power, so its impact on aquatic biodiversity was often neglected. The overall impact of thermal pollution from China's power production on aquatic biodiversity has not been fully understood.

Here we assess the impacts of both water consumption and thermal pollution for power production on freshwater biodiversity for the first time. We also extend the research to include hydropower. As with commodity trade, exchanges of electricity across large grids can result in the shifting of biodiversity impacts across regions. While international commodity trade can have significant biodiversity impacts (17-30% of global biodiversity loss [273]), China exchanges very little electricity internationally. However, the scale of interprovincial power transmission within China is large and increasing (with a 150% growth over 2008-2017). Electricity importers across China are outsourcing biodiversity impacts via transmission to other provinces and we capture these dynamics. Finally, we diagnose the driving factors for changes in biodiversity loss over time (including generation type, scale of electrical generation and others).

4.2 Methods and materials

4.2.1 Methods

The overall modelling approach is shown in Figure 4.1. First, we prepare the input data for assessments (grey box), i.e., the water consumption and thermal pollution of electricity generation, and province-level, generator-specific characterization factors. We then assess the biodiversity loss caused by electricity generation along with the embodied biodiversity loss via power transmission (yellow box). Based on these calculations, we examine the relationships between biodiversity loss and electricity generation along with the driving factors of biodiversity loss via electricity generation (green box).

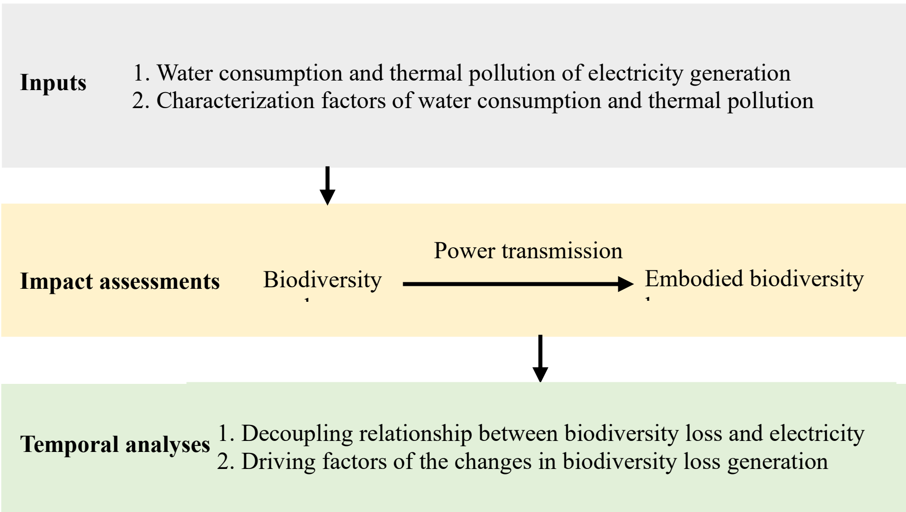


Figure 4.1 Overall schematic of the model.

4.2.1.1 Water use of electricity generation

We assess the provincial water consumption factors for thermal power and hydropower generation using the method described in Jin et al. [16]. Our database covers 96% and 50% of the national installed capacity for thermal power and hydropower, respectively. Thermal plant information included: plant name, installed capacity, the beginning year of operation, unit type, location, operation status, cooling system, and monthly electricity generation. Hydropower information included: plant name, installed capacity, year of operation start, location, operation status, reservoir area, and electricity generation. These representative plants are used to assess provincial water intensities (capacity-weighted water consumption of plants), which are then combined with provincial power production to assess water consumption. The total water consumption of electricity generation in each year is calculated as follows:

$$WC = \sum_i WC_i = \sum_i (TWC_i + HWC_i) \quad (1)$$

Where WC is the national water consumption for electricity generation (m^3); WC_i the water consumption for electricity generation in province i (m^3); TWC_i the water consumption for thermal power generation in province i (m^3); HWC_i the water consumption for hydropower generation in province i (m^3). For further details see Supplementary information S1.

4.2.1.2 Biodiversity loss

Among the three main types of ecosystems (terrestrial, freshwater, and marine), we focus on biodiversity impacts in freshwater ecosystems as much of the impact of water use inland is on freshwater systems [27]. We consider water consumption and water thermal pollution of electricity generation as drivers of biodiversity loss. Freshwater consumption results in reduced river discharge, which is one of the main threats to freshwater life [274]. The impacts can be assessed based on the species-discharge relationship. We consider fishes, given that this species group is larger than most other freshwater taxa [275] and they are better studied. Water consumption is translated to impacts on freshwater biodiversity using characterization factors (CFs) expressed as a potentially disappeared fraction of species (full unit: PDF m^3 yr /m^3) (Supplementary information S7.3.2) [276]. The increased river temperature caused by thermal pollution damages the ingestion and health of freshwater life and can lead to death [268]. The impacts can be assessed based on species sensitivity distributions, considering the temperature tolerance interval of aquatic species (among which we include fishes, mollusks, crustaceans, and annelids) [61]. Thermal pollution is calculated and translated to impacts by CFs with the unit of PDF m^3 yr /MJ [61] (Supplementary information S7.3.3). Ecosystem impacts refer to the fraction of species that is committed to becoming extinct ("potentially disappeared fraction of species" or PDF) if the pressure (e.g., water consumption) continues [27]. As there are typically lag times between the pressure and the effect, the duration of the pressure influences whether the full extent of the effect will happen or not. For this reason, the exposure duration (yr) to the pressure is also included in the unit of ecosystem impacts. Furthermore, impacts are related to the system being affected, here the volume of water (m^3). Hence, impact scores can be interpreted as an increase in extinction risk in a system over a certain exposure period. By multiplying these characterization factors (CFs) with the inventory flows (m^3 in the case of water consumption and MJ for thermal pollution), we find the ecosystem impact scores for different impact categories measured in PDF m^3 yr.

The total freshwater biodiversity loss caused by electricity generation is calculated as follows:

$$BL_i = WBL_i + TBL_i \tag{2}$$

Where BL_i is the biodiversity loss caused by electricity generation in province i (PDF m^3 yr); WBL_i the biodiversity loss caused by water consumption for electricity generation in province i (PDF m^3 yr); TBL_i the biodiversity loss caused by thermal pollution from electricity generation in province i (PDF m^3 yr).

Based on the results of provincial biodiversity loss via electricity generation, we examine the biodiversity loss embodied in power transmission, given by:

$$BE_i = \sum_j BE_{ij} = \sum_j (T_{ij} \cdot PBF_i) = \sum_j (T_{ij} \cdot \frac{BL_i}{EG_i}) \tag{3}$$

Where BE_i is the total biodiversity loss embodied in the power transmission from province i to other provinces (PDF m^3 yr); BE_{ij} the biodiversity loss embodied in the power transmission from province i to j (PDF m^3 yr); T_{ij} the power transmission from province i to j (GWh); PBF_i the biodiversity loss per unit of electricity generation in province i (PDF m^3 yr/GWh); and EG_i the total electricity generation in province i (GWh).

The net outsourcing of biodiversity loss can be obtained for each province with:

$$NBE_i = \sum_j (BE_{ji} - BE_{ij}) \tag{4}$$

Where NBE_i is the net outsourcing of biodiversity loss of province i (PDF m^3 yr). If the NBE_i is positive, province i is a beneficiary of power transmission.

4.2.1.3 Decoupling between biodiversity loss and electricity generation

Analyzing the decoupling of environmental impacts from their driving forces can help to identify the trends in impacts for policymakers . A widely used model proposed by Tapio [277] decouples relationships between various environmental impacts and their drivers. Here we use the Tapio model to examine the decoupling between biodiversity loss and electricity generation, with the decoupling degree (θ_t) calculated by:

$$\theta_t = \frac{\Delta BL/BL_{t-1}}{\Delta EG/EG_{t-1}} = \frac{(BL_t - BL_{t-1})/BL_{t-1}}{(EG_t - EG_{t-1})/EG_{t-1}} \tag{5}$$

Where subscript t refers to the target year; ΔBL the change in biodiversity loss during (t-1, t); ΔEG the change in electricity generation during (t-1, t). The decoupling state quadrant map corresponding to the decoupling indicator is shown in Figure S7.3.1.

4.2.1.4 Decomposition analysis of biodiversity loss

To assess the driving factors of environmental impacts we apply LMDI (Logarithmic Mean Divisia Index) decomposition [278]. LMDI has no residuals and is transparent in the interpretation of results [129, 279]. We decompose the driving factors as:

$$BL = \sum_i BL_i = \sum_i \frac{BL_i}{WEG_i} \cdot \frac{WEG_i}{EG_i} \cdot \frac{EG_i}{EG} \cdot EG \tag{6}$$

Where WEG_i is the water-using electricity generation (hydropower and thermal power) in province i (GWh); EG_i the total electricity generation in province i (GWh); EG is the national electricity generation (GWh); BL_i / WEG_i represents the biodiversity loss per unit of electricity generation using freshwater during its operation (hydropower and thermal power) in province i; WEG_i / EG_i represents the proportion of water-using electricity generation in province i; EG_i / EG represents the proportion of the electricity generation of province i in the national electricity generation; EG represents the national electricity generation. Set:

$$BW_i = \frac{BL_i}{WEG_i},\ WE = \frac{WEG_i}{EG_i},\ EE_i = \frac{EG_i}{EG},\ E = EG$$

Eq. (6) can be transformed into:

$$BL = \sum_i BW_i \cdot WE_i \cdot EE_i \cdot E \tag{7}$$

Where the potential driving factors are: 1) BW_i representing the biodiversity loss intensity of electricity generation; 2) WE_i representing the structure of electricity generation; 3) EE_i representing the distribution of electricity generation; and 4) E representing the scale of electricity generation.

The two LMDI approaches, additive decomposition and multiplicative decomposition, can be related to one another using several expressions [280]. In this study, the additive decomposition method is used to analyze the effects of biodiversity loss intensity, electricity generation structure, electricity generation distribution, and electricity generation scale on biodiversity loss during 2008-2017. The total biodiversity loss from the beginning period (base period) to *t*, the final period (report period) can be expressed as:

$$\Delta BL = BL^t - BL^0 = \Delta BL_{BW} + \Delta BL_{WE} + \Delta BL_{EE} + \Delta BL_E \quad (8)$$

Four effects of biodiversity loss changes are modelled: the biodiversity loss intensity effect (ΔBL_{BW}), the electricity generation structure effect (ΔBL_{WE}), the electricity generation distribution effect (ΔBL_{EE}), and the effect of electricity generation scale (ΔBL_E).

The decomposition equations for each effect are shown as follows:

$$\Delta BL_{BW} = \sum_i \frac{BL_i^t - BL_i^0}{\ln BL_i^t - \ln BL_i^0} \cdot \ln\left(\frac{BW_i^t}{BW_i^0}\right) \quad (9)$$

$$\Delta BL_{WE} = \sum_i \frac{BL_i^t - BL_i^0}{\ln BL_i^t - \ln BL_i^0} \cdot \ln\left(\frac{WE_i^t}{WE_i^0}\right) \quad (10)$$

$$\Delta BL_{EE} = \sum_i \frac{BL_i^t - BL_i^0}{\ln BL_i^t - \ln BL_i^0} \cdot \ln\left(\frac{EE_i^t}{EE_i^0}\right) \quad (11)$$

$$\Delta BL_E = \sum_i \frac{BL_i^t - BL_i^0}{\ln BL_i^t - \ln BL_i^0} \cdot \ln\left(\frac{E^t}{E^0}\right) \quad (12)$$

4.2.2 Materials

Power generation: This study includes 31 provincial-level administrative regions (provinces, autonomous regions, and municipalities; for simplicity, they are referred to as provinces and their names are given in Figure S7.3.2). Provincial power generation during 2008-2017 was obtained from China Electric Power Yearbook [219] and China Electricity Council [217]. We focus on hydropower and coal-fired thermal power in this study as the major users of freshwater. Nuclear power is not included as plants in China are along the coastline and use seawater for cooling, which would impact marine environments rather than the freshwater environments we assess here[8, 145]. Coal, hydropower and nuclear power dominate power production, while gas and oil power plants account for less than 5% of the total during the study period, and they are not included due to data limitations [219]. The operational water consumption of wind and photovoltaic power is negligible and thus not considered. Other electricity-generating technologies accounted for less than 7% of the total during the study period and did not discharge freshwater thermal pollution to rivers [225, 267].

Power transmission: Interprovincial power transmission during 2008-2017 was obtained from the China Electricity Council [51]. These data are mostly reported in the form of province-to-province transmission. A small amount of transmission data is from provinces to the subnational grid. We disaggregate them into the province-to-province transmissions based on existing electricity transmission lines [172, 220].

Water: Water consumption factors are obtained from Jin et al. [8], which assessed the provincial factors based on plant-level data in 2017. The national factors in 2008-2016 were reported by China Electricity Council [48]. We assessed the provincial factors by assuming that they changed in proportion to the national factors. The water consumption factors for hydropower are not reported in this data set, so the data from Jin et al. [8] were used. The provincial water availability and water use were obtained from the Ministry of Water Resources [225, 281-284] and used to assess water stress and Characterization factors (Supplementary information S5).

4.3 Results

4.3.1 Electric power system and its water use

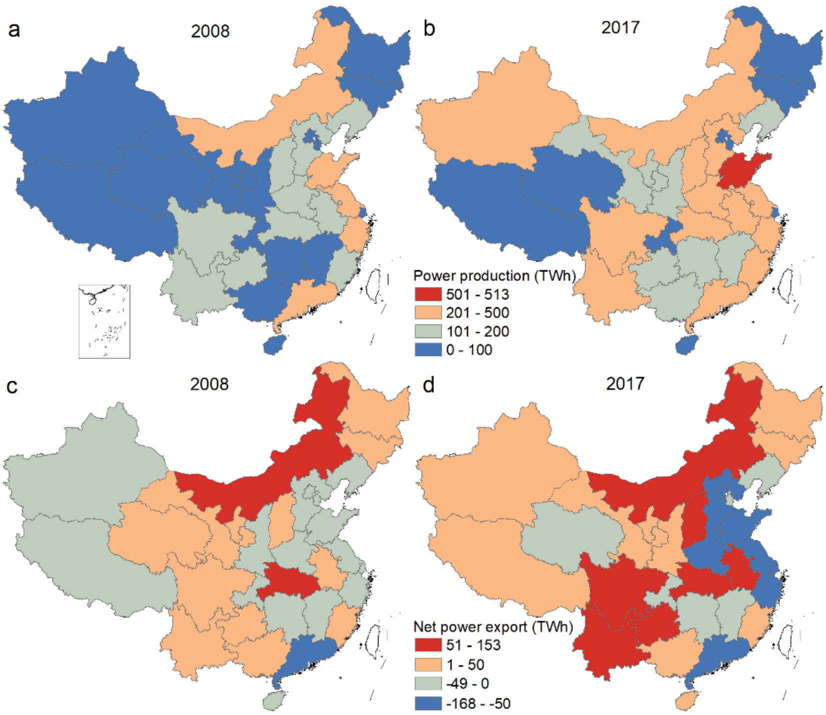


Figure 4.2 Provincial power production (a, b) and net power exports (c, d) in 2008 and 2017.

Nationally, electricity generation almost doubled between 2008 and 2017, from 3451 to 6417 TWh (Figure S7.3.3). The increase was slowest in 2015 (2.4%) and fastest in 2017 (6.5%). Coal power grew continuously and was the largest contributor to the total generation increase throughout the period, but its share in the total electricity generation decreased to 65% by 2017. Wind and solar power developed quickly but still accounted for only 5% and 2% of the total respectively by 2017. All provinces saw an increase in power production during 2008-2017, while many coastal regions experienced an increase in power imports (Figure 2). Shandong province is the largest electricity producer (513 TWh in 2017, 95% of which was from thermal power). Sichuan province is the top hydropower producer (304 TWh in 2017).

During 2008-2017, national freshwater consumption of electricity generation peaked in 2013 (13.6 Gm^3) and declined to 12.4 Gm^3 in 2015 (Figure 4.3). However, freshwater consumption began rising again in 2016 due to hydropower expansions and a stagnation in previous improvements in thermal water intensities. In 2017, total freshwater consumption for electricity generation was 13 Gm^3. Thermal-power water consumption peaked in 2011 (6.5 Gm^3) and then declined to 4.1 Gm^3 in 2017. Water consumption of hydropower increased continuously and reached 8.9 Gm^3 in 2017. Electricity generation accounted for 34% of the total industrial freshwater consumption in 2008, with the proportion rising to 43% in 2017. Hunan province was the largest consumer, with a freshwater consumption of 1.2 Gm^3, whereas Beijing consumed the least (0.02 Gm^3). The freshwater consumption factor of China's electricity generation decreased from 3.2 to 2.0 L/kWh during 2008-2017. Tibet generated electricity with the highest water consumption factor (9.7 L/kWh in 2017), as it relies on hydropower. Shanghai, with once-through cooling systems for thermal power and no hydropower, has the lowest water consumption factor of 0.46 L/kWh in 2017.

The average annual freshwater thermal emission of power production was 2996 and 4771 PJ in 2008 and 2017 respectively. Thermal power accounted for approximately 90% of the total thermal emissions, while the remaining 10% are from hydropower due to its cooling needs. Jiangsu province is the largest emitter of thermal pollution due to its use of once-through cooling systems.

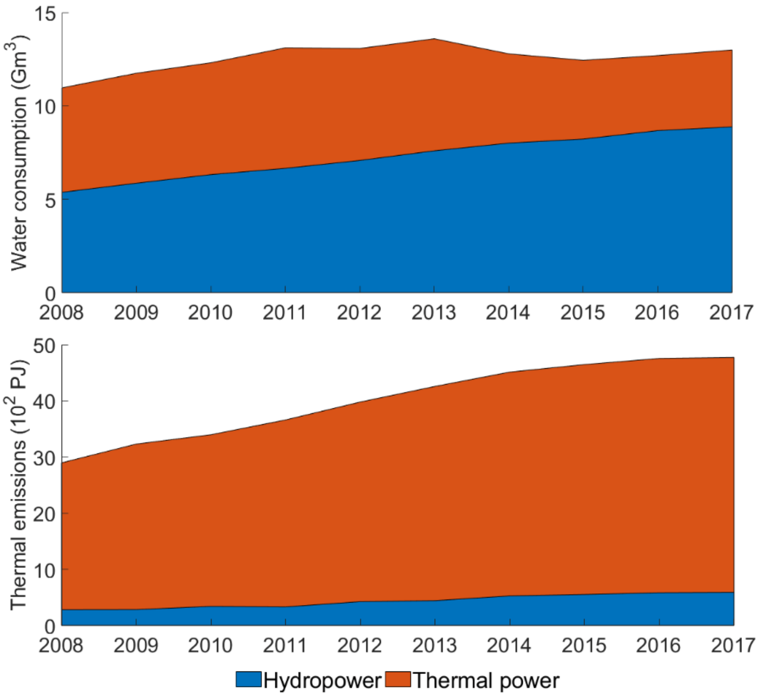


Figure 4.3 The water consumption and thermal emissions of electricity generation in China during 2008-2017.

4.3.2 Biodiversity impacts of electricity generation

The total biodiversity loss by water consumption and thermal pollution of China's electricity generation increased from 1.1×10^8 in 2008 to 1.6×10^8 PDF m^3 yr in 2017 (Figure S7.3.4). Thermal power accounted for 72% and 65% of the total biodiversity loss of power production in 2008 and 2017 respectively. The impact of thermal power peaked in 2013, whereas the impact of hydropower kept increasing during the study period. Despite the increase of the total impact, the biodiversity loss per unit of electricity generation reduced from 3.2×10^{-5} to 2.5×10^{-5} PDF m^3 yr/kWh. Compared to thermal power (2.3×10^{-5} PDF m^3 yr/kWh), hydropower (4.7×10^{-5} PDF m^3 yr/kWh) caused double the biodiversity loss per unit of electricity produced in 2017 because of its higher water consumption. The impact of freshwater thermal emission (1×10^8 PDF m^3 yr in 2017) is 60% larger than that of freshwater consumption (6.2×10^7 PDF m^3 yr in 2017). In China, the south generally faced larger biodiversity impacts than the north (Figure 4.4). Jiangsu, Hunan, Hubei and Anhui provinces alone contributed to 57% of the biodiversity loss of power production in

2017.

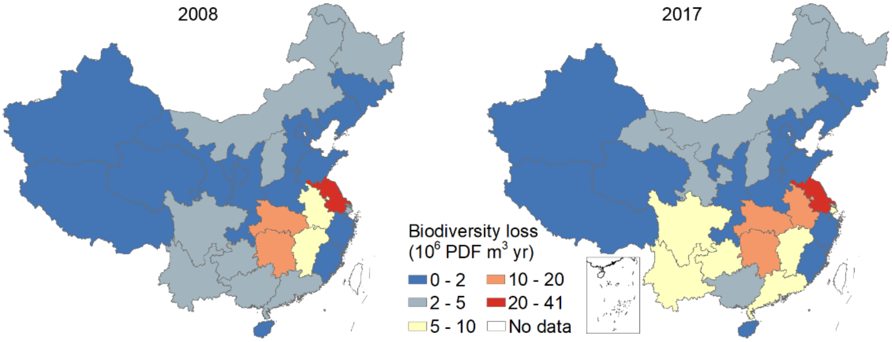


Figure 4.4 The provincial freshwater biodiversity loss caused by electricity generation in 2008 and 2017.

4.3.3 Biodiversity impacts embodied in power transmission

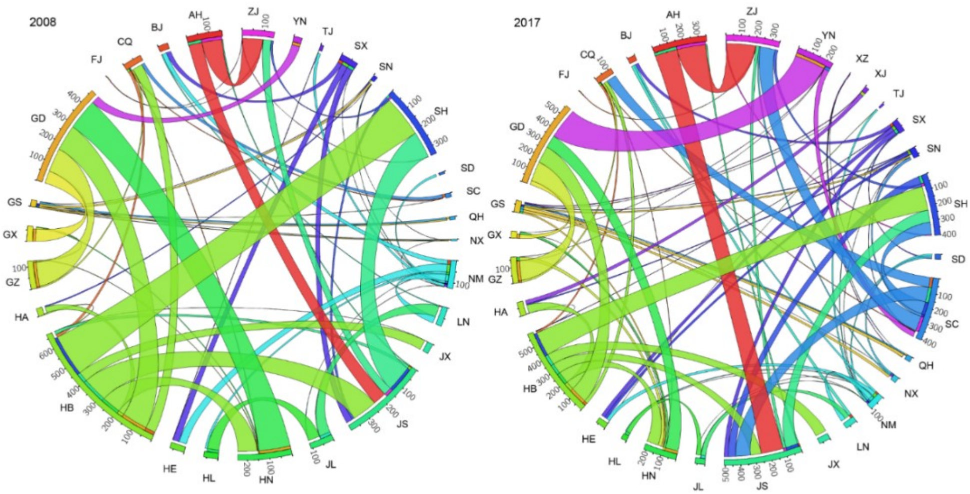


Figure 4.5 The biodiversity loss embodied in interprovincial power transmission in 2008 and 2017. Each color represents an exporting region. Numbers are in the unit of 10^4 PDF m^3 yr. Please see the provinces' full names and abbreviations in Table S7.3.2.

The interprovincial electricity transmission increased more rapidly than electricity generation, from 445 TWh in 2008 to 1130 TWh in 2017 (Figure S7.3.5). The transmission is mainly from the west to the east. Inner Mongolia is the largest electricity exporter (exporting 55 TWh), whereas Guangdong is the largest electricity importer (importing 185 TWh) in 2017. During 2008-2017, embodied thermal pollution via power transmission increased from 12.6 to 17.9 GW, and embodied water increased from 1.5 to 2.0 Gm^3. Across the country, 17 provinces were net water exporters, while 14 provinces were net importers in 2017. There were 15 water-scarce provinces (water stress index larger than 0.5), of which 47% were net water exporters with a contribution of 23% to the total electricity generation.

Power transmission accounted for 15% of total biodiversity loss of power production in 2017. The biodiversity loss embodied in interprovincial power transmission increased by 39% during 2008-2017. Guangdong (GD) province was the largest beneficiary in both 2008 and 2017 by importing a large amount of electricity, with a net import of biodiversity of 4.5×10^6 and 5.4×10^6 PDF m^3 yr, respectively. Hubei (HB) province was the largest net exporter of biodiversity in both 2008 and 2017 (Figure 4.5).

4.3.4 The trends and driving factors of biodiversity impacts

We see an overall decoupling between biodiversity loss and electricity generation during the study period (Table S7.3.3). There was a 45% increase in biodiversity loss and an 88% increase in power production during 2008-2017. During 2011-2013, biodiversity loss and electricity generation experienced an expansive coupling because of the increase in thermal pollution from thermal power. However, their relation turned back into decoupling after 2013 due to the slow increase or even decrease in biodiversity impacts.

During the study period, the increases in biodiversity loss each year from electricity generation slowed (see Figure 4.6). The expansion of electricity generation (the scale parameter in the driving forces) was the main driving factor of the increase of biodiversity loss, whereas the biodiversity loss intensity saw decreases and lowered overall biodiversity loss (Figure 4.6). The impact of electricity generation scale generally decreased from 2011-2015 but began to rise in 2016. The electricity generation structure change, i.e., the decrease of the share of freshwater-using electric power (hydropower and thermal power) in total generation, had a positive but relatively small effect on biodiversity conservation. Although the amount of freshwater-using electric power did not see a decrease, this effect still has increased in recent years due to the increases in wind, solar and nuclear power. In fact, hydropower and thermal power have seen a continual increase since 2011. From the perspective of the cumulative impact, 22 provinces saw an increase in biodiversity loss, whereas 9 saw a decrease. Jiangsu province was the largest contributor to biodiversity loss due to increases in electricity generation, whereas Heilongjiang province was the largest contributor to reducing biodiversity loss due to the decrease in biodiversity loss intensity of electricity generation and proportion in the national electricity generation.

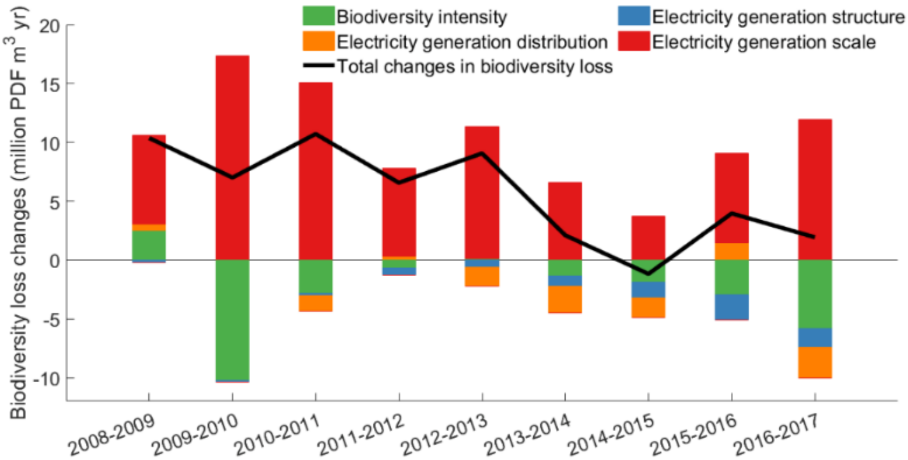


Figure 4.6 Decomposition of the changes in biodiversity loss during 2008-2017.

4.4 Discussion

4.4.1 Energy transition and biodiversity impacts

China's total electricity generation grew continuously over the study period, with a remarkable change in the electricity generation structure towards wind and solar power. Meanwhile, hydropower and thermal power generation also increased by 111% and 63% during 2008-2017, respectively, keeping water consumption high throughout the period. Hydropower is expected to increase [285], indicating the strong possibility of an increase in hydropower-related biodiversity impacts in the future. Recently, China has proposed strict regulations on the water use of thermal power but these have not been formally adopted yet [286, 287]. There has been a program of shutting down small and inefficient thermal power plants while constructing supercritical and ultra-supercritical units, all of which have saved water [288]. Additionally, there are two classes of air cooling: direct air cooling and indirect air cooling [200]. The indirect air-cooling systems, where the condenser system uses water in its cycle but without any evaporation, are increasingly used in water-scarce regions in China [8]. These systems have the advantage of both direct air cooling (low water intensity) and wet cooling (stable cooling efficiency) [289]. Many of the easiest implemented water-saving technologies have already been widely adopted and the potential for further improvements are diminishing (with a reduction in water consumption factor of only 0.02 L/kWh per year during 2017-2019) [23, 48].

Decomposition results show that the structure and distribution of electricity generation had a small overall reducing effect on biodiversity loss, indicating that electricity generation has shifted towards low biodiversity-impact regions and technologies. Electricity transmission has promoted the development of wind, solar, and hydropower in western and northern China. Its continued expansion, along with market developments will enable further optimization of power structure and distribution. However, its impact on biodiversity loss is uncertain and depends on the choices made between water-using and other energy technologies.

4.4.2 Comparison with previous studies

Pfister and Suh assessed the impact of thermal emissions from electric power generation on freshwater ecosystems in the US, finding that less than 5% of values are below 1.0×10^{-5} PDF m^3 yr / kWh and less than 0.1% above 1.0×10^{-3} PDF m^3 yr / kWh [271]. Raptis et al. showed that the thermal emissions impact of China's electricity generation in 2011 was 4.0×10^{7} PDF m^3 yr . Our results showed that the impact was 6.9×10^{7} PDF m^3 yr in 2011 and then increased to 8.6×10^{7} PDF m^3 yr in 2017. The differences between Raptis et al and our results arise mainly from two sources: the lower coverage of thermal power and the lower capacity factors in Raptis et al [61] which are based on data from the U.S. Energy Information Administration. In addition to thermal pollution, water consumption is another major cause of biodiversity impact. We extended previous thermal-power studies to include the water consumption of both hydropower and thermal power. Results show that the impact of freshwater consumption was smaller than thermal emissions. Previous studies have not quantitatively analyzed the temporal changes and driving factors of biodiversity impacts. Our analysis indicated an overall relative decoupling between electricity generation and biodiversity impacts. The expansion of electricity generation scale and the decrease in biodiversity loss intensity of electricity generation were identified as the major driver and preventer of biodiversity loss, respectively.

This study focused on China; in the future, it will be important to make assessments for other nations or on a global scale. While local, regional, and global species losses are relevant, only global losses cannot be recovered. Unfortunately, local or regional relative species loss cannot be easily aggregated or compared on a global level or against other estimates for several reasons. First, the same relative species loss can imply very different absolute species losses in different regions. Second, some regions host more endemic species than others. It is more likely that regional losses in those regions lead to global extinctions than in regions associated with fewer endemic species. We used conversion factors to convert regional species richness impacts into potential global species extinctions [290]. Our results showed that the global impacts increased from 1.0×10^{-4} to 1.6×10^{-4} PDF yr during 2008-2017 (Supplementary information S6). The biodiversity impacts were expressed as the potentially disappeared fraction of species (PDF) caused by water use (freshwater consumption and thermal emissions). In the future, assessments should be conducted for a broader range of impact and sector categories than just water use of electricity generation as done in this study, which will allow a better understanding of anthropogenic impacts on biodiversity.

4.4.3 Limitations and implications

In this study, we focused on operational water use rather than lifecycle water use. The fuel cycle and plant infrastructure may require large amounts of freshwater, depending on the fuel type [16]. Further work could focus on the biodiversity impacts of lifecycle water use in the future when data are available, i.e., the location and way of fuel mining activities, the materials of plant infrastructure and their sources. For thermal power plants, the use of carbon capture and storage (CCS) in the future to meet climate targets will pose a threat to water-related biodiversity, as it heavily relies on water resources [291, 292]. However, the potential impact of CCS was not considered in this study because of the lack of information on the location and scale of CCS deployment in the future. In addition, this study focused on the water-related biodiversity impacts, but the biodiversity loss of other pressures from electricity generation are not included. For example, the land occupation by solar power and windfarms[293, 294] and the freshwater habitat fragmentation[295] and flow alterations[24] caused by hydropower dams have impacts on biodiversity. An impact assessment of habitat fragmentation would require the development of new characterization factors. The species-discharge relationship used to assess impacts from freshwater consumption does not consider impacts from flow alterations, of which also increased discharges can have adverse impacts on freshwater biodiversity. Such flow alterations have so far only been considered within water stress footprints [24], but no characterization factors exist yet that extend the cause-effect chain to biodiversity impacts. Freshwater biodiversity is complex, and the species richness pattern of one taxon is unlikely to be a good indicator of the pattern of another taxon [296]. While we considered four species groups for the impacts of thermal pollution, we focused only on fishes for the impacts of freshwater consumption. Future studies could expand the taxonomic coverage for freshwater consumption impact assessment when related data and models become available. Additionally, the species-discharge relationship would benefit from regionalization[297] to account for factors such as different climatic conditions. The thermal pollution impacts on river temperature and biodiversity may differ across different types of outfalls of power plants [272]. It will be of interest to distinguish outfall types when data become available. There are approximately 47,000 hydropower plants in China [298], of which we only cover about half. This results in uncertainties, as water use differs a lot across hydropower plants.

According to these results, we make several suggestions for mitigating the impacts of the electric power system on freshwater biodiversity. First, it is important to reduce the water use of hydropower and thermal power, as they dominate the current energy system. Our results showed that the water consumption of hydropower has large impacts on biodiversity and is expected to increase in the future, indicating the necessity to build run-of-river hydropower plants (a type of hydroelectric generation plant that has little or no water storage and reservoir evaporation). For thermal power, adopting air cooling systems and using seawater and reclaimed water for cooling are feasible and effective ways of reducing freshwater demand. Air cooling systems are commonly used by newly built plants. Indeed, 29% of operating plants now use this technology, indicating the potential for further reducing water requirements if this proportion was to increase. Seawater use in coastal regions (such as Jiangsu, Shanghai, and Guangdong) is encouraged by the government [8]. We show that 15 billion m^3 of freshwater can be saved by switching to seawater cooling for power plants near the coast (within 10 km). However, the economic costs of retrofitting cooling systems and building seawater treatment facilities need more research. Since hydropower is a renewable resource that can enable greater amounts of other renewables in the electricity system (via the provision of grid stability functions and load matching renewable variations), the net result of associated climate-change driven biodiversity loss through lower hydropower capacity and the freshwater biodiversity loss of hydropower water use is not straightforward. Second, the further development of renewables such as photovoltaics and wind power is crucial since both consume a negligible amount of water. Under the International Energy Agency's (IEA's) sustainable development scenario, China's wind and solar PV will experience a rapid increase by 2439 TWh through the period 2017-2030, equal to 42% of the total hydro and thermal power production in 2017. This suggests a significant opportunity in switching to a low water-intensity power system[285]. Third, we can shift electricity generation from regions with high biodiversity intensities to those with low biodiversity intensities by considering the provincial biodiversity factors of electricity generation assessed in this study.

Chapter 5

Climate change and CCS increase the water vulnerability of China's thermoelectric power fleet

This chapter has been published as: Jin, Y., Scherer, L., Sutanudjaja, E.H., Tukker, A., Behrens, P. Climate change and CCS increase the water vulnerability of China's thermoelectric power fleet. *Energy*, 2022,245,123339.

Abstract: Large numbers of China's thermal power plants are in water-stressed regions. Changes in the availability of water resources due to climate change may impact the vulnerability of regional and national electricity generation. Here we explore this vulnerability for coal-fired power units (CPUs) as the generation-type most exposed to climate risk. We find many plants are already experiencing water scarcity and 120-176 GW of capacity will be exposed to water scarcity for at least one additional month per year in the 2030s. In the absence of carbon capture and storage (CCS) the national usable capacity of CPUs will increase slightly, mainly due to an increase in water availability for power plants in northern China under all climate scenarios except RCP8.5. However, CCS systems have been identified as essential in China's national roadmap for carbon neutrality and their use represents high water requirements. The addition of CCS significantly exacerbates water vulnerability, leading to further usable-capacity reductions of 7.4-7.7%. We assess several adaptations and find that early retirement of power plants is most effective, with interregional power transmission also playing an important mitigating role. Our work highlights the need for improved awareness of water resources in electricity planning.

5.1 Introduction

Despite efforts to increase renewable energy and reduce coal power, 67% of global electric power in 2018 was produced by thermal power with 38% from coal-fired power plants[37]. Global electricity demand is expected to increase with a growing world population and, more significantly, with increasing consumption levels[19, 20, 26]. Water is an essential requirement for operating the global power plant fleet and has knock-on implications for energy security[5, 16, 24, 25]. However, climate change and water shortages have increased the sensitivity of power production to water availability[189], raising both research and policy concerns[299]. During 2011-2015, 43% of the global coal-fired power plant capacity experienced water scarcity for at least one month per year and 32% experienced scarcity for five or more months per year[292]. Severe water shortages can result in power curtailments and reduce the reliability of the electrical power system[53, 300].

China produced 26% of the total global electric power in 2018[37], with thermal power as the main contributor (accounting for 72% nationally[264]). In 2007, thermal power was responsible for roughly 10% of the total national freshwater withdrawal[23, 39, 168,

[201]. This proportion is relatively low compared to other regions, such as the US (45%)[238] and Europe (43%)[239] for the same decade. However, there is a severe geographical mismatch between water resources and thermal-power plant locations across China[42], as many thermal power plants are located in water-stressed regions. Research has focused on the water use of thermal power production[23, 168, 202], but few have connected plants' water use to water availability to assess vulnerability under climate change. Zheng et al.[301] made a step forward by identifying regions where power production is vulnerable to water scarcity, but did not capture finer-scale spatial-temporal variations in water availability, potential usable-capacity reductions of power plants, or the impact of power transmission.

Power plants face reductions in usable capacity if the required water withdrawal – the volume of water diverted from a water source for use – cannot be met[26]. Research on the vulnerability of thermal power to changes in water resources for the US[300, 302] and Europe[26, 241] indicate reductions in usable capacity and power supply shortages under future climate change. Previous studies simulated the available water resources for thermal power, but changes in water use for other sectors (e.g. irrigation) are not often incorporated. These other sectors often result in additional constraints for the electricity sector[53]. While previous work has been conducted on the level of the river basin[53], water scarcity is at the plant level and a higher spatial resolution is needed for localized assessments of water scarcity and its impacts on power production. This is important in China's case, since the power sector sees heavy water competition with other users. For example 84% of China's CPUs being close to residential areas and farmland (Figure S7.4.1).

A plant-level analysis is essential to a vulnerability analysis since individual plants can be significant withdrawers and consumers of water within a region. This requires knowledge of the cooling type for each unit within a plant (since the cooling type is a strong determinant of water use[13, 16]). There are three common cooling types: once-through cooling, closed-loop (wet tower) cooling, and air cooling[200]. A further complication is that there are four main types of cooling water used in China: surface water, groundwater, reclaimed water, and seawater. Distinguishing these different cooling and water types can be a challenge due to data availability, but they are important if we are to gain a deeper understanding of the vulnerability of power production to water scarcity.

We can expect many policy and technological responses to water constraints in the power system, so it is also important to assess adaptation and mitigation strategies. Van Vliet et al.[26] considered strategies for power plants globally but did not include the role of the power transmission network. Interprovincial power transmission plays an increasingly important role in China's energy system (increasing 220% between 2008 and 2019[48]). Increased power transmission facilitates the shifting of generation away from highly water-scarce regions.

Climate change mitigation (e.g. carbon emission reduction) can also have direct and large impacts on water scarcity issues. Zhang et al. indicated that there are conflicts between water conservation and carbon emission reduction of China's thermal power[303]. Tang et al. showed that peaking China's power sector carbon emissions before 2030 may increase the water consumption due to the expansion of nuclear power according to their simulation results[304]. While renewables have much lower water requirements, urgent emission-reduction requirements, political trade-offs, and existing infrastructure mean that China's energy transition strategy utilizes large amounts of carbon capture and storage (CCS) during the 2030s[45, 46, 305]. Many proposed scenarios for meeting Net-Zero carbon by 2060 require significant amounts of CCS, with one model proposing 850 GW coal, gas, and biofuels be retro-fitted with CCS[45]. While CCS is regularly promoted for thermal power plants (which emitted 4.2 $GtCO_2$ in 2019, comprising 41% of China's emissions)[45, 47, 48] and there are some demonstration stage projects [306] CCS will require additional water resources[292]. Reliance on CCS may place significant additional stress on water-scarce regions. There are other CCS approaches with lower water requirements, such as oxyfuel and pre-combustion, but it is generally thought that post-combustion capture technology will be the most common by far, given its ease of implementation and technological maturity[109, 307]. For this reason, we focus on post-combustion technology here.

To address these issues, we developed a hydrological-electricity modelling framework. This framework examines the vulnerability of power production to climate change and water scarcity at a monthly time step and a 5-arcmin spatial resolution of the river network. This contrasts with existing macro-scale studies that typically use a 0.5°-resolution[26, 53, 241]. We include individual water uses of power plants, four water types, electricity-specific water availability and the national transmission grid. We use two indicators to measure the impacts of future water availability on power production: the number of months that CPUs face water scarcity and the usable capacity reduction. The former reflects the time span of impacts and the latter reflects the severity of impacts.

We also tested 5 adaptation options to mitigate power system vulnerabilities that may be exacerbated by CCS. An adaptation is considered effective if the usable capacity increases after its implementation and include: (1) Switching to seawater cooling for all CPUs close to the coast (within 10 km and already encouraged for these geographical areas in national policies)[53, 237]; (2) Replacement of once-through cooling systems with closed-loop systems that decrease water withdrawals; (3) Increasing all power plants' water use efficiency to the same level as today's state-of-the-art units[286]; (4) Improving power transmission between regions of low and high water stress, allowing for closure of generation in water-stressed regions and new generation in regions of lower water stress[53, 308, 309]; and, (5) Closing coal units after 30 years, rather than 40 [304, 307, 310] due to additional energy transition policy pressures (we assume no new CPUs will be built to compensate and that the gap will be filled by low-water intensity renewables).

This study makes several contributions: First, we built and solved an electricity-hydrology model at the individual plant level and a monthly time scale and assess the vulnerability of power production to water scarcity in China under climate change. This provides a template for similar analyses in other nations (previous assessments focused on the USA [308] and the European Union [53]). Second, CCS may be used for power plants but its impacts on power production have not been examined (Zhu et al. investigated overall water use by CCS but did not assess the influence on electricity generation [311]). Here, we quantified the impacts of CCS on the vulnerability of plants. Finally, we quantitatively evaluated the efficacy of several flexible adaptation strategies whereas previous studies only qualitatively analyzed them[301, 312].

5.2 Materials and Methods

The overall modelling approach is shown in Figure 5.1 and we present detailed steps in Sections 5.2.1-5.2.5. Impacts of water scarcity on thermoelectric usable capacity were quantified for the 2030s under four different climate scenarios (Representative

Concentration Pathway (RCP) 2.6, 4.5, 6.0, and 8.5) relative to a reference period 1992-2001.

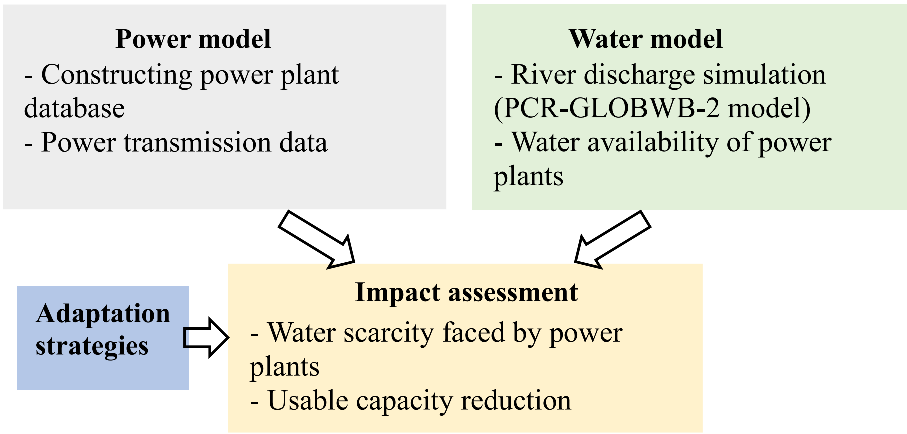


Figure 5.1 Model framework used in this study. The different colors indicate the different models used in the framework.

5.2.1 Power model

We compiled a database of coal-fired power units (CPUs) including plant name, installed capacity, the beginning year of operation, unit type, location, operation status, and cooling system. Data were sourced from the Global Coal Plant Tracker[38], World Electric Power Plants Database[215], and the China Electricity Council[217]. Coal dominates China's thermal power production with gas power plants accounting for less than 5% of the total (Oil power is not included in this assessment due to its very small contribution at only 0.05% of production). In total, 3050 power production units were included (accounting for 98% of the national total installed CPU capacity in 2017). To verify CPU cooling systems, we used Google satellite imagery cross-checked with information from the China Electricity Council[217]. We obtained the water type for cooling from the China Electricity Council[217] and the Power Industry Statistical Information System[222]. This study focuses on plants using surface water rather than groundwater, seawater and reclaimed water, thus 2265 units were investigated (in total 749.8 GW, 75.2% of the total capacity of CPUs). We used China-specific water use factors for power plants (specific water use for 95% of CPUs were obtained from the China Electricity Council[217] and 5% from previous research[16]). Once-through cooling water withdrawals were obtained from Zhang et al.[23], who used the monitoring data of withdrawals for some plants with once-through cooling systems in the Yangtze River basin.

To examine adaptation options using the power transmission network we compiled an inventory of inter-provincial power transmission for 2008-2017 with data from the China Electricity Council[51]. These data are mostly reported in the form of province-to-province transmission. Additionally, there are some data covering transmission from provinces to the subnational grid. We disaggregate these data into the province-to-province transmissions based on actual electricity transmission lines[8, 220]. China's provinces and river basins are shown in Figure S7.4.3.

5.2.2 Water model

Monthly available surface water (WA) at a spatial resolution of 5-arcmin was calculated as the difference between monthly river discharge and the environmental flow requirement. Monthly river discharge was simulated using the PCR-GLOBWB-2 model[313]. For current conditions, we use the PCR-GLOBWB-2 run based on the European Union Water and Global Change (EUWATCH) data where the actual meteorological observation datasets are used. For future conditions we use PCR-GLOBWB-2 runs based on the data from five different global climate models (GCMs) forced with the four representative concentration pathways (RCPs)[314]. The five GCMs are MIROC-ESM-CHEM, IPSLCM5A-LR, HadGEM2-ES, NorESM1-M, and GFDL-ESM2M and their ensemble means are then applied for the final water availability. We obtained the bias-corrected future conditions (based on EUWATCH and GCM runs). The correction procedure is given by:

$$future\_corrected = present\_watch + (future\_gcm - present\_gcm) \qquad (1)$$

Where the *present_watch* represents present-day values based on the EUWATCH run; *present_gcm* and *future_gcm* represent values obtained from GCM runs in historical (under present-day greenhouse gas concentration forcing) and future (for various RCP scenarios) simulation periods.

Environmental flow is defined as the minimum freshwater flow required to sustain ecosystem functions[292]. For the rivers that supply water for human use in China, 60% of the average discharge needs to be preserved for environmental flow[315]. Environmental flow requirements are the most important factor that influences water availability for power production. Rose et al. showed little sensitivity of water scarcity to different environmental flow requirements[292]. Here, we tested the sensitivity of usable capacity changes and adaptations to environmental flow requirements. Upstream water consumption and reduced availability for downstream uses were accounted for by considering all water uses (irrigation, livestock, households, and industry). The water use for thermoelectric cooling of power plants is not included in PCR-GLOBWB-2[313]. Water consumption was assessed by multiplying the withdrawal and the corresponding China-specific factors (sector-specific consumption-to-withdrawal ratios [225, 316]). Factors for agricultural, industrial and domestic sectors are 0.65, 0.23 and 0.40, respectively. Ratios of surface water consumption to total water consumption were obtained using provincial data. We further assessed surface water consumption by multiplying consumption and the above ratios. The proportion was obtained from the Ministry of Water Resources at the provincial level[225]. We made assessments for RCP2.6, 4.5, 6.0 and 8.5 climate scenarios, capturing the largest range of uncertainties in the future greenhouse gas concentration scenarios. RCP2.6 describes a world in which global warming is kept well below 2 °C by 2100 relative to pre-industrial temperatures. RCP8.5 depicts a future that excludes any climate mitigation policies, leading to nearly 5 °C of warming by the end of the century. RCP8.5 should be considered as an unlikely worst case[317].

5.2.3 Impact of water availability changes on power production

The monthly water scarcity (WS) for each grid cell was assessed using the monthly availability and consumption of surface water resources. We extracted river discharge for each grid cell in which each power plant is situated. For cases in which the power plant and river are not in the same grid cell, the river discharge of the grid cell where the river is located is used. In this way, the available river discharge of 566 units (25% of the total) is corrected. CPUs are located in water-scarce areas if the ratio between water consumption (WC) and available water (WA) is > 1 (after the removal of environmental flow requirements and for renewable water availability only)[318].

$$WS = \frac{WC}{WA} > 1 \tag{2}$$

Koch and Vögele[319] and Wang et al.[320] built models to assess the thermoelectric power usable capacity reduction caused by water scarcity. Since these studies do not consider competition for water between the electricity sector and other sectors, we further modified these models. Additionally, the water withdrawals of CPUs in these studies were calculated based on cooling water temperature regulations and power-plant-specific characteristics. There is no regulation on cooling water temperature in China so we did not estimate the withdrawal based on temperature restrictions but used the unit-specific withdrawal data obtained from the sources mentioned above. The equations for estimating the usable capacity reduction are:

$$q = KW \cdot t \cdot WW \quad (3)$$

$$P = \min\,(Q - NEW, q) \cdot \frac{1}{t \cdot WW} \quad (4)$$

Where q = monthly required water withdrawal (m^3); KW = installed capacity of CPU (MW); t = The number of hours in each month (h); WW = water withdrawal factor (m^3/MWh); P = usable capacity of CPUs (MW); Q = monthly river discharge (m^3); NEW = water consumption of non-electricity sectors (m^3).

5.2.4 Assessing the impact of CCS on the vulnerability of thermal power

The use of large amounts of CCS to meet climate targets represents a significant potential threat to water scarcity. Here we assume a CO_2 capture efficiency of 90% based on previous work[307, 310, 311, 321, 322]. Considering that small (≤100 MW) CPUs will probably be shut down before being retrofitted with expensive CCS technologies, we assume that only large (>100 MW) CPUs will be retrofitted (in total 733 GW). Although 100% adoption of CCS is unlikely, this assumption allows us to assess the impacts of CCS retrofit on water future vulnerability. This assumption is in line with the urgent need for rapid carbon reductions to meet climate targets[292]. We assess the above adaptation options for both the non-CCS and CCS scenarios. The water requirements of power production with CCS are obtained from Jin et al. 2019[16]. We assessed the cost of CCS-related usable capacity reductions by assuming that the reduced capacity needs to be compensated by building new capacity. The cost is assessed as:

$$CC = CR \cdot IC \quad (5)$$

Where CC = the cost of new capacity (US$); CR = total usable capacity reduction (kW); IC = the investment cost of CPU (US$/kW). The investment cost of China's CPU is US$617/kW [310].

5.2.5 Adaptation options

We tested five adaptation option. Four options focus on thermal power plants due to their large water withdrawal, while one is based on the transmission:
(1) Switching to seawater cooling for all CPUs close to the coast (within 10 km)[53].

(2) Replacement of once-through cooling systems with closed-loop systems (which have lower surface water withdrawal requirements for all power plants).

(3) Increasing power plant water use efficiency to the same level as today's state-of-the-art units (where the state-of-the-art is defined as the average of the 10% most water-efficient plants per MWh generated for each cooling type)[286].

(4) Improving power transmission between low-vulnerability and high-vulnerability regions.[53, 308, 309]. We assume the plants facing water scarcity in vulnerable regions (those experience significant reductions of >2 GW under all scenarios) will be closed, and the generation displaced to regions with low water scarcity. This reallocation is made in proportion to the transmission capacity between those regions based on 2017 data (the latest available year). Within each low-vulnerability region, generation will be allocated to power plants in proportion to, but not exceeding, the plant capacity.

(5) In the absence of early retirement, 86% of current CPUs will be in operation in the 2030s with an average operation time of 40 years. We assume that plant lifetimes will decrease to 30 years due to additional policy pressures[53, 323]. Older plants with higher water intensity and lower energy efficiency are retired earlier [308]. Under the International Energy Agency's (IEA's) sustainable development scenario, China's wind and solar PV will experience a rapid increase by 4600 TWh during 2019-2040, equal to the total coal power production in 2019[285]. This indicates a possibility fill the power gaps in an early retirement scenario where 58% of current freshwater-using CPUs retire. Energy storage technologies work well with variable renewables and there is a growing trend of pairing battery storage with solar PV and wind. Energy storage typically has little to no water requirements except for pumped hydro and hydrogen [324]. However, it is expected that growing battery capacities will provide most storage requirements [285].

To examine the efficacy of adaptation options when CCS is implemented, the usable capacity changes in the 2030s relative to the reference period are calculated for six scenarios separately: baseline (i.e. without adaptation options) and the five adaptation options. We use the expression:

$C = (P_{2030} - P_r)/P_r$ (6)

Where P_r = usable capacity in the reference period (MW); P_{2030} = usable capacity in 2030 (MW) for each scenario; C = usable capacity change. If C of an adaptation option is larger than that of the baseline, the adaptation option is considered effective; if C of an adaptation option is larger than not only that of the baseline but also 0, the option is effective enough to offset the impacts of CCS and water scarcity under climate change.

5.3 Results

5.3.1 The impact of water availability changes on thermal power

Nationally, annual river discharge increases in the 2030s relative to the reference period (1992-2001) for all scenarios. The Yellow and Yangtze river basins feed 23% and 22% of CPUs respectively and also see increases in annual river discharge under all climate scenarios (Figure S7.4.2). The Southeast basin experiences significant decreases in river discharge but feeds only 1% of total CPUs. However, existing policies already account for some heterogeneity in water availability[325]. Nationally, once-through cooling, air cooling, and closed-loop cooling account for 14%, 29%, and 57% of the total CPU capacity, respectively. Closed-loop cooling systems are used throughout the country. Once-through cooling systems are mainly located along the Yangtze River (due to the need for large water withdrawals). By contrast, air cooling systems are mainly in the north, especially in Continental and Yellow river basins (Figures S7.4.3 and S7.4.4).

Our results show that 40% of CPUs experience water scarcity for at least one month and 22% experience severe water scarcity (six or more months) in the reference period (Figure 2). These historical difficulties are often underreported in media and industry. However, a 2012 *Greenpeace* report highlights some instances where China's CPUs are facing water shortage risks[326]. Of the CPUs starting operations before 2000, 36% face water scarcity, and of the CPUs starting operations after 2000, 41% face water scarcity. This suggests a significant mismatch between water availability and demand for the recently built plants. For most rivers in China, water flow is higher in summer than in winter[327]. Accordingly, February is the most acute month for water scarcity overall with 32% of CPUs facing water availability issues, while September sees only 9%. Large amounts of generation see severe water scarcity across Inner Mongolia and Shandong provinces (22 and 20 GW, respectively). On net, CPU capacity experiencing water scarcity will increase by 43-82 GW in the 2030s, relative to the reference period (ranges indicate the minimum and maximum combinations of scarcity and capacity availability). This net result shows 43-56 GW of capacity faceing water scarcity for at least one fewer month a year, and 120-176 GW of capacity exposed to water scarcity for at least one additional month a year.

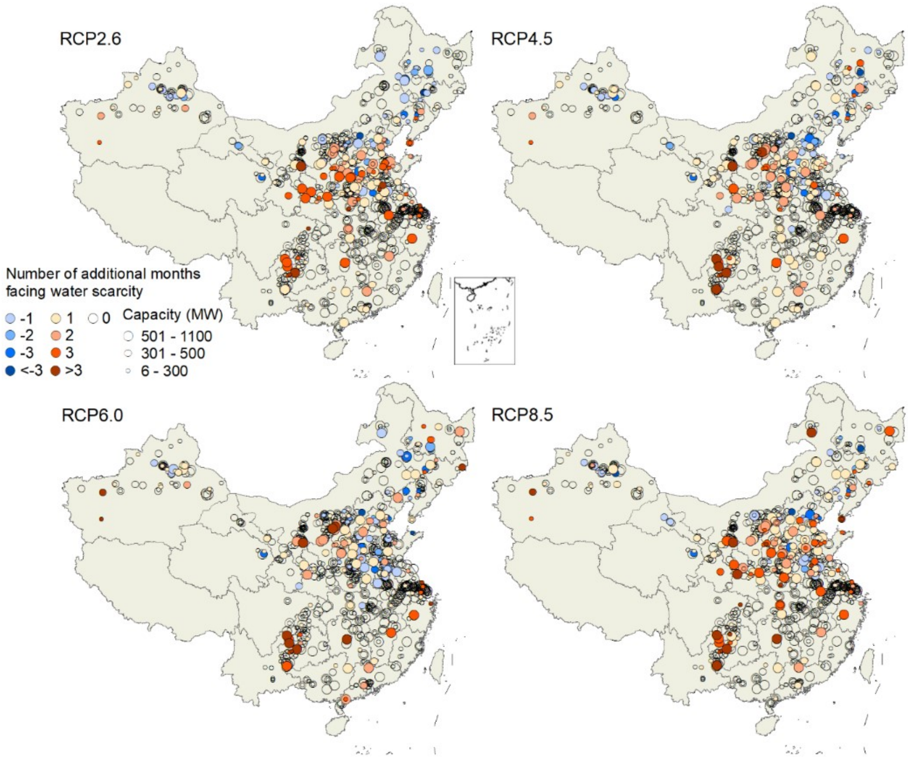


Figure 5.2 Additional water scarcity faced by CPUs. The number of additional months per year when coal-fired units face water scarcity in the 2030s compared to the reference period. Negative values refer to the number of fewer months of water scarcity per year that CPUs face.

Nationally, CPUs experience an increase in the usable capacity of 0.3-1.4% in the 2030s relative to the reference period for RCP2.6, RCP4.5, RCP6.0, whereas RCP8.5 sees a decrease of 1.0%. The impact of water availability changes on usable capacity differs across plants and provinces (Figure 5.3). We show that 22-34% of CPUs face usable capacity reductions and 3-6% face severe reductions (>30%). Between 30 and 38% of CPUs face usable capacity increases. Xinjiang, Hebei and Inner Mongolia provinces experience remarkable usable capacity increases (>0.5 GW), while Guizhou and Shaanxi experience remarkable reductions (>0.5 GW) in all scenarios (Table S7.4.1). There are also seasonal variations in usable capacity reductions (Figures S7.4.5- S7.4.8). Summer sees usable capacity reductions by 0-6.4 GW in all scenarios. Winter sees an increase of usable capacity by 6.5-12 GW in RCP2.6, 4.5 and 6.0, and a decrease by 2.5 GW in RCP8.5.

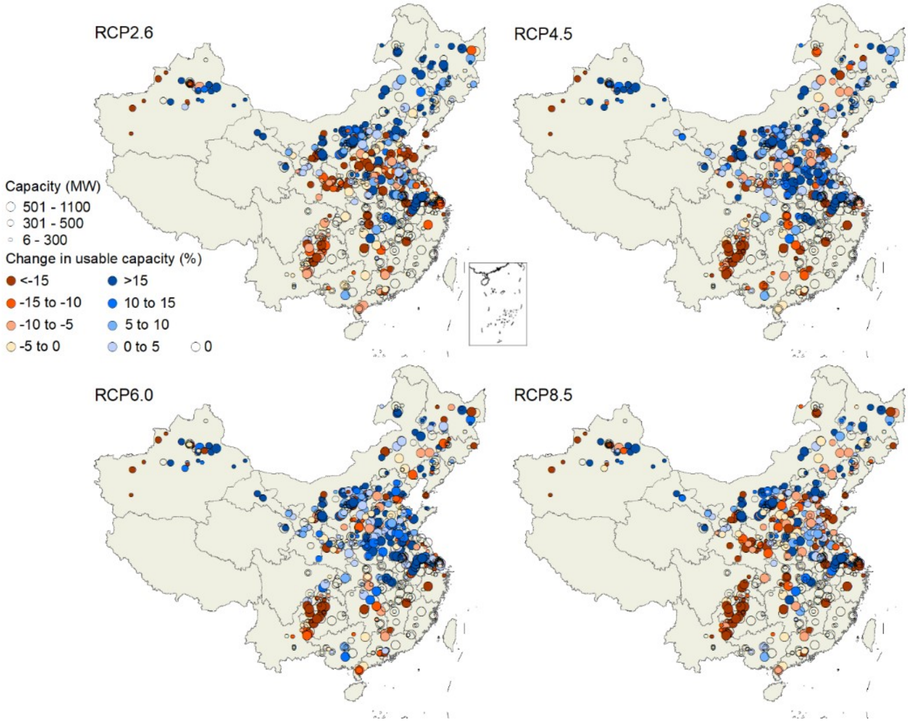


Figure 5.3 Impacts of climate and water resources change on annual usable capacity of CPUs. The changes in the annual usable capacity under four climate scenarios in the 2030s compared to the reference period.

5.3.2 The impact of CCS and adaptation options

The water requirement of power production with CCS can be 53-77% higher, depending on CPU configuration [16]. We find that requirements can be as high as 14.8 billion m^3 per $GtCO_2$ sequestered. Given these water requirements, the addition of CCS increases vulnerability to water stress broadly across all plants. Our results show that adding CCS to plants leads to additional usable capacity reductions of 7.4-7.7%. Between 49 to 55% of CPUs face usable capacity reductions and 15-21% of CPUs face severe reductions (>30%) (Figure S7.4.9). All provinces experience CCS-related capacity reductions except Xinjiang, Qinghai, and Beijing. Several provinces including Guizhou, Henan, Shaanxi and Jiangsu experience significant reductions (>2 GW) under all scenarios (Table S7.4.2).

Given these CCS-related reductions, out of all adaptation strategies only early retirement can increase usable capacity across all RCP scenarios (by 2.8-4.5%, Figure 5.4). Retrofitting the existing plants for seawater use can only slightly mitigate the vulnerability. Cooling type switches and increased water use efficiency may help but are not sufficient to increase capacity compared to the reference period (when CCS is implemented). Increases in transmission can effectively mitigate the CCS-related vulnerability. Water availability is the largest factor affecting our findings. Here, the environmental flow, defined as the minimum freshwater flow required to sustain ecosystem functions, is the most important factor that influences water availability for human purposes[292]. We find little sensitivity to the changes in environmental flow requirements for all adaptation strategies (within 2% of variations) except for early retirement. When the level of protection for ecosystems is high, e.g., 80% of discharge, the efficacy of retirement is extremely high.

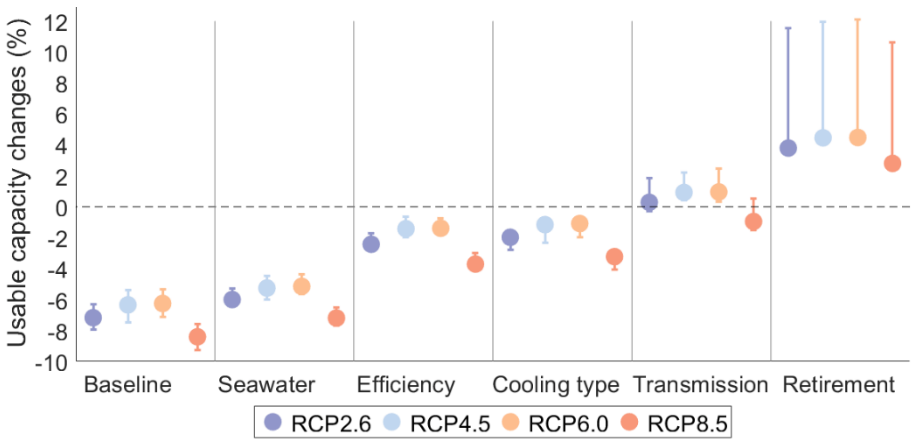


Figure 5.4 Impacts of adaptation on CCS-related CPU vulnerability to water constraints. Usable capacity changes in the 2030s relative to the reference period for the baseline settings (i.e., without adaptation options) and various adaptation options are shown as markers per climate scenario. Ranges indicate the sensitivity of usable capacity to the changes in environmental flow requirements (40-80% of river discharge for environmental flows).

5.4 Discussion

5.4.1 Comparisons with previous studies

On an aggregated, national level, we show that 40% of installed capacity experiences water scarcity for at least one month during the reference period (1992-2001). For comparison, Rosa et al.[292] found that 47% of installed capacity experience water scarcity in China during a later period (2011-2015). Water availability of power production can significantly influence vulnerability. Rosa et al assumed 80% of the monthly river discharge for environmental flow which may be high. According to Han et al.[315], an 80% proportion is recommended for protected rivers, reservoirs, and national parks, whereas 60% is considered sufficient for rivers that supply water for human use (and is the value used here). In terms of usable capacity reductions, van Vliet et al.[26] show reductions in usable capacity for 81-86% of the thermoelectric power plants worldwide in 2040-2069 relative to 1971-2000, with reductions in Asia lower than the world's average. Our results show that 22-34% of China's CPUs will face usable capacity reductions in the 2030s relative to the reference period, depending on future climate changes.

5.4.2 Challenges of mitigating vulnerability

Policymakers are becoming increasingly aware of water supply issues and, in some cases, have implemented water-saving regulations. For instance, there are now restrictions for adding new capacity to the Jing-Jin-Ji area (Beijing, Tianjin, and Hebei). We show that while this key area should be of focus, policies could be expanded to surrounding areas (e.g., Shaanxi, Shanxi, Shandong, and Guizhou). The most effective adaptation, early retirement, faces challenges. If plant lifetime is limited to 30 years, 58% of current CPUs will retire in 2040, resulting in a large power generation gap that would have to be rapidly met with alternatives. However, this assumption of early retirement is less radical than other scenarios. Under the IEA's sustainable development scenario, Chinese coal power production would decrease by 69% in 2040 relative to 2019[285]. In another, faster phaseout, Cui et al.[328] proposed a scenario whereby conventional coal-fired power plants without CCS decline by more than 90% in 2040. If the power gap caused by the shutdown of thermal power plants is filled by water-intensive energy technologies rather than renewables (e.g., wind and solar power, which generally consume orders of magnitude less water than thermoelectric generation), the effect of early retirement on water resources will be lower than expected. Early retirement also has economic and social issues, i.e., the impacts on profitability and employment of coal-fired power plants and coal mining[328]. Cooling technology is also a policy concern, with the share of air cooling systems increasing quickly since it became a government requirement for water-scarce regions in 2004[231]. Air cooling is effective in water-saving but does require higher investment and has a lower energy efficiency[200]. Using seawater is also a useful adaptation strategy and China's power sector is already the largest seawater user, accounting for more than 90% of the national total volume of seawater utilization[23]. However, the price of desalinated seawater is still higher than freshwater[329]. Although constructing coastal power plants can save freshwater, there is a challenge for coal resources far from the coast. Long-distance coal transport from inland to coastal regions is also energy- and water-consuming, to an extent that is not fully understood[330]. The trade-offs between water use and other environmental and economic issues need to be weighed before plant and cooling system construction. Power transmission enables the shifting of generation away from highly water-scarce regions. At the national scale, power transmission enables a lower water requirement for power production. An estimated 10 billion m^3 of withdrawal is saved due to current power transmission[8].The vulnerability decreases in power-importing regions but increases in power-exporting regions. If the closure of highly vulnerable thermal power plants can be compensated for by using wind and solar power in power-exporting regions and increasing power transmission, water vulnerabilities would see further mitigation. With the proposed development of west-to-east transmission lines and hydropower in the southwest[236, 331], power transmission will play a more important role in vulnerability mitigation for water-scarce regions.

China has pledged to make efforts to be carbon neutral before 2060[44, 332], which could be realized with several different energy system choices. Renewables will play an important role in achieving the target. Previous research indicated that China would have to ramp up solar and wind capacity over the next 40 years, including a 16-fold increase in solar and a 9-fold increase in wind, which would represent a significant shift in the temporal and spatial supply of electrical power and require further efforts to ensure energy supply, including short- and long-term energy storage[45, 333]. Hydropower is the second largest electricity supplier in China, it contributes to a low-carbon system but also relies on water resources. As an effective technology for carbon emission reduction, CCS has not been widely adopted in large part because of its high investment[45]. From the perspective of economic costs, the usable capacity reduction caused by CCS should be accounted for in CCS investments. In this case, the capacity-related cost due to water-scarcity of large-scale CCS adoption is US$34-36 billion (or approximately US$10 per tCO_2 sequestered). Fan et al. compared the investment benefits of CCS retrofitting of coal-fired power plants and renewable power generation projects in China, finding that Ningxia, Xinjiang, and Gansu Provinces would be most suitable for the development of CCS retrofitting pilot projects[334]. We suggest that the capacity-related costs of CCS be included in future economic assessments, as should several other factors such as air pollution and ash disposal. In reality, policymakers and entrepreneurs need to incorporate several different adaptation options simultaneously to achieve multiple objectives in terms of, among other factors, power system reliability, economic cost, and compliance with regulations.

5.4.3 Limitations and implications

Although this work integrates water and electricity models and we assess different scenarios, we are unable to exactly predict power production or new power transmission lines due to the difficulty of predicting the capacity, cooling type and location of future CPUs. The optimization of the transmission network is not only important for the reliability of the power supply but can also mitigate the vulnerability of power production to water scarcity. Further research is needed to optimize the network with the consideration of future transmission lines, regional electricity mix, economic cost, etc. The water use factors of thermal plants, specific to China's power plants, were assumed to be constant throughout the year, yet plants often have higher water requirements in summer than in winter due to lower thermodynamic efficiencies[16, 241]. This assumption may lead to an underestimation of the seasonal variations in power-related water use.

In this study, we focus on showing the changes in the performance of thermal power when faced with changing climate and water resources under different climate scenarios rather than the situation in a reference period. Each RCP runs for the period 2006-2099 with different trajectories in radiative forcing and temperature. The reference period used in our study makes the impacts under different RCPs comparable and enables us to see the impacts of changing water availability on power plants. As such the actual water availability in 2021/2022 is not an input of our model, having been calibrated until the end of our reference period in 2001. Further research could use a more recent reference period as and when sufficient meteorological data are available. Other future work could investigate the role of China's power plants in exacerbating water scarcity threats of other nations and the electricity-hydrology model can be used for the nations where the data on power plants are available. Further, more adaptation strategies should be tested according to the local conditions such as resources, infrastructure, and policies. Early decommissioning of coal in China would likely preclude the possibility of RCP8.5 and potentially even RCP6.0. However, given disagreements in the literature between energy and climate modellers we include the full spectrum of results[317, 335]. While updated pathways for carbon emissions are available, PCR_GLOBWB-2 was produced with bias-corrected RCP trajectories. It would be useful to update the analysis using Shared Socioeconomic Pathways in future analyses.

We recommend three actions to further mitigate the vulnerability of thermal power plants. First, it is important to take into account climate and water-scarcity changes when planning the power plant construction. Thermal power plants generally have a long lifetime, which requires assessments on both current and future water resources. Second, the competition for water between the energy sector and other users (agricultural, industrial, domestic, and environmental water requirements) needs to be considered in water resources assessments. Third, the role of adaptation strategies should be considered from the perspective of both individual plants and the power system as a whole, since early retirement is key to reducing water vulnerabilities. With the improvement in transmission technologies and the lowering of transmission costs[336], it is becoming more feasible and important to replace vulnerable power plants in water-scarce regions with ones in regions with sufficient water resources.

5.5 Conclusion

This study presents an assessment of the vulnerability of China's thermal power production to changing climate and water resources using a coupled hydrological-electricity modelling framework. The following conclusions are reached:

China sees a significant spatial heterogeneity in water resources. Nearly half of freshwater-using plants are located in the two major river basins (Yellow and Yangtze) due to their need for water withdrawals. Many plants are close to residential areas and farmland, and are already facing the challenge of competing water with other users, which is an issue that has become worse for the newer power plants over the last two decades. Further, there are seasonal variations in water scarcity, with February the most acute month and September the least. The plant capacity experiencing water scarcity will increase in the 2030s.

The main contributions of the study are showing the water scarcity faced by power production in China, and to what extent CCS will exacerbate the issue.. On the national scale, power production experiences slight changes in the usable capacity in the 2030s relative to the reference period. When CCS is implemented, the vulnerability of power production increases, with additional usable capacity reductions of 7.4-7.7%. Early retirement and interregional power transmission are more effective in vulnerability mitigation than other adaptation strategies from the perspective of usable capacity. However, strategies may also face other challenges from economic, energy security, employment etc. Policymakers and industry will need to be cognizant of the challenges when implementing these adaptation options.

Chapter 6

General Discussion

Energy and water systems are often perceived as unconnected and managed separately. However, electric power is the main water consumer in energy systems and is already a major driver of water stress worldwide. Due to the significant spatial heterogeneity of power production and its water requirement, such problems have not been well addressed by national- and regional-level analyses. This thesis attempted to examine the energy-water nexus at a higher resolution using China as an example, with the aim of answering the following overarching research question:

What are the impacts and challenges of water use of electric power production in China?

This chapter first reviews the progress made toward the specific research questions proposed in Chapter 1 and then answers the overall research question (section 6.1).

6.1 Answers to the research questions

***SQ1**. What are the water requirements of different electricity technologies and what is the availability of regionally specific data? (Chapter 2)*

The literature reported large differences in water requirements of electricity generation technologies. This was a barrier to decision-making. In Chapter 2, we presented a meta-analysis, compiled an inventory of the water use of power production, and investigated the characteristics of water use and uncertainties in assessments. Photovoltaics, wind power, and run-of-river hydropower consume relatively little water; concentrated solar power and geothermal power consume intermediate volumes of water; woody and herbaceous biomass and reservoir hydropower may consume considerable water resources. The deployment location of power production has an important effect on water use of power systems in a country, because of different climate conditions and available types of water resources. Coal power plants have relatively low variability in life cycle water consumption compared with hydropower, which has significant variability due to factors like reservoir area and evaporation factor, with a coefficient of variation of 634% across the water intensity of hydropower assessed in the literature. For thermal power plants, the operational water consumption increases by up to 81% (natural gas) if carbon capture and storage (CCS) is added.

Inconsistent system boundaries caused uncertainties in water use estimates across studies. The fuel cycles of biomass and shale gas merit further consideration in the future because they are large water consumers, driven by the irrigation required to grow plants for biomass and water use in the fracturing process of shale gas extraction. Current studies focus on showing the results of assessments rather than the key influencing factors, such as the conversion efficiency, capacity factor, and lifetime of energy systems. Moreover, the type of water use (consumption vs. withdrawal: as explained in Chapter 1, these are different concepts and need to be distinguished in research) was often not clarified, creating barriers to understanding the results and comparing different studies. Water withdrawal refers to the volume of water diverted from a water source for use, all or part of which may be returned, while consumption refers to the volume of water not returned to the water body due to evaporation, transpiration, or incorporation into products. Clarification of the water sources would also help in interpreting values for water use estimates for biomass (precipitation vs. irrigation), geothermal (geofluid vs. freshwater), and fuel-based thermal and nuclear power (seawater vs. freshwater). The emphasis for future studies should be to increase the transparency and clarity of such factors and terminologies.

***SQ2**. How much water is required for power production in China and how much water is virtually transferred via power transmission? (Chapter 3)*

We assessed the water use of power production in China from the perspective of both water consumption and water withdrawal at the power plant level and then aggregated it to the regional and national levels. Chapter 3 showed that China's power production withdrew 62.7 billion m^3 of freshwater in 2017, of which 13 billion m^3 was consumed. The extensive inventory of plant information allowed for a detailed analysis of the drivers of water use. There were large heterogeneities in the water use of power production across plants. Hydropower plants with large reservoirs were large freshwater consumers due to evaporation, whereas thermal power plants with once-through cooling systems were large freshwater withdrawers. This study showed that it is important to distinguish water sources (surface water, groundwater, reclaimed water, and seawater). Hydropower was the main consumer of surface water, while all nuclear power plants in China were located along the coastline and used seawater for cooling.

Interprovincial electricity transmission increased more rapidly than electricity generation in the last decade. Approximately 16% of China's water consumption for power generation was driven by demand in other provinces and 'virtually used' via power transmission. Power transmission led to a more equal distribution of water stress across provinces. Compared with the east, the west generally had a larger water consumption factor but a lower withdrawal factor. Power generation that consumed large amounts of water was often transmitted from west to east, consequently reducing water withdrawal but increasing water consumption nationally. The impact of power transmission is expected to grow with the rapid development of transmission infrastructure.

SQ3. *What are the impacts of power production on freshwater biodiversity in China? (Chapter 4)*

Current electric power systems require large amounts of freshwater in the thermodynamic conversion of heat to work or the water held in hydropower reservoirs. These processes can result in the consumption of water and warming of the water in the environment (termed 'thermal emissions'). Both freshwater consumption and thermal emissions have impacts on water systems. Chapter 4 assessed the impacts from the perspective of freshwater biodiversity loss.

We found that the total biodiversity loss caused by China's electricity generation increased by 45% during 2008-2017, while the biodiversity impact per unit of electricity generation decreased by 23%. 62% of the biodiversity loss was due to thermal pollution, while 38% was due to freshwater consumption. Electricity transmission resulted in the shifting of biodiversity impacts across regions. The results showed that 15% of total biodiversity loss was driven by electricity transmission to provinces other than those where electricity production and hence biodiversity loss took place. In terms of electric power system drivers of biodiversity loss, the total generation was the main driving factor of the increase in loss (rather than shifts in generation type, for example). Our results highlighted the need to assess the biodiversity impacts of electricity generation and to incorporate them in power system planning. For example, in the future it is important to shift electricity generation from regions with high biodiversity intensities to those with low biodiversity intensities by taking into account the provincial biodiversity factors of electricity generation that we found.

SQ4. *What are the changes in water stress and the consequent impacts on power* *production in the future, and how might future carbon capture and storage (CCS) requirements exacerbate water issues in China? (Chapter 5)*

Thermal power production requires large amounts of water, therefore changes in the availability of water resources due to climate change may affect the vulnerability of power production. Chapter 5 examined the current and future water availability in China based on the outputs of a global hydrological model (PCRGLOBWB-2) under various climate scenarios, and geographically matched this availability to thermal power plants to reveal the impacts on power production. The results showed that there are quite some plants already experiencing water stress, while the impact will be slightly mitigated before 2040 due to an increase in water availability for power plants in northern China in all climate scenarios except one extreme case (Representative Concentration Pathway 8.5).

Many proposed scenarios for meeting China's net-zero carbon targets require that a large number of existing power plants are retrofitted with CCS. Yet the water requirements of CCS mean that its addition exacerbates vulnerability to water constraints compared with the existing situation, leading to additional usable-capacity reductions of 7.4-7.7%. To mitigate such negative implications, we assessed several measures that can enhance usable capacity. We found that early retirement of older, generally more water-intensive power plants and interregional power transmission were more effective in vulnerability mitigation than other adaptation strategies, such as retrofitting cooling systems and switching to seawater cooling. However, strategies may also face other challenges from the economic, energy security, and employment perspectives. Policymakers and industry will need to have insight into such challenges when implementing these adaptation options.

In addressing these research questions, this thesis offers several answers to the overall research question: *What are the impacts and challenges of water use of electric power production in China?*

The thesis answered this question by first providing a meta-analysis of previous assessments of the water requirement of electricity generation. The analysis showed that there were large differences in water requirements of electricity technologies (Chapter 2). Large amounts of water were needed because of the large-scale thermal and hydropower production in China. Water was used for power production and then transferred, virtually, across the power transmission. Water stress was found to be more equally distributed due to the virtual water transfer via power transmission (Chapter 3). In addition to water use, the heat released into water from power plants also had impacts on the water system. We showed that both freshwater consumption and thermal emissions can result in freshwater biodiversity loss (Chapter 4). As a result of the large demand for water, changes in the availability of water resources can affect the vulnerability of electricity generation, with further uncertainties arising from the changing climate and countermeasures (Chapter 5). Both the meta-analysis in Chapter 2 and Chinese case studies in Chapters 3, 4, and 5 showed the importance of increasing transparency in electricity and water systems, and the importance of a joint, spatiotemporally explicit analysis of the electricity and water systems to understand problems and solutions in their nexus.

6.2 Limitations and future research

During our research, we found limitations of data, methods, and scopes that formed barriers to understanding the energy-water nexus. Here we discuss these issues in depth and give some suggestions for future research.

6.2.1 Data limitations

Although energy and water are important resources supporting human activities, data for the two systems are not easily accessible, which limits the scope and transparency of studies.

First, the power system is often divided into three stages when assessing its water use: 1) the fuel cycle, referring to fuel extraction, refining, and transport; 2) the operational stage, referring to the process of electricity production; and 3) plant infrastructure, referring to all the material inputs for plant construction. The operational stage is the focus of the existing research because of its large water requirements, especially for thermal power and hydropower. Yet studies still face many challenges in the availability of high-quality data, which may result in large uncertainties. For example, while the volume of electricity generation is required for making operational water use calculations, it is not usually available and is often estimated by multiplying the installed capacity by a capacity factor. However, capacity factors are seldom reported. If inaccurate values are used, they can lead to substantial uncertainties in the final results. In addition to the capacity factor, key information such as plant location, generator type, and cooling type is also generally not publicly available, which makes it difficult to perform detailed analyses for the operational stage. Information on the other two stages (fuel cycle and plant infrastructure) is also under-reported by governments and businesses, although partly included in some Life Cycle Inventory (LCI) databases, such as *ecoinvent*. Our own study focused on the operational water use rather than lifecycle water use. For some power types, however, the fuel cycle (e.g., biomass) and plant infrastructure (e.g., wind power) may require large amounts of freshwater [16]. Future work needs to enhance data availability of, for example, the location and impacts on water use of extraction of fossil fuels, impacts on water use embodied in materials used for electricity infrastructure, and so on.

Second, information on the type of water is another major source of uncertainty in energy-water studies, and a clear differentiation should also be made between water withdrawal and consumption. Power systems can use various types of water (surface water, groundwater, reclaimed water, and seawater). This thesis mainly distinguished water sources for hydropower, nuclear power, and coal-based thermal power, while more information on water sources for other power technologies (e.g., biomass-based power plants, geothermal power) will be required.

6.2.2 Method limitations

Methodological differences may also yield different results. For example, input-output analysis (IOA) and lifecycle assessment (LCA)-based process analysis have been widely used for assessing water footprints. IOA often leads to larger lifecycle water withdrawal and consumption estimates for most power types, especially wind and photovoltaic (PV). This is because there is an additional water input from economic sectors not covered by LCA-based process analysis [16]. Our studies are mainly based on process data and largely neglect water use in supply chains of fuels and infrastructures, so our estimates could lead to underestimations or represent a lower bound of potential water stresses. Researchers have been attempting to link the two methods in hybrid LCAs for a more comprehensive assessment. However, some issues still need to be addressed. For example, using IO-based hybrid LCA presents a challenge in sector disaggregation. Power production is a homogenous sector in many IO tables, even though each power type has a distinctive water use, as shown in Chapter 2. Work is needed to isolate the targeted power type from the power production sector. IO tables are also normally released later than process-based data, which is an issue for emerging energy technologies. There is a growing trend for battery storage to be paired with renewables, such as solar PV and wind, but these installations are not separated from the power sector in IO tables. Future research could put more effort into disaggregating generation and storage across datasets.

In Chapter 5 we examined the future changes in water availability due to climate change and the associated impacts on electric power production, taking into account the competition for water resources between various sectors. There are still some aspects that need to be improved for the climate, water, and energy models used in the thesis:

- It would be useful to update our analysis using other climate and socioeconomic scenarios, e.g., Shared Socioeconomic Pathways.
- Most power plants withdraw water from the rivers on which they are built, therefore water scarcity is at the plant level rather than the regional and river basin level. In our work, we simulated the water availability at grid cell level at a 5-arcmin spatial resolution, so there are still opportunities to improve the spatial resolution of hydrological simulation in order to obtain more accurate results of the energy-water nexus.
- Our reference period for the hydrological simulation is 1992-2001. Further research could use a more recent reference period as and when sufficient meteorological data become available.
- Our study links a water model with a power production model, while in reality power demand is also important in the power system and affects power production. It would be better to take account of the variations in power demand and the balance between power production and demand when analyzing the relations in the water-electricity nexus.

6.2.3 Scope limitations

Analyzing water and electricity systems may not be enough for assessing how we can realise sustainable development related to these two resources. For example, the energy transition faces constraints with regard to e.g. required greenhouse gas emissions and limitations related to land use. Moreover, the water system is influenced by other developments, such as expansion of agriculture and industrial production. Researchers have been trying to perform analyses with the addition of other nexuses, such as energy-water-CO_2 [337], energy-water-land [338], and energy-water-food [339]. But even with such additions, the system boundary is still limited. Combining the water-electricity nexus with other nexuses requires an integrated assessment model (IAM) that fully integrates environmental systems and economic systems. At the same time, water-electricity models can offer insights into water and electricity systems at a much higher resolution than is possible in IAMs. It would be interesting and important for future research to combine the highly spatiotemporal water-electricity nexus with a traditional IAM to gain a comprehensive view of all nexuses.

The studies presented here mainly analyzed the two systems – water and energy – from the environmental perspective, yet solving environmental issues sometimes involves conflicts with the demands of governments, industries, and residents for economic developments. For example, energy transition and water conservation strategies sometimes require adjustments in other industries [160], which is a serious challenge because of the different policymaking bodies that deal with these issues in most political systems. Industries also face challenges in addressing environmental issues. For example, the regulations on reducing carbon emissions require energy industries to add carbon capture facilities, thus increasing the energy suppliers' costs, which they may partly pass on to consumers. Measures to protect freshwater biodiversity can have impacts on people who live from fishing. For example, the Chinese government imposed a ten-year ban on fishing in the Yangtze River [340] in 2020 to alleviate the reduction in fish stocks. This meant that 111,000 fishing boats, providing the livelihood of 231,000 fishers, had to cease activities. Future research in this area needs to have a wider perspective, addressing but not limited to the challenges above, since many issues extend beyond the natural environment and require trade-offs across various systems.

6.3 Scientific and policy implications

6.3.1 Scientific implications

This thesis provides scientific contributions to both methods and databases in this field. First, in terms of methodological contributions, we conducted a global meta-analysis and concluded that analyses could improve on the terminologies (withdrawal versus consumption), data (which type of water is used), and system boundaries (fuel supply, power production stage, infrastructure). We also provide a methodology to study the impacts of power production and transmission on water resources and biodiversity, which may be useful for similar research in other countries. In addition, we built and implemented an electricity-hydrology model to examine the water vulnerability of power production, with a new research framework for taking account of water competition among various users and evaluating the effectiveness of adaptation strategies for the power system. Second, in terms of data contributions, the data on water requirements of electricity technologies provided by our global meta-analysis can be used for future research on the water-electricity nexus and extended to other related issues, such as studies on the food-water-energy nexus. Further, the information on water use and power production and transmission for China, as one of the major energy users worldwide, can make an important contribution to a database of global energy and water use.

6.3.2 Policy implications

To address global development challenges, the United Nations has set Sustainable Development Goals (SDGs) for 2030, including goals related to the provision of energy and water. Achieving the SDGs requires all relevant stakeholders to work together and manage the synergies and trade-offs among different resources [3]. Currently, energy and water are often managed separately, although there are many connections between them. Understanding the energy-water nexus is an important step toward both a sustainable energy transition and sustainable water management. Against this background, this thesis may provide the following policy-relevant information.

First, from the perspective of guiding the development of energy and water systems, an effective way for policymakers to mitigate energy-water conflicts is to facilitate decoupling of the two systems. We can reduce the water demand of power by developing photovoltaics and wind power, which is also an approach to achieving a low-carbon energy system. However, there is still a long way to go, despite the progress that has been made. Global power production remains dominated by water-intensive technologies. Since freshwater is not the only option, using other, alternative types of water (e.g., seawater and reclaimed water) is an effective way to save freshwater. For example, building coastal power plants with seawater desalination systems can mitigate the reliance of power production on freshwater, and the plant could even become a freshwater supplier. If a region cannot overcome the challenges by itself, using these methods, it could receive assistance from other regions via power transmission. By improving the interconnectivity of electricity grids, power production capacity in water-scarce regions can be downsized and made less susceptible to water stress. The options mentioned here are not the only ones that can help optimize energy and water systems. However, we need to be aware that several options may have trade-offs. For example, air cooling saves water but reduces the energy efficiency of power plants [200]. Wind and solar PV consume negligible amounts of water but face issues in terms of output variability, often creating a need for storage infrastructure. In such cases it is necessary to combine information on the energy-water nexus with other aspects to optimize policy decisions for sustainable development.

Second, from the perspective of supporting further research, an emphasis of future policies should be to increase the availability of data relevant for assessing water and energy systems. Global electricity generation is growing rapidly, dominated by thermal power (77% of the total) and hydropower (16%) [37]. Both are water-intensive energy technologies. Their carbon emissions have been widely monitored and assessed at the power plant level, whereas their water use has not been fully identified and reported. It is important to increase the transparency of water use (quantity, source), as it can vary greatly across power plants of the same type. For example, hydropower is a large water consumer with significant variations across plants, but currently the transparency of its water consumption-related characteristics (e.g., reservoir area, evaporation) is relatively low. Compared with the energy system, the information on the water system is somewhat more difficult to obtain. Improving the accessibility of official data on observational water discharge is a straightforward way to contribute to the nexus studies.

Third, policymakers themselves need to strengthen the links across units responsible for policy development of different resources. At present, resources such as energy and water are often managed by different departments. For example, in China, energy is mainly managed by the National Energy Administration, whereas water is managed by the Ministry of Water Resources. Considering the intractable nexus, it is necessary to develop mechanisms for cooperation between different agencies and ministries, so that synergies in resources conservation can be achieved. In addition, as discussed above, extending the links to departments responsible for socioeconomic topics would provide comprehensive insights and improve policy effectiveness. Furthermore, international collaboration on resources policies is important in transboundary river basins, where environmental impacts may take place across borders.

Chapter 7

Supplementary Information

7.1 Supplementary information to chapter 2

7.1.1 Methodologies

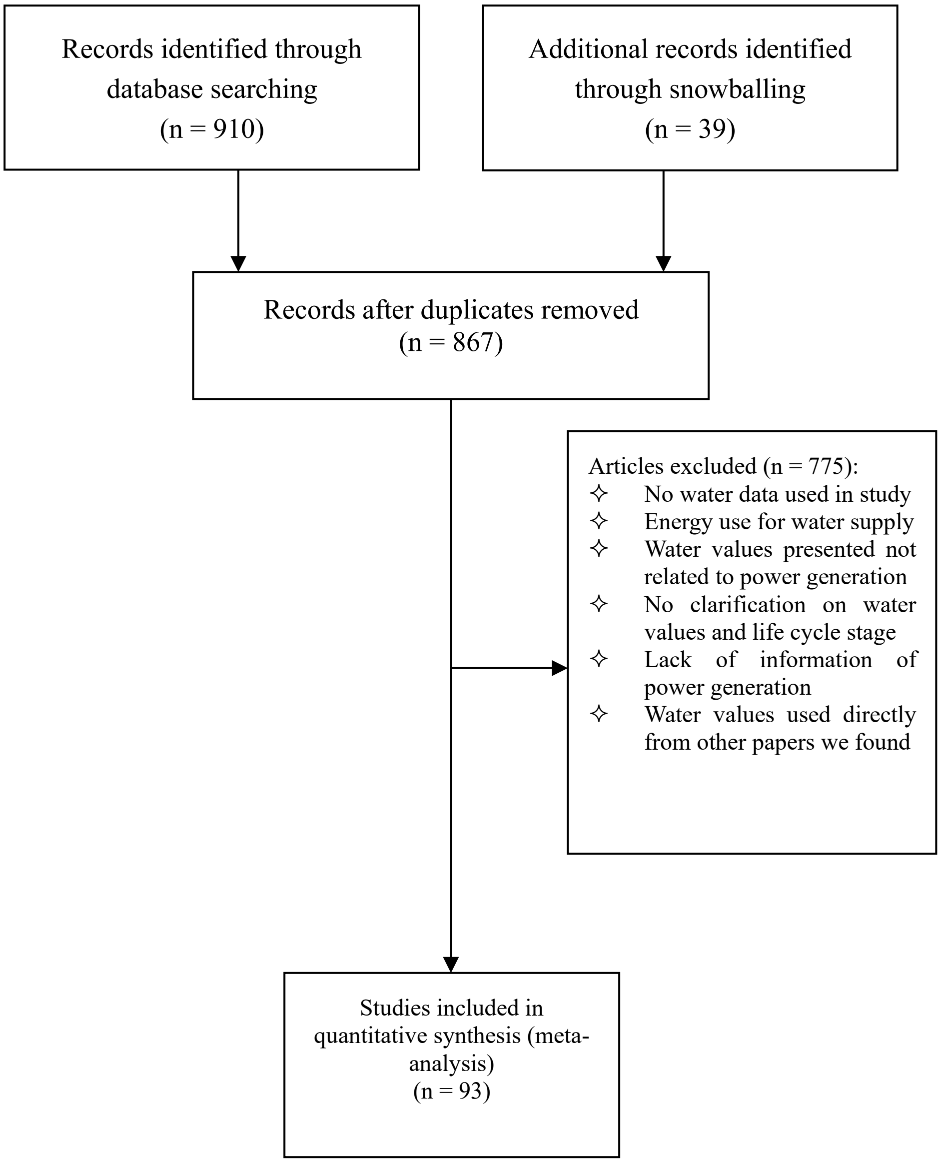


Figure S7.1.1 Flow diagram of meta-analysis.

Table S7.1.1 Search terms used in meta-analysis for each database

Database	Code
Web of Science	TS=(electricity) AND TS=("renewable*" OR "non*renewable*" OR "fossil fuel*" OR coal OR oil OR "natural gas" OR "shale gas" OR nuclear OR "hydro" OR "hydropower" OR biomass OR biofuel OR geothermal OR wind OR solar OR photovoltaic) AND TS=("water footprint" OR "water use" OR "water consumption" OR " water withdrawal" OR "water demand" OR "water requirement") AND LANGUAGE:(English)
ScienceDirect	(ttl(electricity AND (coal OR oil OR “natural gas” OR nuclear OR "hydro" OR "hydropower" OR biomass OR geothermal OR wind OR solar)) OR (key(electricity AND (coal OR oil OR “natural gas” OR nuclear OR "hydro" OR "hydropower" OR biomass OR geothermal OR wind OR solar)) AND (ttl("water footprint" OR “water use” OR “water consumption” OR “water withdrawal” OR “water requirement” OR “water demand”)) AND Article types:(Research articles OR Book chapters OR Data articles)

The operational stage is the focus in previous studies, and the operational water consumption shows the great agreement when grouped by cooling types. In addition to the cooling type, unit type is another determinant of operational water consumption [139], and its main manifestation is the conversion efficiency [65]. Both the changes in cooling type and conversion efficiency would lead to the changes in operational water consumption. Particularly, the effects of changing conversion efficiency on water consumption differ across cooling types, e.g. the 1% change of conversion efficiency is expected to result in more water-saving amounts for closed-loop cooling than for dry cooling due to their different ways and scale of water consumption. Zhang et al. (2014) [139] investigated the relations between operational water consumption and its influencing variables (cooling type and unit type) for coal power plants. The effects of both cooling type and the conversion efficiency on the operational water consumption were considered in this study. In our study, five power types (coal, natural gas, oil, nuclear and biomass) were considered in the model without distinction since their operational water consumption have similar characteristics and shows great agreement when grouped by cooling type as opposed to fuel type (Figure 2.2), the model is expressed as follows:

$$WC_i = \alpha_0 + \alpha_1 CT_{1,i} + \alpha_2 CT_{2,i} + \left(\sum_k \beta_k C_{i,k}\right) CE_i + \varepsilon_i \tag{S7.1.1}$$

In which,

$$CT_{1,i} = \begin{cases} 1, & once - through\ cooling \\ 0, & others \end{cases} \tag{S7.1.2}$$

And,

$$CT_{2,i} = \begin{cases} 1, & closed - loop\ cooling \\ 0, & others \end{cases} \tag{S7.1.3}$$

Through this model, the operational water consumption for once-through cooling, closed-loop cooling, and dry cooling can be expressed by eq (S4), eq (S5), and eq (S6), respectively:

$$WC_i = \alpha_0 + \alpha_1 + \beta_1 CE_i + \varepsilon_i \tag{S7.1.4}$$

$$WC_i = \alpha_0 + \alpha_2 + \beta_2 CE_i + \varepsilon_i \tag{S7.1.5}$$

$$WC_i = \alpha_0 + \beta_3 CE_i + \varepsilon_i \tag{S7.1.6}$$

Where WC_i represents the operational water consumption of sample *i*. the unit of WC_i (L/MWh). CE_i represents the conversion efficiency of sample *i*, with the unit: %. Three types of cooling are distinguished, i.e., once-through cooling, closed-loop cooling, and dry cooling. $CT_{1,i}$ is a binary variable that indicates the cooling type used by sample *i*: 1 for once-through cooling, 0 for other cooling types (closed-loop cooling and dry cooling). $CT_{2,i}$ is a binary variable that indicates the cooling type used by sample *i*: 1 for closed-loop cooling, 0 for other cooling types (once-through cooling and dry cooling). Dry cooling type is the baseline here.

Subscript *k* represents cooling types. $C_{i,k}$ is a binary variable that indicates the cooling type of sample *i*. If sample *i* belongs to cooling type *k*, then $C_{i,k}$ is designated as 1; otherwise $C_{i,k}$ is designated as 0.

α_0, α_1, α_2, and β_k are parameters to be estimated. α_0 is a constant parameter. β_k represents the contribution of the variance of conversion efficiency to the reduction of water consumption for power generation with cooling type *k*. ε_i represents the random error.

7.1.2 Water use of global power generation

Meldrum et al. (2013) reviewed the life cycle water consumption and water withdrawal of power production in the USA and harmonized the estimates from the literature [65]. Since most estimates in previous studies are not accompanied by enough information for harmonization, many studies could not be included in the review. All the estimates of coal thermal power were harmonized to the thermal efficiency of a sub-critical pulverized coal power plant; all the estimates of natural gas thermal power were harmonized to the thermal efficiency of a combined cycle plant. As a result, the water use estimates of coal power could be higher due to the low efficiency of sub-critical generating units compared to other coal power technologies (e.g. sup-critical); the water use estimates of natural gas power could be lower due to the high efficiency of combined cycle plant compared to other natural gas power technologies (e.g. steam cycle).

For biomass, the water use of fuel cycle was often expressed as water use from irrigation or precipitation instead of water consumption or water withdrawal. In this study, referring to the expression in [76, 85-88, 165, 341], the water use of biomass is presented as blue water consumption, with precipitation excluded as we study blue water only. [58, 165] estimated both blue and green water consumption for biomass power from the perspective of global average. Based on their results, the ratio of blue water to the total water consumption was obtained and can be used to adjust the water consumption of biomass power in [75, 85] to the blue water consumption. Both [75, 85] focused on the biomass in Canada and the USA. Mathioudakis et al. (2017) provided both blue water (15840 L/MWh) and green water (186480 L/MWh) of the fuel cycle of corn stover for power production through direct combustion [165]. The total value of blue and green water is comparable to the counterpart in [85], where the value is 256600 L/MWh. Besides, the heat content (17.55 MJ/kg and 18 MJ/kg, respectively) and moisture content (15%, 15%, respectively) of corn stover in both studies are very close. The ratio of blue water to the total water consumption for corn stover is used to adjust the water consumption of corn stover in [85] to the blue water consumption. The blue and green water consumed by pine in [75] were separated using the ratio in [165]. For wheat straw, there are two values of the proportion of blue water in [165], and both of them are 0.27. This proportion is used to separate the blue water from the sum of blue and green water in [75, 85].

Coefficient of variation of life cycle water consumption was calculated as shown in **Table S7.1.2**.

Table S7.1.2 Coefficient of variation (CV) of life cycle water consumption for each power type and technology

Power type	Cooling type	Generating technology	CV (%)
Coal	CL	IGCC	24
Coal	CL	SBC	19
Coal	CL	SPC	23
Coal	CL	USPC	23
Coal	DC	IGCC	14
Coal	DC	SBC	13
Coal	DC	SPC	13
Coal	DC	USPC	13
Coal	OT	SBC	29
Coal	OT	SPC	13
Natural Gas	CL	CC	44
Natural Gas	CL	ST	23
Natural Gas	DC	CC	72
Natural Gas	DC	ST	70
Natural Gas	OT	CC	24
Natural Gas	OT	ST	7
Oil			75
Nuclear	OT	ST	24
Biomass	CL	ST	68
Biomass	DC	ST	77
Biomass	OT	ST	76
Geothermal	DC	EGS-B	89
Geothermal	DC	HT-B	90
Geothermal	HD	HT-B	32
Geothermal	WC	HT-F	23
CSP	DC	Power tower	53
CSP	DC	Trough	46
CSP	HD	Power tower	58
CSP	HD	Trough	52
CSP	WC	Fresnel	10
CSP	WC	Power tower	30
CSP	WC	Trough	13
PV			68
Wind			119
Hydropower			634

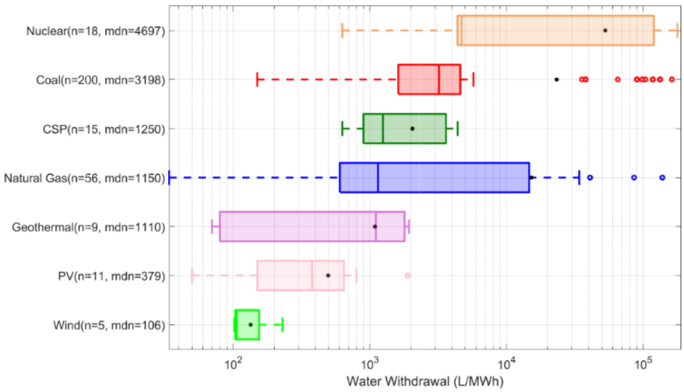


Figure S7.1.2 Life cycle blue water withdrawal for each type of power production. Water withdrawal is visualized on a log scale. The annotation mdn gives the median value of water consumption for each fuel type. The circles represent the outliers while the dots represent the mean for each power type. Hydropower, and biomass are excluded in this figure because the data of their life cycle water withdrawal were not available.

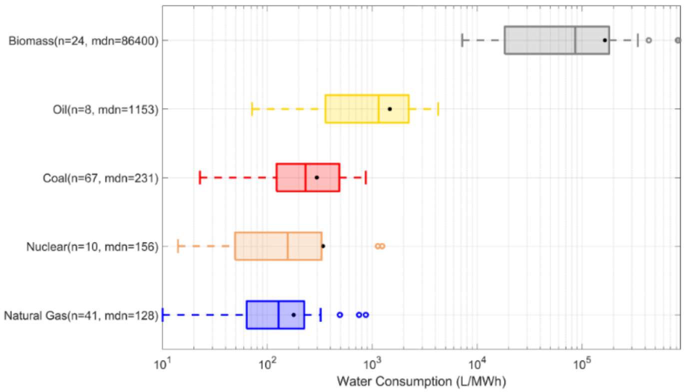


Figure S7.1.3 Blue water consumption in the fuel cycle. The value of water consumption for biomass power from [86] is particularly large (see below) and not included in the figure.

Dominguez-Faus et al. (2009) assessed the irrigation water requirements of corn ethanol and soybean biodiesel used for power generation. Corn ethanol consumed 2.27×10^6 to 8.67×10^6 liters of irrigation water for 1 MWh of power generation. Soybean biodiesel consumed 1.39×10^7 to 2.79×10^7 liters of irrigation water for 1 MWh of power generation [86].

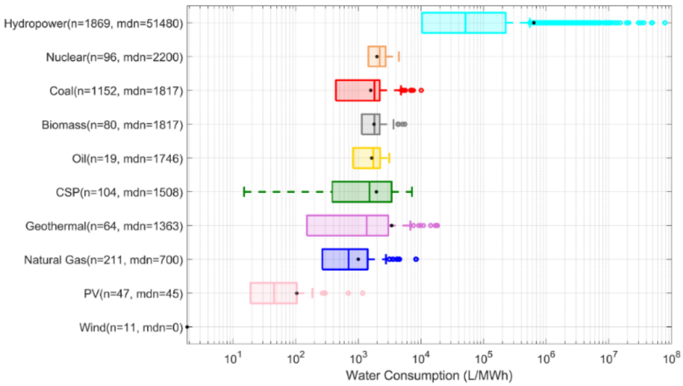


Figure S7.1.4 Blue water consumption of operation. For hydropower, the manifestation of the water use is water evapotranspiration, which is regarded as water consumption, as hydroelectric power generation does not withdraw or divert water [64]. For wind power, a boxplot is not used since most of the values are zero; the median and mean values are 0 and 1.85, respectively. For power types except CSP, their minimum water consumption of operation is zero and the values of q_1 - 1.5×(q_3-q_1) are negative. Thus, values between q_1 and zero are not recognized as outliers. The left whiskers of boxplots extend from q_1 to zero, with all the values between q_1 and zero contained. These left whiskers are not shown in this figure for simplicity.

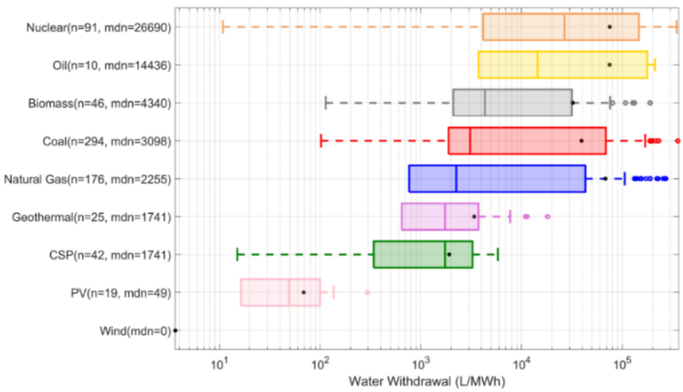


Figure S7.1.5 Blue water withdrawal of operation.

For consistency, the cooling type for coal, natural gas, oil, nuclear and biomass falls into closed-loop (CL), once-through (OT) and dry cooling (DC) referring to [22, 77, 89, 90, 92, 124, 131, 139, 342, 343]; Cooling type falls into wet cooling (WC), hybrid cooling (hybrid) and dry cooling (DC) for geothermal and CSP, referring to [104, 121, 344-351].

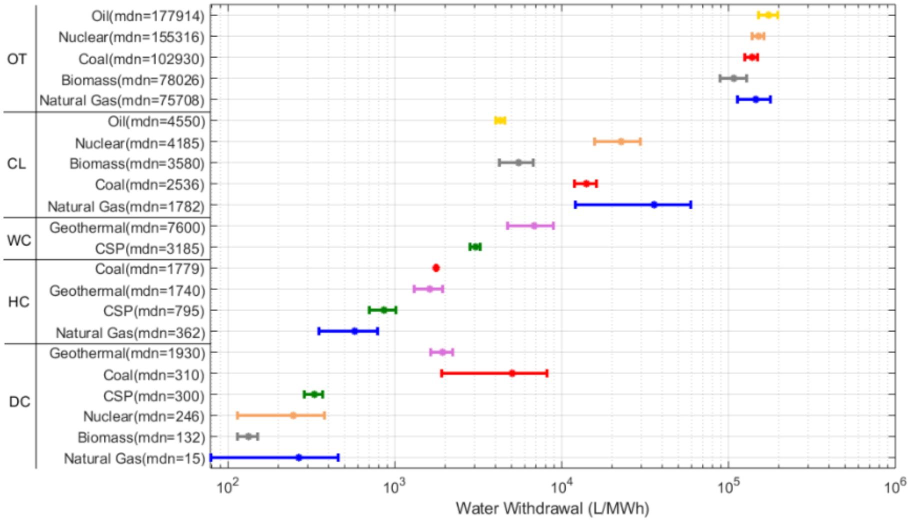


Figure S7.1.6 Blue water withdrawal of operation distinguished by power type and cooling type. The dots represent mean water consumption while the line segments represent the standard error of mean. The annotation mdn gives the median value. Hydropower, wind and PV do not have cooling needs and are not included. WC denotes wet cooling, CL denotes closed-loop cooling, HC denotes hybrid cooling (combining wet and dry cooling); OT denotes once-through cooling, and DC denotes dry cooling.

Water consumption (WC) and water withdrawal (WW) are usually calculated as a function of the lifetime of a power plant. However, the values of lifetime generally came from assumptions, as it is unavailable for plants under operation. The differences in the lifetime assumptions within the same power type makes the estimates less comparable. Therefore, we normalized the WC and WW to the same lifetime assumption if the estimates were provided with lifetime information for the power type. Otherwise, the original WC and WW were used. The normalization is conducted by assuming the water use changes proportionally to the lifetime [65]. Most frequently used lifetimes in the literature are used for normalization. Lifetimes used for normalization are shown in Table S7.1.3. WC estimates for power types including nuclear, wind and PV provided the same lifetime value, thus normalization is not needed within these power types. Only estimates for geothermal, natural gas and coal are normalized referring to the lifetime most frequently used in studies. The results are shown in Figure S7.1.7 and Figure S7.1.8.

Table S7.1.3 The lifetime value used in WC and WW estimates for each power type (unit: years)

Power type	Coal	Natural Gas	Nuclear	Geothermal	Wind	CSP	PV	Hydropower	Oil	Biomass
Lifetime for WC	30	30	40	30	20	30	30	NA	NA	NA
Lifetime for WW	30	30	40	30	NA	NA	NA	40	/	/

Note: NA indicates not all the estimates were provided with lifetime information for that power type, thus normalization was not conducted. No data of water withdrawal were available for oil. There is no water withdrawal for biomass since the water requirement of biomass is considered as water consumption in this study. [65] and [91] provided the lifetime of coal- and natural gas plant. The lifetimes of these two types of plant in this study referred to the values in [65] where the lifetimes were determined based on a review of published literature.

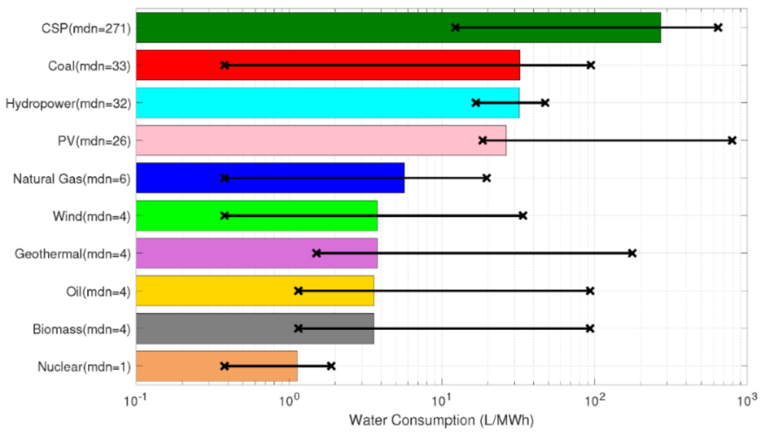


Figure S7.1.7 Blue water consumption of plant infrastructure. The bar shows the median water consumption, while the symbol 'x' represents the minimum and maximum water consumption in plant infrastructure for each power type. The minimum water consumption of wind power was derived from [65] where the value was described as 'less than 0.1 gal/MWh' instead of a precise value. It is assumed to be 0.1 gal/MWh (i.e. approximately 3.79 L/MWh) here. Some of the original values of water consumption were adjusted according to the lifetime of the power plant.

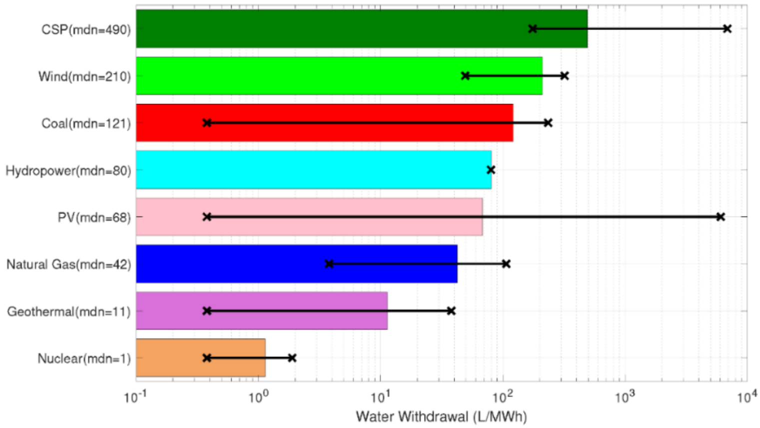


Figure S7.1.8 Blue water withdrawal of plant infrastructure. The bar shows the median water consumption, while the symbol 'x' represents the minimum and maximum water consumption in plant infrastructure for each power type. Only one value of water withdrawal for hydropower was available, which was derived from [64].

7.1.3 Water use of power generation at country level

The median operational water consumption for each country and power type are shown in Figure 7.1.5. For power plants with cooling needs, only wet cooling technologies (closed-loop, once-through and hybrid cooling technology) are included in Figure 7.1.5 since dry cooling is an obviously water-saving cooling technology for all countries, and wet cooling technologies account for 81% of the estimates extracted from the literature. The water consumption and water withdrawal for power generation at each life cycle stage are aggregated at country level. The median, minimum, and maximum values for countries mostly investigated in existing studies are shown in Tables S7.1.4-S7.1.8.

At the plant infrastructure stage, all the studies on the water consumption only focused on the USA, except [96] in which the water consumption for the hydropower station construction has been investigated. The water withdrawal for plant infrastructure of CSP in China is as high as 6863 L/MWh [158] where the material inputs within the whole economic system have been incorparated into the assessment by using a hydrid method. Similarly, the large water withdrawal for plant infrastructure of coal- and gas-supported power generation were obtained from [91] where an economic input-output life cycle assessment tool was employed.

Table S7.1.4 Country-specific blue water consumption of the fuel cycle (L/MWh)

Power Type	Country/Region	Median	Min	Max
Biomass	EU	10,800	7,200	14,400
Biomass	USA	435,600	435,600	435,600
Coal	Spain	120	26	130
Coal	USA	216	23	871
Coal	Canada	238	65	704
Coal	China	246	233	285
Natural Gas	Spain	28	10	45
Natural Gas (Con)	USA	83	23	220
Natural Gas (SG)	USA	222	64	871
Natural Gas (SG)	China	622	492	751
Nuclear	France	48	45	50
Nuclear	USA	212	14	1,249
Oil	Spain	891	281	1,500
Oil (Con)	USA	1,019	72	1,966
Oil (Oil Sand)	USA	1,658	806	2,509
Oil (Oil Shale)	USA	2,342	436	4,248

Note: median denotes median value; min denotes minimum value; max denotes maximum value. Con denotes conventional gas/oil, SG denotes shale gas. The data for Spain contain the water use of fuel production and processing, but not transportation, which is also a determinant of the water use of fuel cycle.

At the operational stage, the only study on the natural gas power in Egypt [352] did not clarify the turbine type of power plant, but the plant studied had similar characteristics of water consumption with combined cycle power plants according to the estimates for other countries. Egypt does not have operable nuclear power plants; Kotb and Abdelaal (2018) estimated the water consumption of nuclear power by scenario analysis [352]. China does not have operable inland nuclear power plants; Guo et al. (2012) and Chen et al. (2010) might made assessment based on research reactors [125, 353]. For biomass power with closed-loop cooling, studies on the USA focused on the water required for cooling [56, 63, 110, 354]. Few research investigated the water consumption of biomass power in China. The two available studies [100, 168] were based on the results in [355], where the water withdrawal instead of water consumption was assessed, and the water used for wastewater processing was included into operational water consumption. The estimates for Spain was provided based on the data of the USA from [63], and consequently had a bias towards the characteristics of water consumption in the USA.

Plants in coastal area can use sea water for cooling, but they also need some freshwater to produce demineralized water, which can be used in the water-steam cycle to drive the electricity generation turbine. Mertens et al. (2015) provided the freshwater consumption of CCGT (combined cycle gas turbine) for cooling tower in Italy (14 L/MWh) and once-through cooling in France (6.6 L/MWh) [173].

Table S7.1.5 Country-specific blue water consumption of operation (L/MWh)

Power type	country	median	min	max
Biomass	USA	1,355	0	5,076
Biomass	Spain	2,152	1,734	2,414
Biomass	China	3,955	2,400	5,530
CSP	USA	1,414	15	7,192
CSP	China	3,415	750	4,000
Coal	India	399	242	4,035
Coal	Canada	1,020	90	3,650
Coal	Egypt	1,150	400	1,900
Coal	Spain	1,552	756	1,815
Coal	USA	1,628	0	10,107
Coal	China	1,889	20	7,070
Coal	UK	1,953	1,193	3,557
Geothermal	USA	1,363	0	18,400
Hydropower	Norway	109	57	161
Hydropower	New Zealand	1,692	324	70,884
Hydropower	Austria	1,962	0	3,924
Hydropower	Spain	9,500	3,000	109,000
Hydropower	USA	11,150	38	210,000
Hydropower	China	23,760	0	15,243,480
Hydropower	Vietnam	24,840	5,040	133,452
Hydropower	DR Laos	49,932	3,708	3,103,200
Hydropower	Turkey	85,860	4,680	118,440
Hydropower	Brazil	133,200	93,600	172,800
Hydropower	Ethiopia	147,240	0	750,240
Hydropower	Egypt	356,500	329,000	6,249,960
Hydropower	Thailand	552,312	1,548	3,160,872
Hydropower	India	1,273,000	1,273,000	1,273,000
Hydropower	Ghana	2,656,080	2,656,080	2,656,080
Natural Gas	Egypt	550	400	700
Natural Gas	Spain	684	0	1,814
Natural Gas	Belgium	722	13	1,431
Natural Gas	USA	795	0	8,438
Nuclear	China	130	40	3,682
Nuclear	Egypt	1,750	1,000	2,500
Nuclear	USA	2,197	0	4,452
Nuclear	Spain	2,590	1,020	3,460
Oil	USA	2,214	698	3,130

Oil	Spain	1,216	0	1,814
Oil	Egypt	800	800	800
PV	China	19	19	19
PV	USA	44	0	1,173
PV	Egypt	100	100	100
Wind	China	0	0	0
Wind	Egypt	0	0	0
Wind	USA	0	0	8

Table S7.1.6 Country-specific blue water withdrawal of operation (L/MWh)

Power type	country	median	min	max
Biomass	USA	3,324	114	189,384
Biomass	China	4,459	3,430	5,530
Biomass	Spain	31,647	1,734	189,600
CSP	USA	1,741	15	5,853
Coal	Canada	1,935	110	199,110
Coal	USA	4,486	102	457,895
Coal	India	5,255	500	159,000
Coal	China	7,610	320	521,251
Coal	Egypt	43,905	2,310	85,500
Geothermal	USA	2,158	0	18,110
Natural Gas	Italy	2,100	2,100	2,100
Natural Gas	Belgium	2,160	1,070	3,250
Natural Gas	USA	2,332	0	1,944,873
Natural Gas	France	2,800	2,800	2,800
Natural Gas	China	4,540	568	79,500
Natural Gas	Spain	13,675	0	189,000
Natural Gas	Egypt	22,050	1,000	43,100
Nuclear	USA	9,842	114	230,000
Nuclear	France	72,318	72,318	72,318
Nuclear	Spain	75,362	1,890	347,200
Nuclear	Egypt	86,050	4,200	167,900
Nuclear	China	178,000	87,000	227,000
Oil	Egypt	1,030	1,030	1,030
Oil	Spain	24,322	0	189,000
Oil	USA	68,520	3,748	211,983
PV	USA	45	0	295
PV	Egypt	100	100	100
Wind	Egypt	0	0	0
Wind	USA	4	0	11

Table S7.1.7 Country-specific blue water consumption of plant infrastructure (L/MWh)

Power type	country	median	min	max
Biomass	USA	4	1	94
CSP	USA	271	12	644
Coal	USA	4	0	95
Geothermal	USA	4	2	177
Hydropower	Norway	33	17	71
Natural Gas	USA	2	0	4
Nuclear	USA	1	0	2
Oil	USA	4	1	94
PV	USA	26	19	795
Wind	USA	4	0	34

Table S7.1.8 Country-specific blue water withdrawal of plant infrastructure (L/MWh)

Power type	country	median	min	max
CSP	USA	375	175	644
CSP	China	6,863	6,863	6,863
Coal	USA	4	0	45
Geothermal	USA	11	0	38
Hydropower	USA	80	80	80
Natural Gas	USA	4	4	4
Nuclear	USA	1	0	2
PV	USA	68	0	6,057
Wind	USA	98	49	314
Wind	Denmark	200	170	320
Wind	Spain	210	210	210
Wind	Italy	250	250	250

Tables S7.1.9-S7.1.11 show the water consumption of the fuel cycle, the water consumption of operation, and the water withdrawal of operation in China, respectively. Tables S7.1.12-S7.1.14 show the water consumption of the fuel cycle, the water consumption of operation, and the water withdrawal of operation in the USA.

In China, the water use of the fuel cycle for shale gas is significantly larger than for coal [89], whereas in the USA they are comparable due to the relatively lower water input for extraction [356]. Replacing coal by conventional gas for power generation can achieve a significant decrease in the water use of the fuel cycle in the USA. At the operational stage, apart from dry cooling, the minimum cooling water consumption and withdrawal in China are 130 L/MWh for nuclear with once-through cooling type and 2180 L/MWh for coal with closed-loop cooling type, respectively. In the USA, the minimum cooling water consumption and withdrawal are both around 360 L/MWh for NGCC (natural gas combined cycle) with hybrid cooling type. Particularly, water consumption of geothermal energy with wet cooling differs greatly, depending on both plant type and resources type. In previous studies, the values of water consumption of coal power plants in China were assumed to be comparable to the median values of corresponding ones in the USA [77, 357, 358], while the values of water withdrawal can vary between the similar plants in China and the USA according to this study. Both in China and the USA, the water use of renewables could exceed that of non-renewables when hydropower, biomass and CSP with wet cooling type are deployed because of their high water use either in fuel cycle or in operation.

Gao et al. (2018) [100] presented the operational water consumption for nuclear power plants with closed-loop cooling technology based on the data from [125, 353]. However, [100] did not provide the calculation process, and the data in [125, 353] would not result in the value in [100]. Therefore, we recalculated the operational water consumption according to [125, 353]. Chen et al. (2010) [353] did not provide the annual operating time of the nuclear power plant and we assume it to be the same as that (i.e. 8147 hours) in [125].

Table S7.1.9 Blue water consumption of the fuel cycle of each power type in China (L/MWh)

Power type	Median	Min	Max
Coal	246	233	285
Shale Gas	622	492	751

Table S7.1.10 Blue water consumption of operation of each power type in China (L/MWh)

Power type	Cooling type	Technology	Median	Min	Max
Coal	CL	IGCC	1,210	1,210	1,210
Coal	CL	USPC	1,873	842	2,269
Coal	CL	SPC	2,029	150	6,900

Coal	CL	SBC	2,174	668	4,043
Coal	CL	SHV	3,270	3,170	3,370
Coal	CL	HV	4,800	4,800	4,800
Coal	DC	USPC	327	242	430
Coal	DC	SBC	336	152	790
Coal	DC	SPC	368	180	717
Coal	OT	USPC	228	190	380
Coal	OT	SPC	310	180	3,000
Coal	OT	SBC	367	170	1,450
Coal	OT	SHV	1,380	1,190	1,570
Natural Gas	CL	ST	2,760	2,120	4,160
Natural Gas	OT	CC	587	76	1,136
Nuclear	CL		3,380	3,077	3,682
Nuclear	OT		74	40	130
CSP	DC		750	750	750
CSP	WC		3,650	3,180	4,000
Biomass	CL		3,955	2,400	5,530
PV			19	19	19
Wind			0	0	0
Hydropower			23,760	0	15,243,480

Table S7.1.11 Blue water withdrawal of operation of each power type in China (L/MWh)

Power type	Cooling type	Technology	Median	Min	Max
Coal	CL	SPC	2,180	2,000	11,470
Coal	CL	SBC	2,625	1,500	3,750
Coal	CL	FB	11,659	11,659	11,659
Coal	CL	LMP	11,848	11,848	11,848
Coal	DC	USPC	1,022	1,022	1,022
Coal	OT	SPC	504,595	504,595	504,595
Coal	OT	FB	512,923	512,923	512,923
Coal	OT	LMP	521,251	521,251	521,251
Natural Gas	CL	ST	4,540	4,540	4,540
Natural Gas	OT	CC	15,045	568	79,500
Nuclear	OT	ST	178,000	87,000	227,000
Biomass	CL		4,459	3,430	5,530

Table S7.1.12 Blue water consumption of the fuel cycle of each power type in the USA (L/MWh)

Power type	Median	Min	Max
Coal	216	23	871
Conventional Gas	83	23	220
Shale Gas	222	64	871
Conventional Oil	1,019	72	1,966
Oil Sand	1,658	806	2,509
Oil Shale	2,342	436	4,248
Nuclear	212	14	1,249
Biomass	435,600	435,600	435,600

Table S7.1.13 Blue water consumption of operation of each power type in the USA (L/MWh)

Power type	Cooling type	Technology	Median	Min	Max
Coal	CL	IGCC	1,620	132	3,140
Coal	CL	SPC	1,878	15	4,350
Coal	CL	FB	2,120	2,120	2,120
Coal	CL	SBC	2,875	757	5,030
Coal	DC	IGCC	227	189	265
Coal	OT	SPC	390	242	469
Coal	OT	SBC	475	269	1,325
Coal	OT	IGCC	625	510	740
Coal	OT	FB	872	795	950
Natural Gas	CL	CC	908	4	1,900
Natural Gas	CL	ST	2,506	19	8,438
Natural Gas	DC	CC	15	0	940
Natural Gas	DC	ST	57	0	114
Natural Gas	Hybrid	CC	360	326	757
Natural Gas	OT	CC	380	8	8,267
Natural Gas	OT	ST	1,200	246	2,358
Oil	CL		2,625	1,100	3,130
Oil	OT		1,100	910	2,233
Nuclear	CL		2,540	1,408	3,403
Nuclear	DC		151	0	265
Nuclear	OT		1,437	106	4,452
Biomass	CL	CC	1,515	1,120	2,080
Biomass	CL	ST	1,817	1,136	3,653
Biomass	DC	ST	0	0	114
Biomass	OT	CC	625	510	740
Biomass	OT	ST	1,136	1,136	1,249
CSP		Dish stirling	19	15	23
CSP	DC	Trough	297	121	625
CSP	DC	Power tower	415	98	606
CSP	Hybrid	Power tower	795	341	4,111
CSP	Hybrid	Trough	1,287	416	4,198
CSP	WC	Power tower	2,909	1,514	3,452
CSP	WC	Trough	3,683	2,120	7,192
CSP	WC	Fresnel	3,785	3,785	3,785
Geothermal	DC	EGS-O	0	0	0
Geothermal	DC	GP-B	76	0	151
Geothermal	DC	HT-B	303	151	1,022
Geothermal	DC	EGS-B	1,363	151	2,725
Geothermal	Hybrid	HT-F	1,200	1,200	1,200
Geothermal	Hybrid	HT-B	5,350	4,200	6,500
Geothermal	WC	EGS-F	0	0	0
Geothermal	WC	HT-F	95	0	14,300
Geothermal	WC	HT-O	8,350	6,800	9,900
Geothermal	WC	EGS-O	9,300	7,600	11,000
Geothermal	WC	HT-B	14,300	5,700	17,200
PV			44	0	1,173
Wind			0	0	8

Note: For geothermal, only freshwater consumed is included.

Table S7.1.14 Blue water withdrawal of operation of each power type in the USA (L/MWh)

Power type	Cooling type	Technology	Median	Min	Max
Coal	CL	IGCC	1,817	606	26,980
Coal	CL	FB	3,785	3,785	3,785
Coal	CL	SPC	4,156	2,196	57,200
Coal	CL	SBC	4,633	1,136	98,421
Coal	DC	IGCC	379	379	379
Coal	OT	IGCC	66,815	53,400	80,230
Coal	OT	FB	75,708	75,708	75,708
Coal	OT	SPC	85,552	85,365	87,064
Coal	OT	SBC	102,587	56,781	215,768
Natural Gas	CL	CC	1,098	53	172,520
Natural Gas	CL	ST	4,069	10	1,944,873
Natural Gas	DC	CC	15	0	379
Natural Gas	DC	ST	114	114	114
Natural Gas	Hybrid	CC	362	329	1,230
Natural Gas	OT	CC	75,708	27,255	266,190
Natural Gas	OT	ST	189,384	37,854	1,595,509
Oil	CL		4,550	3,748	4,550
Oil	OT		177,914	132,490	211,983
Nuclear	CL		4,168	1,893	171,960
Nuclear	DC		246	114	379
Nuclear	OT		145,595	80,304	230,000
Biomass	CL	ST	2,082	1,136	5,527
Biomass	CL	CC	10,090	1,480	26,980
Biomass	DC	ST	132	114	150
Biomass	OT	CC	66,815	53,400	80,230
Biomass	OT	ST	132,489	75,708	189,384
CSP	DC	Trough	282	125	625
CSP	DC	Power tower	454	98	606
CSP	Hybrid	Power tower	644	341	946
CSP	Hybrid	Trough	1,287	1,287	1,287
CSP	WC	Power tower	2,423	1,514	3,100
CSP	WC	Trough	3,577	2,196	4,997
Geothermal	WC	EGS-F	0	0	0
Geothermal	WC	EGS-O	9,300	7,600	11,000
Geothermal	WC	HT-O	9,500	7,700	11,300
PV			45	0	295
Wind			4	0	11

Note: For geothermal, only freshwater consumed is included.

7.1.4 Influencing factors of water use

From the literature, influencing factors of water consumption were collected for each lifecycle stage of power generation: for operational stage, conversion efficiency and capacity factor were investigated; for fuel cycle and plant infrastructure stage, heat content of fuel and lifetime of plants was investigated, respectively. Tables S7.1.15-S7.1.18 show the values of these influencing factors in the literature.

Table S7.1.15 Conversion efficiency of each power type and technology

Power type	Technology	Median	Min	Max
Biomass	ST	25	16	38
Biomass	CC	37	37	59
Coal	FB	35	35	36
Coal	SBC	38	33	54
Coal	SPC	40	26	44
Coal	USPC	43	39	45
Coal	IGCC	45	39	45
CSP	Fresnel	10	9	11
CSP	Trough	15	9	16
CSP	Dish stirling	16	9	22
CSP	Power tower	20	9	20
Geothermal	EGS-B	9	9	9
Geothermal	HT-B	9	8	10
Geothermal	HT-F	11	11	11
Geothermal	DS	12	8	15
Natural gas	CT	33	30	33
Natural gas	ST	33	32	33
Natural gas	SC	40	40	40
Natural gas	CC	51	39	75
PV	Thin film	13	12	13
PV	c-Si	13	13	13
PV	Thin film, III-V	37	37	37
Nuclear		33	31	40
Oil		36	36	36
Wind		39	39	39

Table S7.1.16 Capacity factor of each power type

Power type	Median	Min	Max
CSP	48	22	56
Coal	85	75	90
Geothermal	95	95	95
Natural gas	85	80	85
Nuclear	92	92	92
PV	23	23	23
Wind	27.4	19	46

Table S7.1.17 Lifetime of each type of power plant

Power type	Median	Min	Max
Biomass	30	30	30
CSP	30	20	30
Coal	30	30	60
Geothermal	30	20	30
Natural gas	45	30	60
Nuclear	33	20	50
PV	30	30	30
Wind	20	20	20

Table S7.1.18 Heat content of fuel

Power type	Median	Min	Max
Biomass	17.83	13.45	20.00
Coal	20.64	16.28	27.14
Natural gas	53.54	49.61	55.00

The relations between influencing factors and water consumption were investigated by calculating their correlation coefficients, as shown in Table S7.1.19. For operational water consumption, conversion efficiency and capacity factor were investigated in this study; for water consumption of fuel cycle and plant infrastructure stage, heat content of fuel and life time of plants was investigated, respectively.

The negative signs of these coefficients show that in general, water consumption can be to some extent reduced by increasing the values of these factors. However, the coefficients cannot accurately measure the effects of these factors on water consumption, because variables influencing water consumption differed across different literature and could influence the coefficients. For example, fuel-cycle water consumption of conventional natural gas was larger in [92] than in [91]. This large difference in water consumption could be caused by different extraction approaches, burning conditions and other variables. Even within [92], The natural gas with the same heat content has different fuel-cycle water consumption. There are only two studies available for calculating correlation coefficients for both PV and CSP. Although the correlation coefficient shows the strong relations between water consumption and conversion efficiency, this relation is not reliable due to the unknown backgrounds of studies since the operational water consumption can be largely affected by other various conditions, such as practical habit of panel cleaning, sunshine duration etc. Generally, water consumption can be to some extent reduced by increasing the values of these factors. However, the correlation coefficients cannot accurately measure the effects of these factors on water consumption due to other differences in previous studies that cannot be controlled in the analysis.

Meldrum et al. (2013) harmonized the results from the literature using assumed values for influencing factors and assuming water consumption changed proportionally to the values of factors. We could not obtain the original values of factors and water consumption, thus the results in [65] were not used in this section.

Table S7.1.19 Correlations between influencing factors and water consumption

Heat Content	Cooling Types	Correlation Coefficients
Coal	/	-0.14
Natural gas	/	-0.88
Conversion Efficiency		
Coal	CL	-0.12
Coal	DC	-0.29
Coal	OT	-0.03
CSP	DC	-0.20
Nuclear	CL	-0.68
Natural gas	OT	-0.27
Natural gas	DC	-0.71
Biomass	CL	-0.91
Natural gas	CL	-0.19
PV	/	-1.00
CSP	WC	1.00
Capacity Factor		
Coal	CL	-0.34
Life Time		
Geothermal	/	-0.58

Note: For natural gas power plants, only combined cycle units were investigated due to data limitation. For capacity factor, only coal power plants with sup-critical units were investigated.

The regression model (eq. A1-A6) was solved by “regstats” function in Matlab version 2018a. The residuals of regression model are distributed at the both sides of the line: residual = 0, as shown in Figure S7.1.9. The outliers are not discussed here since they are included in the analyses of other sections of this study. Besides, there is no reason to excluded outliers in previous studies for a better fitness of regression.

Table S7.1.20 Parameter estimation of the operational water consumption for power generation

Parameter	Estimation	p-value of t stats
α_0	742	0.00***
α_1	626	0.10*
α_2	2867	0.00***
β_1	-16.22	0.02**
β_2	-36.83	0.00***
β_3	-10.33	0.10*
Number of samples: 720		
R^2: 0.7442, adjusted R^2: 0.7424		
F: 415.37, and its p-value: 0.00		

Note: Significance symbol: *** p<=0.01, ** p<=0.05, *p<=0.1; k = 1 for once-through cooling, k = 2 for closed-loop cooling, and k = 3 for dry cooling.

Figure S7.1.9. The residual distribution of regression model.

For hydropower, the literature generally provided both key influencing factors of water use, i.e. reservoir area and evaporation rate. An analysis of variance was conducted to look at the effects of both factors on the operational water consumption of hydropower. Results (Table S7.1.21) showed that evaporation rate generally plays a more important role in determining the operational water consumption of hydropower compared to reservoir area.

Table S7.1.21 Analysis of variance in hydropower

Source	Sum of Squares	F	Sig.
Reservoir area	6.46×10^{12}	133.84	0.00
Evaporation rate	1.19×10^{14}	2791.39	0.00
Error	6.99×10^{9}		
Total	3.70×10^{14}		

7.1.5 Uncertainties in methodological choice

Studies using Hybrid LCA focused on life cycle water consumption for power generation. The average values of life cycle water consumption estimates for PLCA and hybrid LCA method are shown in Table S7.1.22, respectively. Only PLCA was used for estimating water consumption for geothermal.

The water use of biomass and hydropower mainly originates from the direct water use, which is included in the LCI phase of both methods, indicating that the water use of these two power types is determined by the actual water use rather than assessment methods. The low estimates based on hybrid LCA for biomass and hydropower were assessed in [83], where both power types used low direct water consumption. Thus, the counterintuitive result occurs that estimates based on hybrid LCA is significantly lower than the counterparts based on conventional PLCA.

Table S7.1.22 The average values of life cycle blue water consumption estimates using PLCA and hybrid LCA methods

	Coal	Natural gas	Oil	Nuclear	Wind	CSP	PV
LCA	2,190	861	3,185	2,062	14	2,152	234
Hybrid LCA	2,537	1,229	3,220	3,100	633	2,400	1,295

For hydropower, approximately 25% of the world's reservoirs with a dam higher than 15 m serve multiple purposes [141], such as flood control, drinking water supply, irrigation, power production, recreation, navigation and more [94, 140]. The hydroelectric water use of a multi-purpose power station may be overestimated if a reservoir's water use is attributed entirely to power production [95]. Existing allocation models have been built based on the following allocation factors: the ranking of hydropower production among multiple purposes [24], the ratio of economic value of hydropower to the total economic value derived from the reservoir [141], the ratio of water volume for hydropower to the total water volume for all the functions of a reservoir [359], and the ratio of power production of hydropower to the total power (power production of hydropower and lost power production due to other functions of a reservoir) [359].

Table S7.1.23 The range of allocation factors of water use for hydropower

	Economic allocation	Energy allocation	Volume allocation	Ranking of purpose
Min	0	0.02	0.02	0.33
Max	0.96	0.43	0.39	1.00

7.1.6 Uncertainties in system boundary delineation

Table S7.1.24 The ratio of blue water consumption per stage to life cycle water consumption (%)

	Fuel cycle	Plant infrastructure
Coal	3-38	~0-15
Nuclear	3-47	~0
Oil	9-64	~0-12
Natural gas	2-79	~0-1
Biomass	~100	~0
Hydropower	/	~0-46
CSP	/	8-91
PV	/	12-93
Wind	/	~0-~100
Geothermal	/	~0-~100

Note: Water withdrawal is not investigated because it is generally assumed to be equal to water consumption for fuel cycle and plant infrastructure. "~0" and "~100" means the ratio is approximate to 0 and 100, respectively. "a-b" means the ratio ranges from a to b.

Within the fuel cycle, there are two main sub-stages: fuel production and fuel transport. The former typically includes fuel extraction, processing, and often revegetation, while the latter are performed mainly in two ways for coal, gas, and oil: conventional transport (e.g. truck, train, shipping) and pipelines [65, 89-92, 169]. Compared to fossil fuels such as coal and natural gas, biomass has generally a lower energy density. Large amounts of biomass feedstock need to be transported from the field to the power plant. To reduce transport costs, biomass power plants are generally close to the field of biomass feedstock and highway transport is the primary transport mode

[360-362]. In this case, the water consumption is relatively small at this stage [360].

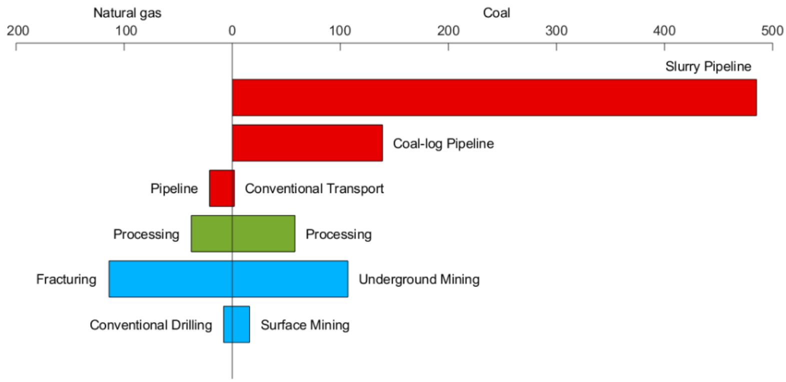


Figure S7.1.10 The median blue water consumption of the sub-stages of fuel cycle for coal and natural gas. Unit: L/MWh. For the range of water consumption, see Table S8.1.25 and Table S7.1.26.

Table S7.1.25 and Table S7.1.26 shows the water consumption (median, minimum, and maximum values) of sub-stages of fuel cycle for coal and natural gas, respectively. Detailed information within fuel cycle are not available for oil. The extraction approaches are generally used according to the existing state of resources. For coal, extraction approach can be further divided into: surface mining and underground mining [64, 90, 363, 364]. For natural gas, the extraction approaches are: conventional drilling and fracturing [65, 91, 92, 109]. For coal, the transport approach can be further categorized as: conventional transport (train, truck, shipping), coal-log pipeline, and slurry pipeline [90]. Coal slurry pipelines transport a slurry of water and pulverized coal over long distances and the ratio of coal to water is about 1 to 1 by weight [124, 365]. Coal-log pipelines works in the similar way as slurry pipelines, but the mass ratio of coal to water is lower-- is about 3:1 [90].

Table S7.1.25 Blue water consumption of coal fuel cycle (L/MWh)

Sub-stages of fuel cycle		Water consumption		
		Median	Min	Max
Extraction	Surface mining	16	2	49
	Underground mining	107	30	681
Processing		58	30	2785
Transport	Conventional transport	2	0.38	4
	Coal-log pipeline	139	117	150
	Slurry pipeline	485	379	1552

Table S7.1.26 Blue water consumption of natural gas fuel cycle (L/MWh)

Sub-stages of fuel cycle		Water consumption		
		Median	Min	Max
Extraction	Conventional drilling	8	0.38	72
	Fracturing (shale gas)	114	72	163
Processing		38	2	55
Transport	Pipeline	21	4	32

7.2 Supplementary information to chapter 3

Table S7.2.1 The capacity factor and its variation for electricity-generating units by region in 2017.

Provinces	Capacity factor	Standard deviation	Coefficient of variation
Anhui	0.71	0.05	6.40
Chongqing	0.69	0.10	15.29
Fujian	0.74	0.05	6.27
Gansu	0.68	0.06	9.05
Guangdong	0.70	0.06	8.06
Guangxi	0.66	0.11	17.20
Guizhou	0.72	0.05	7.45
Hainan	0.68	0.01	1.26
Hebei	0.71	0.12	16.23
Heilongjiang	0.62	0.06	10.31
Henan	0.64	0.07	11.09
Hubei	0.76	0.04	5.10
Hunan	0.63	0.06	10.15
Inner Mongolia	0.72	0.06	8.71
Jiangsu	0.72	0.04	5.22
Jiangxi	0.72	0.03	3.56
Jilin	0.58	0.03	5.25
Liaoning	0.61	0.07	11.04
Ningxia	0.70	0.07	9.39
Qinghai	0.75	0.06	8.02
Shaanxi	0.73	0.08	10.91
Shandong	0.74	0.05	6.23
Shanghai	0.66	0.08	11.90
Shanxi	0.70	0.04	6.22
Tianjin	0.69	0.05	6.71
Xinjiang	0.63	0.05	7.69
Yunnan	0.69	0.12	18.03
Zhejiang	0.72	0.04	5.12

Table S7.2.2 The descriptive statistics of electricity-generating units by region in 2017 in our database.

Provinces	Number of units	Installed capacity (MW)	Electricity output (GWh)	percentage of coal power (%)	percentage of hydropower (%)
Anhui	168	54963	248119	78	19
Beijing	22	2187	5222	22	72
Fujian	187	59689	273271	39	45
Gansu	143	28714	99048	65	35
Guangdong	307	102412	502646	63	25
Guangxi	191	50968	192730	34	57
Guizhou	195	50570	167709	52	47
Hainan	30	6082	26112	50	42
Hebei	202	52445	279989	93	7
Henan	261	75833	279662	85	13
Heilongjiang	102	20855	76039	95	0
Hubei	225	66208	240477	40	56
Hunan	176	32053	95201	33	62
Jilin	126	25162	62372	64	23
Jiangsu	318	94615	477357	86	2
Jiangxi	106	30951	139894	56	40
Liaoning	175	45012	202602	83	9
Inner Mongolia	386	109235	486316	100	0
Ningxia	95	32780	133836	90	10
Qinghai	54	13769	38609	23	77
Shandong	417	107724	435635	95	1
Shanxi	261	76283	304116	94	6
Shaanxi	182	54442	275537	82	18
Shanghai	35	15119	54777	89	0
Sichuan	192	36777	123620	17	83
Tianjin	38	14084	60525	100	0
Tibet	18	188	310	0	27
Xinjiang	293	66588	294537	93	7
Yunnan	216	49435	152169	19	79
Zhejiang	200	67577	317632	52	28
Chongqing	87	18829	58800	63	34

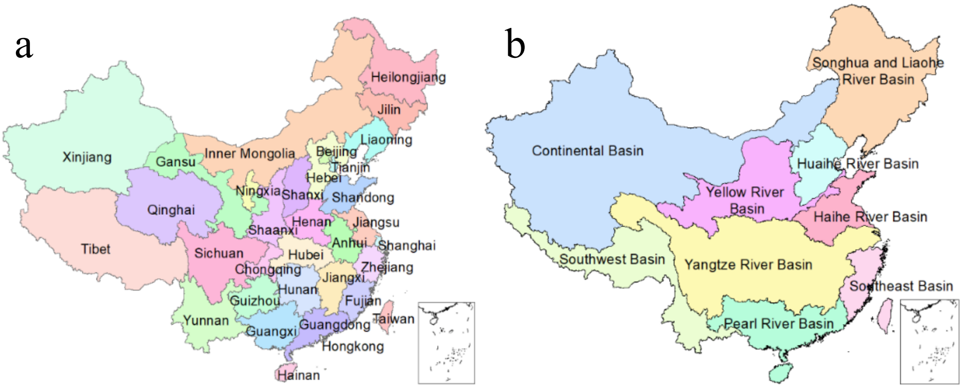
Table S7.2.3 Virtual water transfer and co-benefits on water-saving of power transmission unit: million m^3

Water sources		Water consumption	Water withdrawal
Surface water	Transfer	2086	6047
	Savings	1867	15806
	Co-benefits	-	+
Groundwater	Transfer	29	130
	Savings	37	490
	Co-benefits	+	+
Reclaimed water	Transfer	42	64
	Savings	103	178
	Co-benefits	+	+
Seawater	Transfer	15	5036
	Savings	91	28223
	Co-benefits	+	+

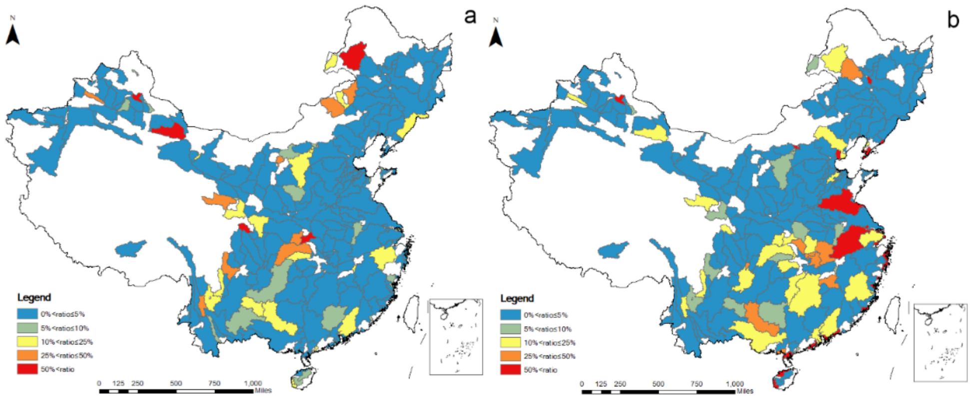
Note: '+' indicates power transmission decreases water use, '-' indicates power transmission increases water use.

Table S7.2.4 Short names of provinces.

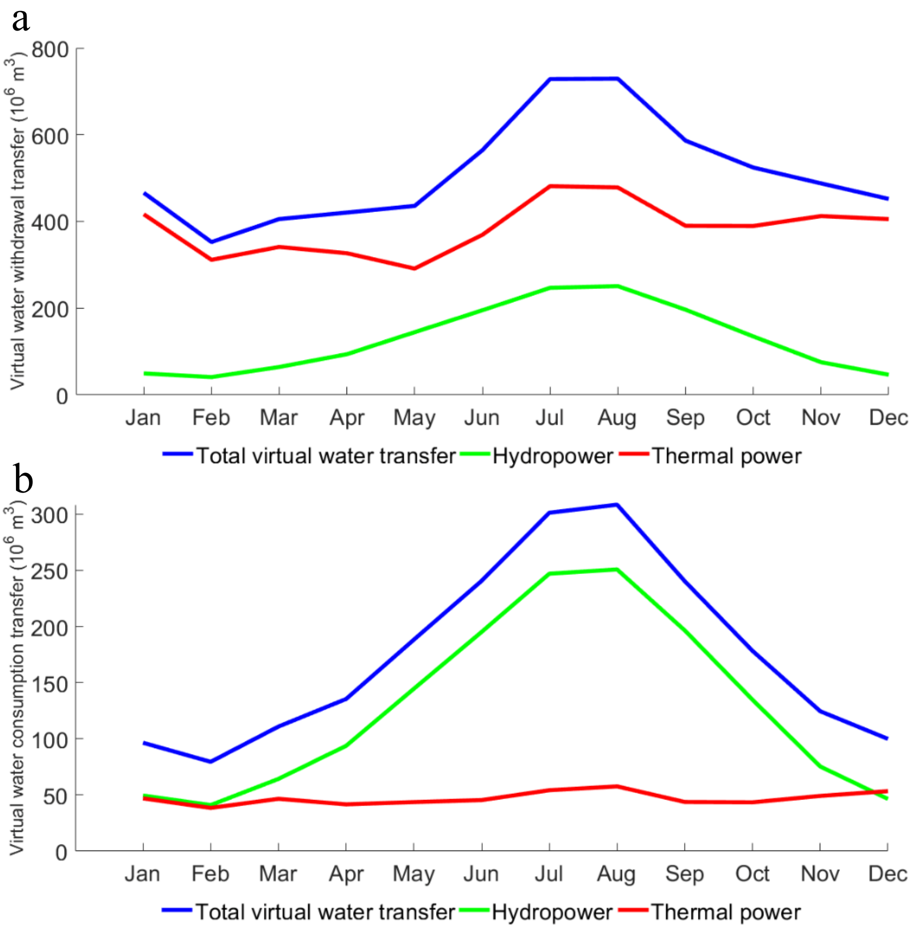
Provinces	Short names	Provinces	Short names
Anhui	AH	Liaoning	LN
Beijing	BJ	Inner Mongolia	IM
Fujian	FJ	Ningxia	NX
Gansu	GS	Qinghai	QH
Guangdong	GD	Shandong	SD
Guangxi	GX	Shanxi	SX
Guizhou	GZ	Shaanxi	SHX
Hainan	HAIN	Shanghai	SH
Hebei	HEB	Sichuan	SC
Henan	HEN	Tianjin	TJ
Heilongjiang	HLJ	Xizang	TI
Hubei	HB	Xinjiang	XJ
Hunan	HN	Yunnan	YN
Jilin	JL	Zhejiang	ZJ
Jiangsu	JS	Chongqing	CQ
Jiangxi	JX		



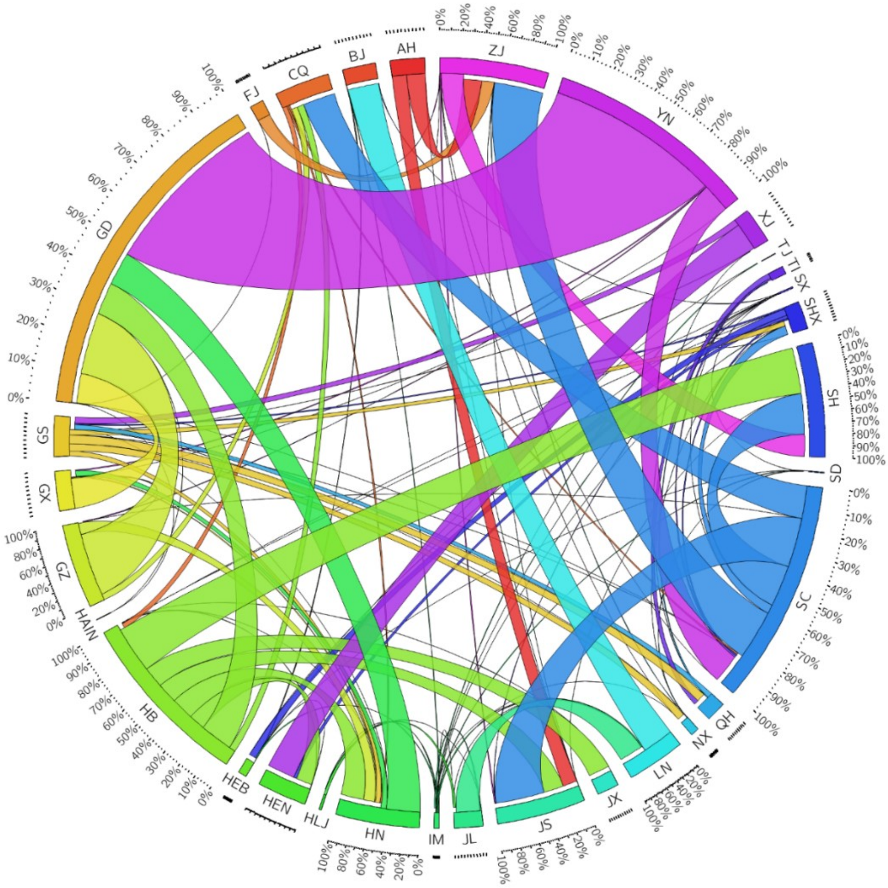
Figures S7.2.1 Map (a) presents provinces and map (b) shows the major river basins.



Figures S7.2.2 The ratio of water consumption of power production to the total water consumption (a) and the ratio of water withdrawal of power production to the total water withdrawal (b) at basin level in China in 2017.



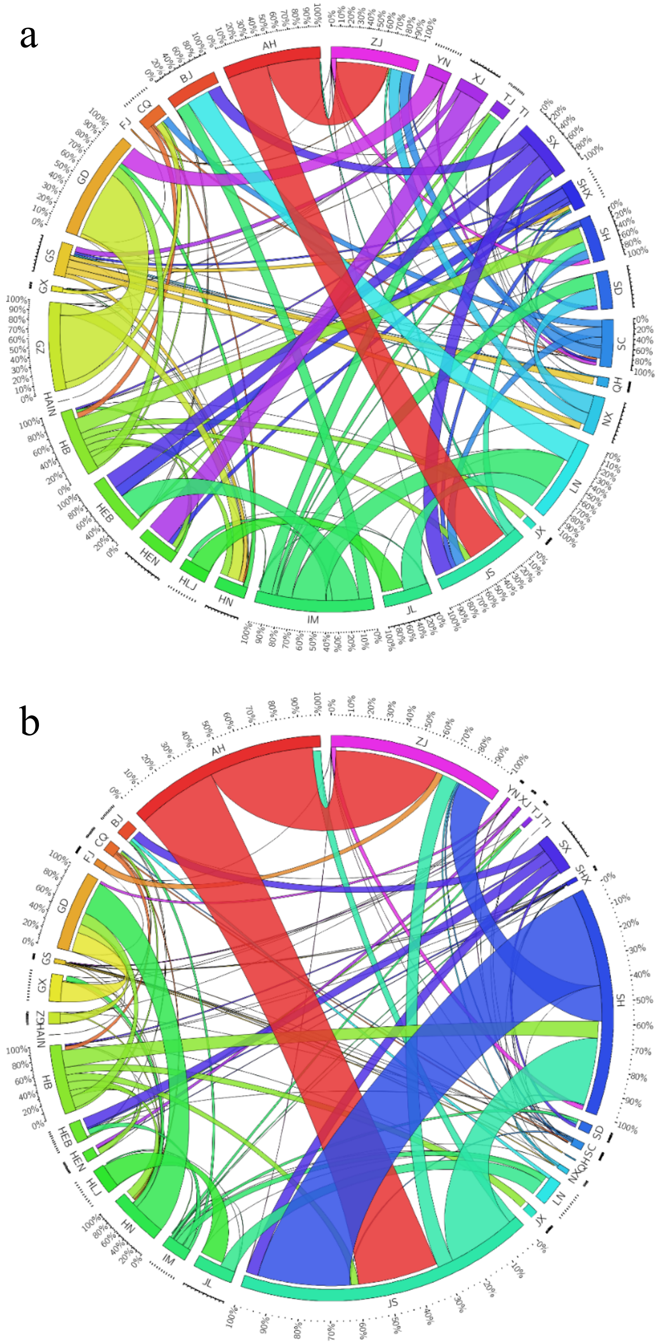
Figures S7.2.3 The monthly virtual water consumption (a) and withdrawal (b) transfers via hydropower and thermal power transmission in China.



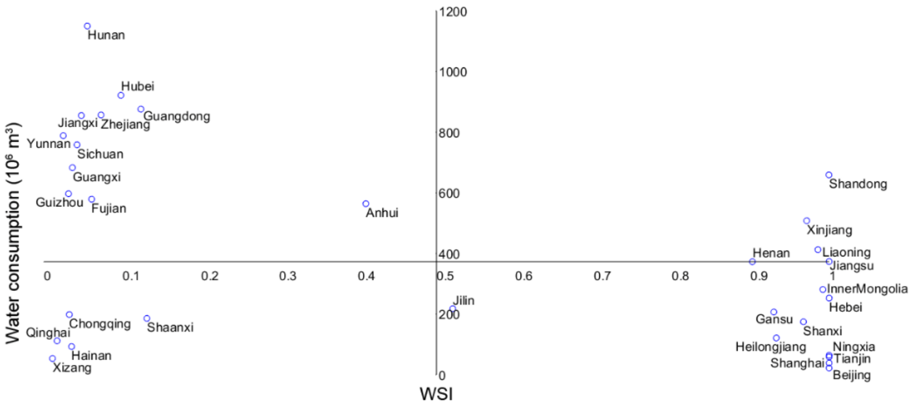
Figures S7.2.4 Virtual water consumption/withdrawal transfer via hydropower transmission. The full and short names of provinces are shown in **Figure S7.2.4.**

a

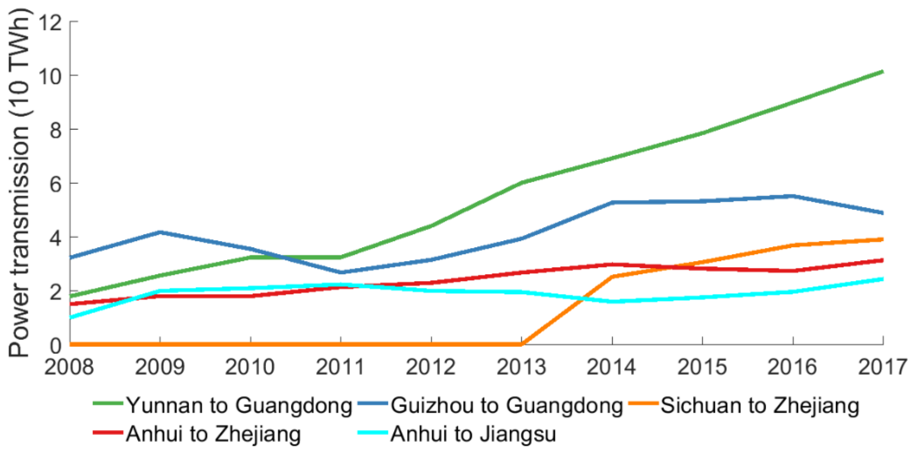
b



Figures S7.2.5 Virtual water consumption (a) and withdrawal (b) transfer via thermal power transmission.



Figures S7.2.6 The provincial WSI and freshwater consumption.



Figures S7.2.7 Power transmission of main corridors between provinces.

Construction of databases

Power: The information on thermal power plants was from Global Coal Plant Tracker [38] and World Electric Power Plants Database [215]; the information on hydropower was from GRanD v1.3 [216] and Liu et al. [204]. For the cooling system of thermal power plants, we used Google satellite imagery cross-checked with information from the China Electricity Council [217]. According to previous studies [118, 366], it is easy to identify a power plant in a satellite image by visual inspection using the images of the cooling facilities. For example, recirculating cooling systems are equipped with cooling towers and air cooling systems are equipped with air cooling islands; once-through cooling systems do not have such cooling equipment. It is also worth noting that indirect air cooling systems also have cooling towers, but with a different appearance that can be identified by visual inspection.

Transmission: For each province, the sum of electricity generation and imports should theoretically equal to the sum of electricity consumption and exports [220]. Due to statistical discrepancies, there are small differences of around 2% for each province (except for Hebei, Heilongjiang and Shaanxi where the difference is 3%). We do not consider transmission loss in this study, mainly due to data unavailability.

Water consumption and withdrawal: Water consumption and withdrawal within basins were obtained from the World Resources Institute Aqueduct database for the baseline year 2010 [53, 224]. By summing up the water consumption/withdrawal of each basin, we get the national total water consumption/withdrawal assessed by the Aqueduct database in 2010, denoted by A. We also get the total amount of water consumption/withdrawal in 2017 from the National Bureau of Statistics [201], denoted by B. And then we can get the ratio between B and A, denoted by C. The water consumption and withdrawal of each basin in our study (2017) was obtained by multiplying the corresponding basin's water consumption/withdrawal in 2010 by C.

7.3 Supplementary information to chapter 4

S7.3.1 Estimates of water consumption

We assessed the provincial water consumption factors for thermal power and hydropower generation using the method in Jin et al. [16]. Based on the factors, the water consumption of electricity generation in each year is calculated as follows:

$$WC = \sum_i WC_i = \sum_i (TWC_i + HWC_i) = \sum_i (TF_i \cdot TP_i + HF_i \cdot HP_i) \qquad \text{(S7.3.1)}$$

In which, WC gives the national water consumption for electricity generation (m^3); WC_i the water consumption for electricity generation in provinces i (m^3); TWC_i the water consumption for thermal power generation in province i (m^3); HWC_i the water consumption for hydropower generation in province i (m^3); TF_i the water consumption factor for thermal power generation in province i (m^3/GWh); HF_i the water consumption factor for hydropower generation in province i (m^3/GWh); TP_i the thermal power generation in province i (GWh); HP_i the hydropower generation in province i (GWh).

S7.3.2 Characterization factors for water consumption

Water consumed for electricity generation is not returned to the river. The influence of reduced flow rates on aquatic biodiversity can be quantified with the global species-discharge model, an index of habitat space, feeding and reproductive opportunities. This model is developed based on native fish species and river discharges in various river basins [274]. This model assumes a positive correlation between the number of freshwater fish species and average river discharges at the mouth of river basins.

$$R = 4.2 \cdot Q_{mouth,i}^{0.4} \tag{S7.3.2}$$

Where R is the freshwater fish species richness and $Q_{mouth,i}$ is the annual average river discharge at the river mouth of basin i (m^3/s).

The species-discharge relationship can be used to calculate characterization factors for water consumption that specify freshwater fish species loss per unit of reduced river discharge for river basins in different regions. Characterization factors (CF_c) for water consumption reflect the impact of water use due to human activities on freshwater biodiversity loss.

$$CF_{c,i} = FF_i \cdot EF_i = \frac{dQ_{mouth,i}}{dW_i} \cdot \left(\frac{dPDF_i}{dQ_{mouth,i}} \cdot V_i\right) \tag{S7.3.3}$$

Where FF_i is the fate factor of river basin i, EF_i is the effect factor of river basin i ($PDF \cdot m^3 \cdot yr \cdot m^{-3}$), $dQ_{mouth,i}$ is the marginal change in water discharge at the river mouth in basin i ($m^3 \cdot yr^{-1}$), dW_i is the marginal change in water consumption by human activities in river basin i ($m^3 \cdot yr^{-1}$), $dPDF_i$ is the marginal change in the potentially disappeared fraction of the freshwater fish species due to the marginal river discharge change $dQ_{mouth,i}$ and V_i is the volume of river basin i (m^3). The $dQ_{mouth,i}/dW_i$ is assumed to be equal to one, indicating that a change in water consumption is fully reflected in a change in water discharge at the mouth for that river basin.

$$\frac{dPDF_i}{dQ_{mouth,i}} = \frac{dR_i}{R_i \cdot dQ_{mouth,i}} = \frac{4.2 \cdot 0.4 \cdot Q_{mouth,i}^{0.4-1}}{4.2 \cdot Q_{mouth,i}^{0.4}} = \frac{0.4}{Q_{mouth,i}} \tag{S7.3.4}$$

The river volumes (m^3) for all river basins are calculated according to Hanafiah et al. [276] as follows:

$$V_i = 0.47 \cdot \left(\frac{Q_{mouth,i}}{2}\right)^{0.9} \cdot L_i \tag{S7.3.5}$$

Where V_i is the water volume in river basin i (m^3), $Q_{mouth,i}$ is the discharge at the river mouth in basin i ($m^3 \cdot yr^{-1}$), Li is the length of river i (m).

China can be divided into the following river basins: Huaihe, Haihe, Yellow, Yangtze, Pearl, Southeast, Southwest, Continental, Songhua and Liaohe river basins. The characterization factors are calculated for these river basins. Specifically, Qiantang and Min rivers are the representatives of the Southeast river basin. The characterization factors of Qiantang and Min river basins are calculated for Zhejiang and Fujian provinces since they are the largest river basins of the two provinces, respectively [282, 367]. Talimu is the largest river in the Continental basin, and its characterization factor is calculated for this basin. In terms of Southwest, Brahmaputra is the largest river basin of Tibet, and its characterization factor is calculated for Tibet [368]. Nandu river is the largest river of Hainan province, and its characterization factor is calculated for Hainan. The discharges at the river mouth and the river length are obtained from the Ministry of Water Resources [281-284].

S7.3.3 Thermal pollution of power production

S7.3.3.1 Thermal pollution from thermal power

In power plants with once-through cooling systems, water from a freshwater body is used to absorb heat from the working fluid in the condenser. The entire volume of heated water is then discharged back into the water body. In the Rankine cycle of steam-electric generating units, pumps and boilers add heat to liquid water, which is converted into steam during that process. The high-pressure steam then expands in the turbine producing power. Upon exiting the turbine, the steam passes through the condenser where heat is rejected from the system turning the working fluid into a saturated liquid, ready to re-enter the pump. To calculate the rate of heat rejected in each cycle, the difference in enthalpy of the working fluid on either side of the condenser must be multiplied by the steam flow rate. The thermal pollution to water bodies from thermal power is calculated using the method of Raptis and Pfister [267]. The heat rejection rates of thermal power are assessed as follows:

$$Q_t = LF \cdot m_{steam}(h_b - h_a)/1000 \quad \text{(S7.3.6)}$$

Where Q is the heat rejection rate (MW), LF is the load or capacity factor of electricity generating units, which are derived from Jin et al. [8], m_{steam} is the steam flow rate at the high-pressure turbine (kg/s), h_b-h_a is the difference in enthalpy of the working fluid on either side of the condenser (kJ/kg).

The steam flow rate can be calculated as:

$$m_{steam} = \beta \cdot C_{gross} \quad \text{(S7.3.7)}$$

Where C_{gross} is the gross generating capacity (MW) and β is a constant (0.830 kg s^{-}

[1] MW[-1 267]).

Reheat cycles can be added into the Rankine cycle, increasing the generation efficiency and thus reducing fuel inputs. When reheat cycles are employed, the steam passes first through a high-pressure turbine and, after being reheated, through a low-pressure turbine. 94% of China's units with an installed capacity of 100-220 MW use a reheat system, whereas all 300-1000 MW units use a reheat system [217]. For a reheat system, the ratio (*r*) of the steam flow at the entry of low-pressure turbine to the steam flow at the entry of high-pressure turbine is inserted to scale the rejection rate:

$$Q_t = LF \cdot m_{steam} \cdot r \cdot (h_b - h_a)/1000 \quad \text{(S7.3.8)}$$

Where r=0.85 is used for China's units in this study, referring to Yan et al. [369] and Cheng et al. [370]. h_a is related to the water temperature withdrawn for use in the condenser [267]. An additional necessary piece of information for all thermodynamic cycles is the temperature of the freshwater withdrawn for use in the condenser. To obtain these values, the georeferenced power plants are overlaid onto gridded estimates (at 10 km spatial resolution) of water temperatures [371]. The average over 15 years (2000-2014) is used to minimize the impacts of very warm or very cold years on the water temperature estimates. For every generating unit then, mean monthly naturalized water temperatures are extracted. The information on power plants is sourced from the China Electricity Council [217], Global Coal Plant Tracker [38], World Electric Power Plants Database [215] and our previous study [8]. The results of plant-level thermal pollution and its impacts are then aggregated to the provincial level.

S7.3.3.2 Thermal pollution from hydropower

Hydropower also produces heat during operation, though its thermal pollution is smaller than that of thermal power with a once-through cooling system because of its higher energy efficiency. The thermal emission of hydropower can be calculated as follows:

$$Q_h = LF \cdot C_{gross} \cdot HR \quad \text{(S7.3.9)}$$

Where *HR* is the heat emission rate of China's hydropower, referring to Xu et al. [372] and Yan and Hao [373]; here, *HR* is 1.8%. The definitions of *LF* and C_{gross} are the same as those in Equations 7-9.

There are approximately 47,000 hydropower stations in China [298]. It is infeasible to assess the thermal emission and biodiversity impacts at the plant level because of data limitations. We made assessments at the provincial level by changing equation 10 to 11:

$$Q_h = LF_p \cdot C_p \cdot HR \quad \text{(S7.3.10)}$$

Where LF_p is the provincial load or capacity factor of hydropower, C_p is the provincial installed capacity of hydropower (MW). The values of LF_p and C_p are obtained from the National Bureau of Statistics [218].

S7.3.4 Biodiversity impacts assessments

Biodiversity loss caused by freshwater consumption: Electricity generation can cause aquatic biodiversity loss because of its water use [61, 374]. Surface water consumption impacts aquatic biodiversity. Water consumption is translated to impacts on aquatic biodiversity by characterization factors expressed as a potentially disappeared fraction of species (unit: PDF m^3yr /m^3) [276].

$$WBL_i = WC_i \cdot CF_{c,i} \quad \text{(S7.3.11)}$$

Where WBL_i gives the biodiversity loss caused by water consumption for electricity generation in province i (PDF m^3 yr); WC_i the water consumption for electricity generation in province i (m^3); $CF_{c,i}$ the biodiversity loss per unit of water consumption for electricity generation in province i (PDF m^3 yr/m^3);

Biodiversity loss caused by thermal emissions: The factor of local biodiversity impacts from thermal emissions is obtained from Raptis et al. [61] .

The biodiversity loss caused by electricity generation is calculated as follows:

$$TBL_i = TBLT_i + TBLH_i = \sum_n Q_{t,n,i} \cdot TBF_{n,i} + Q_{h,i} \cdot PTBF_i \quad \text{(S7.3.12)}$$

Where TBL_i gives the biodiversity loss caused by thermal emissions of electricity generation in province i (PDF m^3 yr); $TBLT_i$ the biodiversity loss caused by thermal emissions from thermal power in province i (PDF m^3 yr); $TBLH_i$ the biodiversity loss caused by thermal emissions from hydropower in province i (PDF m^3 yr); $Q_{t,n,i}$ the thermal emissions from the thermal power plant n in province i (MJ); $TBF_{n,i}$ the biodiversity loss per unit of thermal emissions at the location of plant n in province i (PDF m^3 yr/MJ); n the thermal power plants with once-through cooling systems in province i; $Q_{h,i}$ the thermal emissions from hydropower in province i (MJ); $PTBF_i$ the biodiversity loss per unit of thermal emissions in province i (PDF m^3 yr/MJ); the characterization factors are derived from Raptis et al. [61], where global gridded freshwater thermal pollution CFs are assessed. We extract the gridded CFs for thermal power and China's CFs for hydropower at monthly resolution.

S7.3.5 Estimates of water stress

The Water Stress Index is calculated according to Pfister et al. [375], which is adapted from the water withdrawal-to-availability indicator by applying a logistic function to acquire continuous values between 0.01 and 1. The equation is as follows:

$$WSI = \frac{1}{1+e^{-6.4 \cdot WTA^{*}\left(\frac{1}{0.01}-1\right)}} \qquad \text{(S7.3.13)}$$

Where *WSI* is the water stress index. *WTA** is a modified *WTA* indicator considering the difference for watersheds with and without strongly regulated flows. Four levels of water stress are classified in the *WSI*, i.e. minor (0.01-0.09); moderate (0.09-0.5); severe (0.5-0.91); and extreme (0.91-1). The water stress indexes are calculated for 2017 based on the water withdrawal and availability from the Ministry of Water Resources [225].

S7.3.6 Converting local impacts to global impacts

Kuipers et al. estimated global extinction probabilities (GEPs) based on species range sizes, species vulnerabilities, and species richness, indicating to what extent regional species loss in the respective area may contribute to global species loss. They generate them for marine, terrestrial, and freshwater species groups on the local (i.e., 0.05° × 0.05° grid) and ecoregion scale [290]. The regional fractions of freshwater species losses are then multiplied with the corresponding GEPs to calculate potential global fractions of extinctions:

$$GBL_i = BL_i/V_i \cdot GEP_i \qquad \text{(S7.3.14)}$$

Where GBL_i gives the potential global biodiversity loss in province *i* (PDF yr); V_i is the volume of the representative river in province *i*; GEP_i is the global extinction probability in province *i*, calculated by aggregating the cell-level GEPs from Kuipers et al. [290] within province *i*.

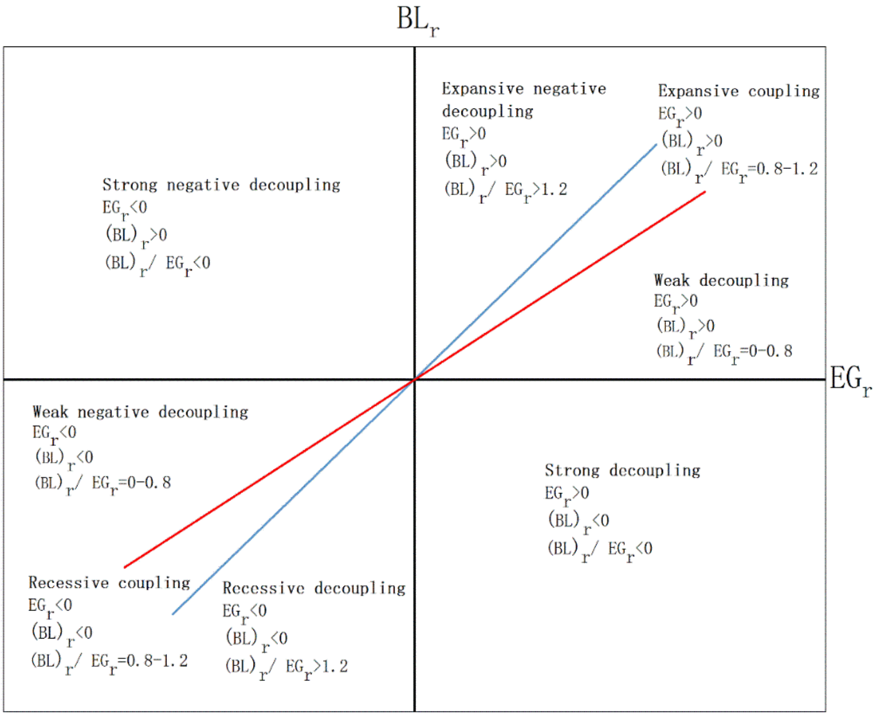


Figure S7.3.1 The decoupling state quadrant map corresponding to the decoupling degree. Here, $BL_r = \Delta BL/BL_{t-1}$, $EG_r = \Delta EG/EG_{t-1}$. This map is modified from Tapio [277].

Figure S7.3.2 China's provinces.

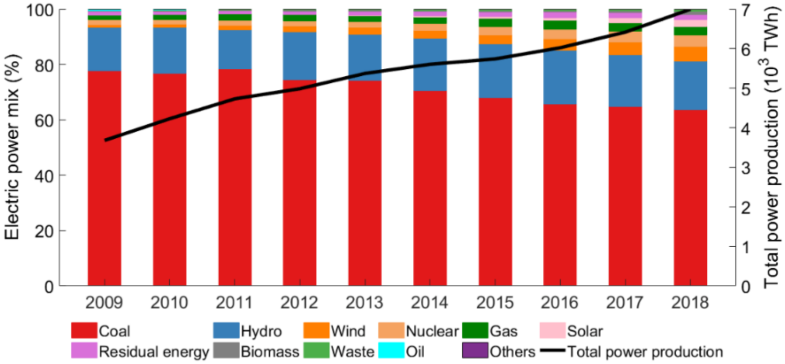


Figure S7.3.3 The electric power mix in China.

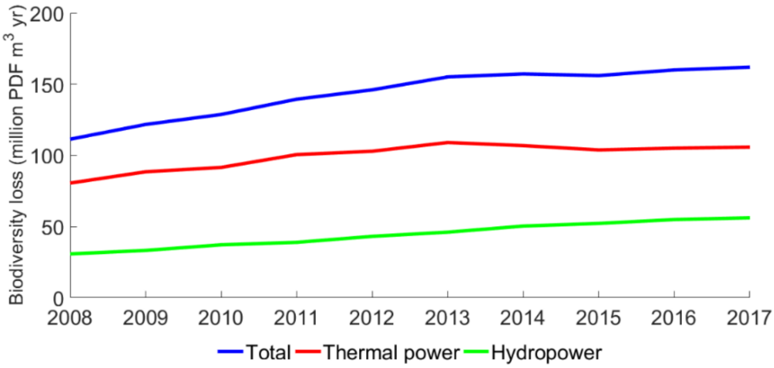


Figure S7.3.4 The biodiversity loss by freshwater use for China's electricity generation during 2008-2017.

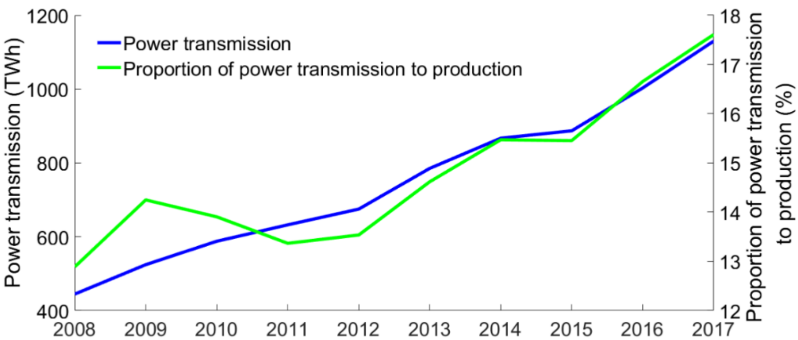


Figure S7.3.5 The total amount of interprovincial power transmission in China.

Table S7.3.1 The provincial characterization factors of water consumption impacts on local biodiversity.

Province	**Characterization factor (PDF·m^3·yr·m^{-3})**	**Province**	**Characterization factor (PDF·m^3·yr·m^{-3})**
Beijing	1.76E-03	Hubei	7.19E-03
Tianjin	1.76E-03	Hunan	7.19E-03
Hebei	1.76E-03	Guangdong	2.78E-03
Shanxi	8.17E-03	Guangxi	2.78E-03
InnerMongolia	8.17E-03	Hainan	5.79E-04
Liaoning	2.51E-03	Chongqing	7.19E-03
Jilin	2.84E-03	Sichuan	7.19E-03
Heilongjiang	2.84E-03	Guizhou	7.19E-03
Shanghai	7.19E-03	Yunnan	3.99E-03
Jiangsu	7.19E-03	Xizang	2.79E-03
Zhejiang	6.76E-04	Shaanxi	8.17E-03
Anhui	7.19E-03	Gansu	8.17E-03
Fujian	8.14E-04	Qinghai	8.17E-03
Jiangxi	7.19E-03	Ningxia	8.17E-03
Shandong	1.50E-03	Xinjiang	3.34E-03
Henan	1.50E-03		

Table S7.3.2 The provinces' full names and abbreviations used in Figure 4.4.

Full name	**Abbreviation**	**Full name**	**Abbreviation**
Anhui	AH	Liaoning	LN
Beijing	BJ	Inner Mongolia	NM
Fujian	FJ	Ningxia	NX
Gansu	GS	Qinghai	QH
Guangdong	GD	Shandong	SD
Guangxi	GX	Shanxi	SX
Guizhou	GZ	Shaanxi	SN
Hainan	HI	Shanghai	SH
Hebei	HE	Sichuan	SC
Henan	HA	Tianjin	TJ
Heilongjiang	HL	Xizang	XZ
Hubei	HB	Xinjiang	XJ
Hunan	HN	Yunnan	YN
Jilin	JL	Zhejiang	ZJ
Jiangsu	JS	Chongqing	CQ
Jiangxi	JX		

Table S7.3.3 The decoupling degree and decoupling state (see Figure S.7.3.1) between biodiversity loss and electricity generation.

Time period	**Decoupling degree**	**Decoupling state**
2008-2009	1.4	Expansive negative decoupling
2009-2010	0.39	Weak decoupling
2010-2011	0.7	Weak decoupling
2011-2012	0.87	Expansive coupling
2012-2013	0.8	Expansive coupling
2013-2014	0.31	Weak decoupling
2014-2015	-0.31	Strong decoupling
2015-2016	0.52	Weak decoupling
2016-2017	0.19	Weak decoupling

7.4 Supplementary information to chapter 5

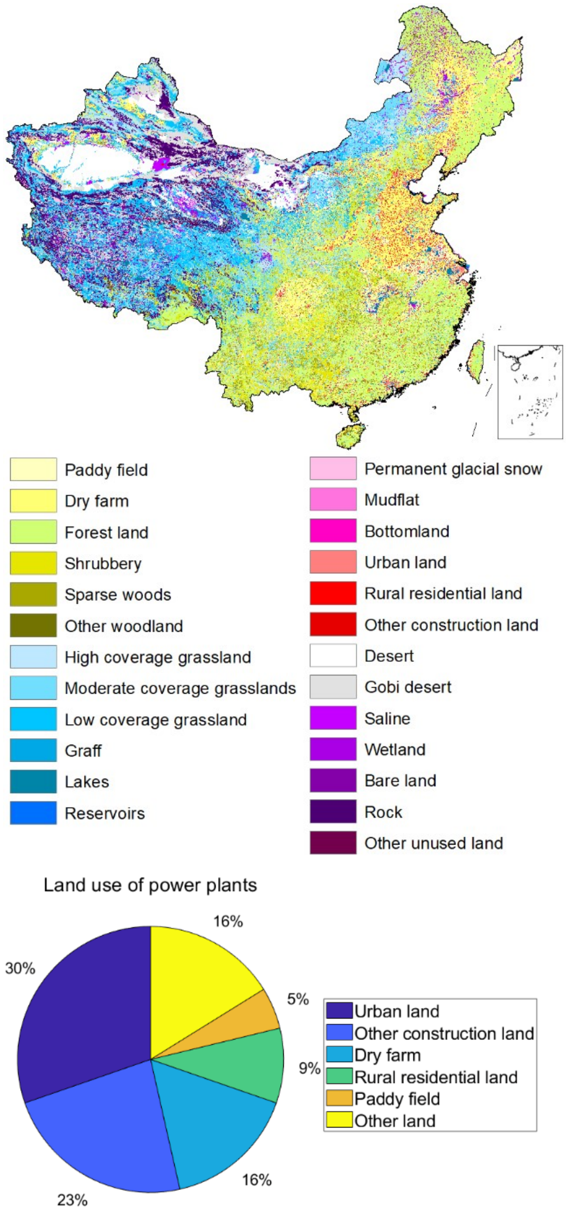


Figure S7.4.1 China's land use type (1km resolution) and the type of power plants' location. Data source: Resource and Environment Science and Data Center [376].

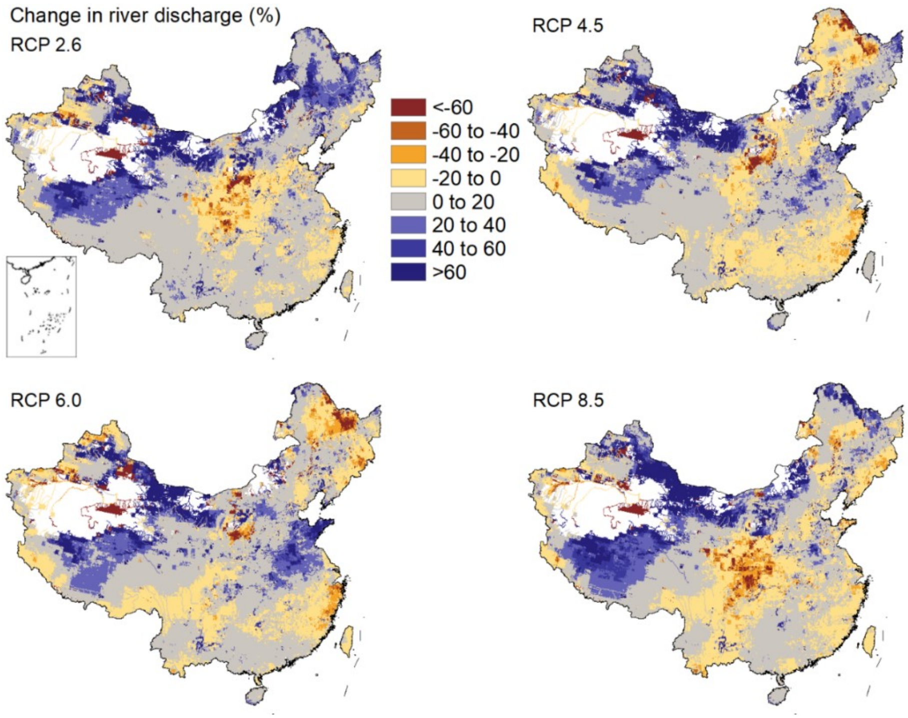


Figure S7.4.2 Impacts of climate change on annual average river discharge. Maps of changes in river discharge for climate scenarios RCP2.6, RCP4.5, RCP6.0, and RCP8.5 in the 2030s relative to the reference period.

Figure S7.4.3 Map (left) presents provinces and map (right) shows the major river basins.

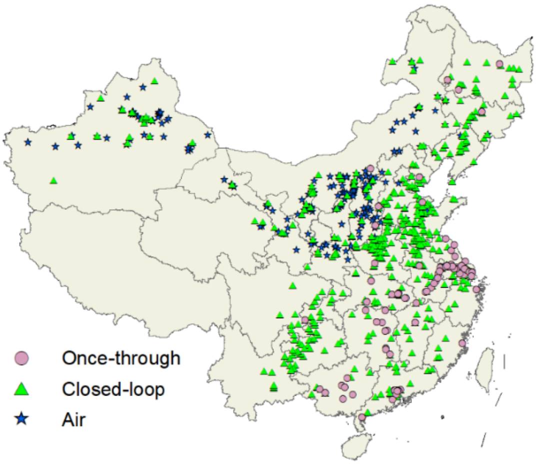


Figure S7.4.4 Coal-fired power plants' location and cooling type.

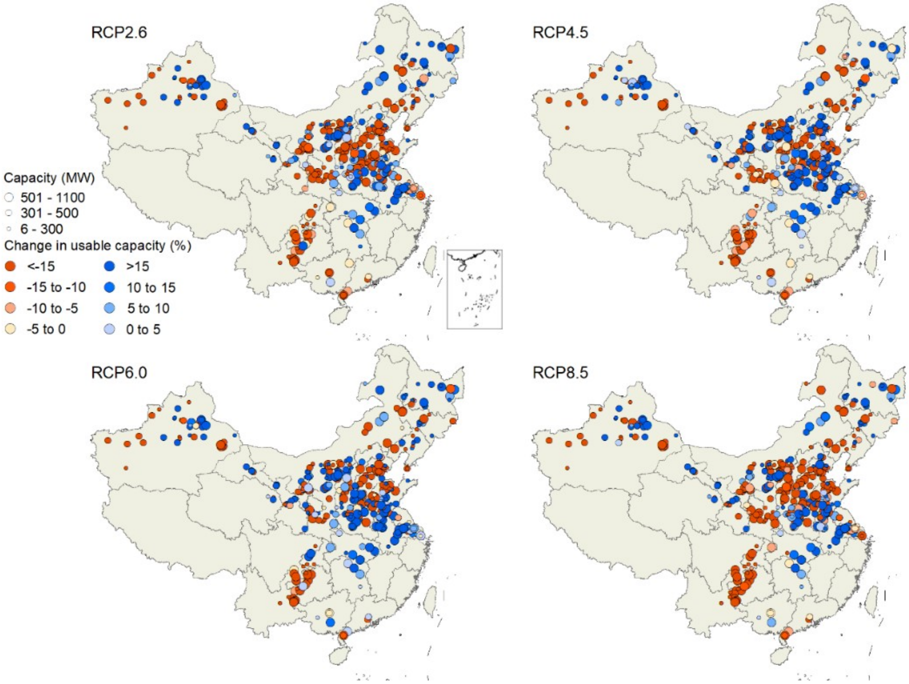


Figure S7.4.5 Impacts of climate and water resources change on usable capacity of CPUs in spring (March, April, May). The changes in the annual usable capacity under four climate scenarios in the 2030s compared to the reference period. Those that do not experience changes in usable capacity reductions are not shown on the maps.

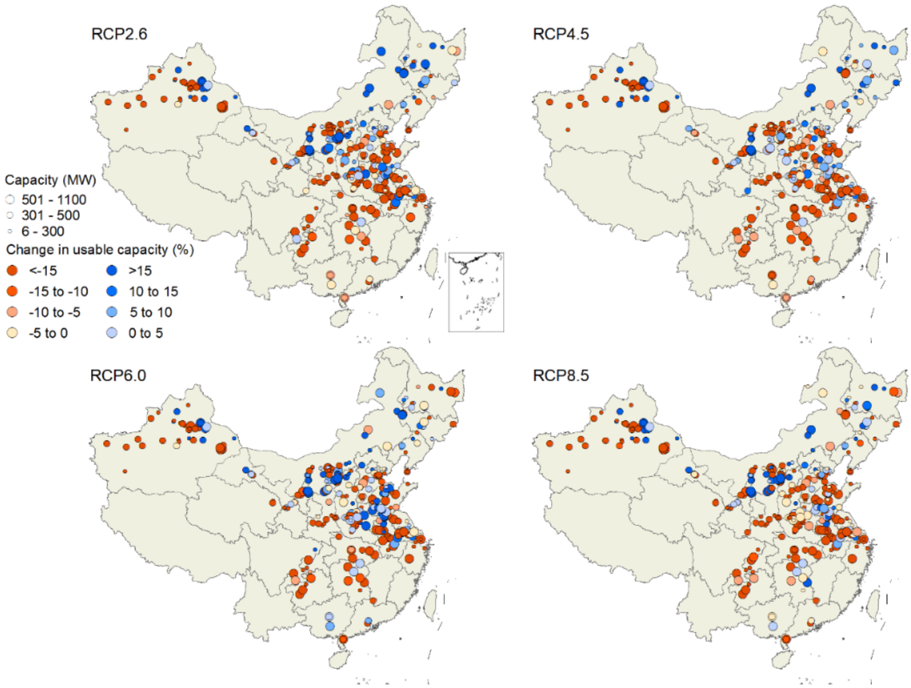


Figure S7.4.6 Impacts of climate and water resources change on usable capacity of CPUs in summer (June, July, August). The changes in the annual usable capacity under four climate scenarios in the 2030s compared to the reference period. Those that do not experience changes in usable capacity reductions are not shown on the maps.

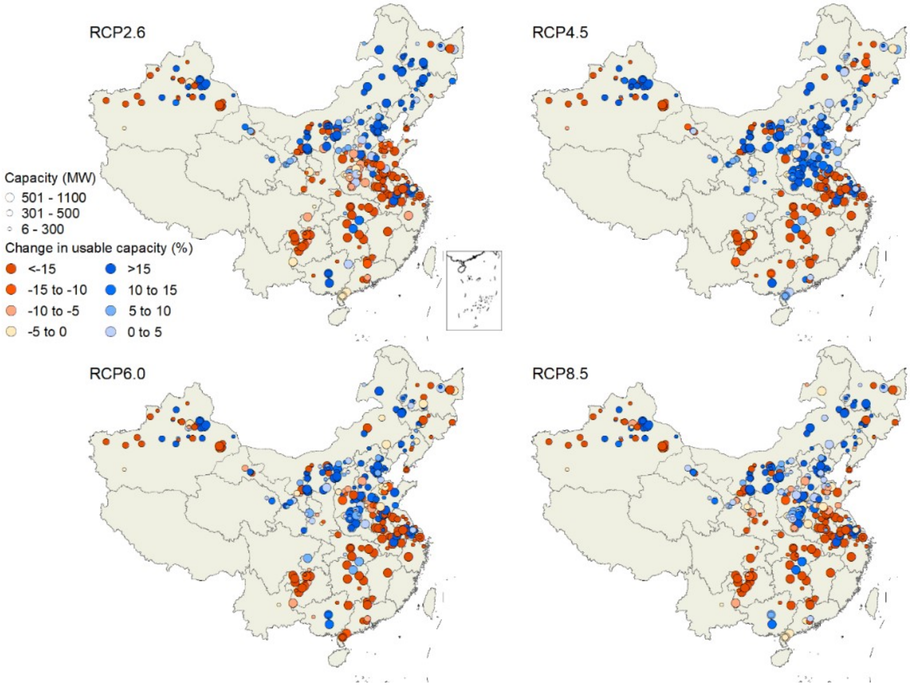


Figure S7.4.7 Impacts of climate and water resources change on usable capacity of CPUs in autumn (September, October, November). The changes in the annual usable capacity under four climate scenarios in the 2030s compared to the reference period. Those that do not experience changes in usable capacity reductions are not shown on the maps.

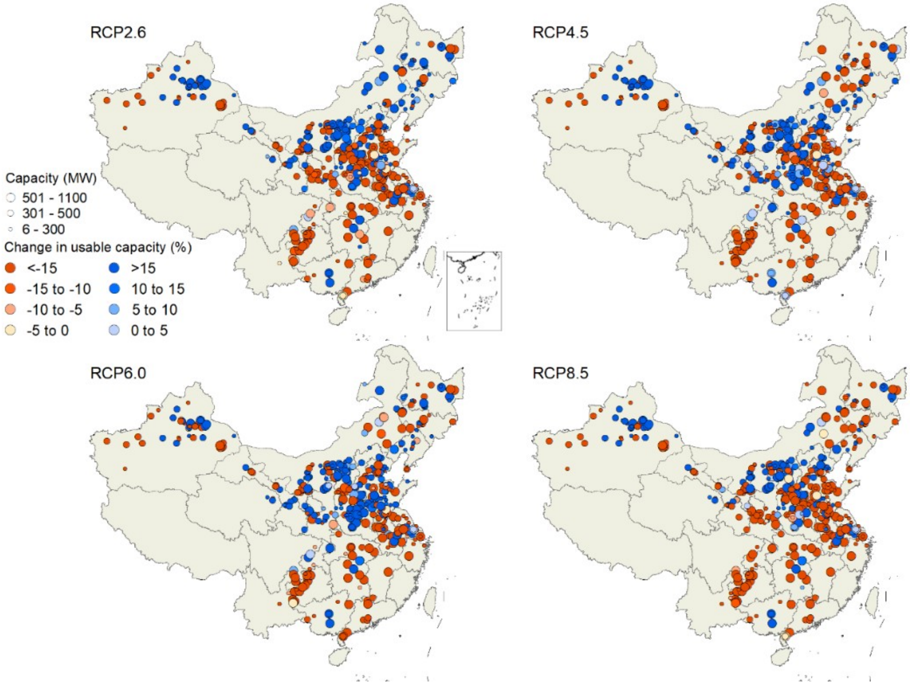


Figure S7.4.8 Impacts of climate and water resources change on usable capacity of CPUs in winter (December, January, February). The changes in the annual usable capacity under four climate scenarios in the 2030s compared to the reference period. Those that do not experience changes in usable capacity reductions are not shown on the maps.

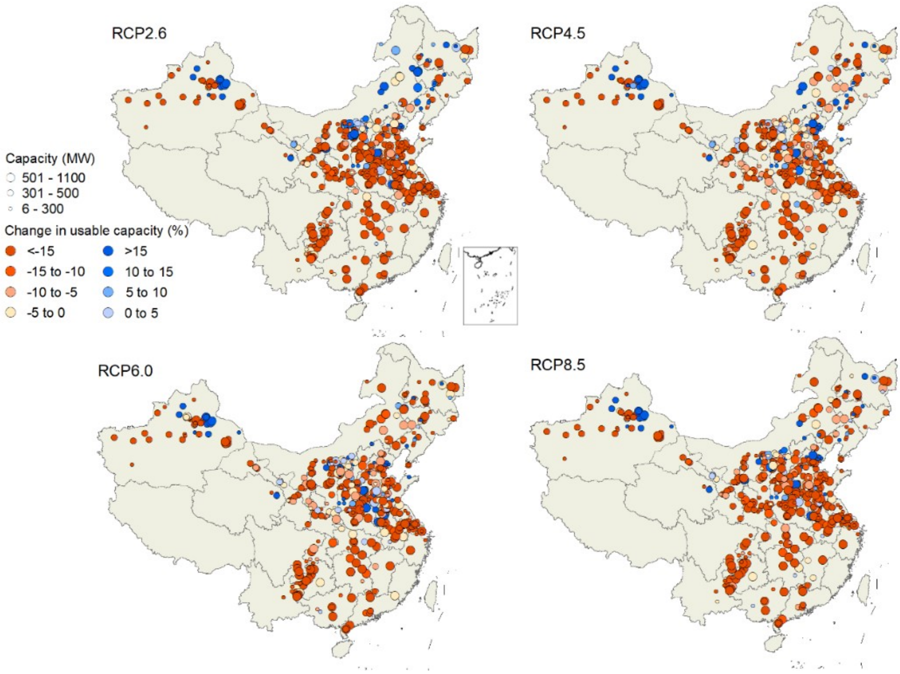


Figure S7.4.9 Impacts of climate and water resources change on annual usable capacity of CPUs with CCS. The changes in the annual usable capacity under four climate scenarios in the 2030s compared to the reference period. Those that do not experience changes in usable capacity reductions are not shown on the maps.

Table S7.4.1 The changes in provincial usable capacity under scenarios RCP2.6, RCP4.5, RCP6.0, and RCP8.5 for the 2030s relative to the reference period. Unit: MW

	RCP2.6	RCP4.5	RCP6.0	RCP8.5
Anhui	-85	-4	428	-141
Beijing	108	110	111	109
Chongqing	-147	-15	18	-359
Fujian	0	0	0	0
Gansu	42	266	520	-7
Guangdong	-512	-665	-567	-451
Guangxi	75	-136	-15	-57
Guizhou	-1921	-2974	-2846	-3573
Hebei	1048	1589	1250	614
Heilongjiang	1081	653	392	563
Henan	-540	1100	1586	-166
Hubei	-279	-95	-31	-529
Hunan	-54	-208	-121	15
Inner Mongolia	3797	2168	1530	2190
Jiangsu	346	754	908	334
Jiangxi	-457	-427	-472	-464
Jilin	312	157	-56	-81
Liaoning	213	397	107	-33
Ningxia	101	275	266	260
Qinghai	184	199	198	195
Shaanxi	-2047	-1273	-873	-2302
Shandong	-2820	15	1161	-2488
Shanghai	-16	-7	4	-23
Shanxi	-1021	242	689	-1055
Sichuan	-432	-315	-192	-978
Tianjin	85	690	-96	82
Xinjiang	4816	5417	3610	5030
Yunnan	-322	-967	-705	-1148
Zhejiang	-289	-477	-486	-492

Table S7.4.2 The changes in provincial usable capacity when CCS is implemented under scenarios RCP2.6, RCP4.5, RCP6.0, and RCP8.5 for the 2030s relative to the reference period. Unit: MW

	RCP2.6	RCP4.5	RCP6.0	RCP8.5
Anhui	-1495	-1269	-997	-1505
Beijing	57	58	56	52
Chongqing	-795	-696	-684	-963
Fujian	0	0	0	0
Gansu	-1010	-923	-699	-1098
Guangdong	-1338	-1578	-1444	-1362
Guangxi	-335	-494	-379	-372
Guizhou	-4405	-5327	-5136	-5708
Hebei	-588	-97	-440	-1019
Heilongjiang	-414	-860	-1047	-880
Henan	-4495	-2830	-2515	-4095
Hubei	-721	-576	-453	-932
Hunan	-852	-973	-886	-801
Inner Mongolia	-165	-2327	-2575	-1745
Jiangsu	-2857	-2524	-2250	-2815
Jiangxi	-494	-601	-539	-524
Jilin	-206	-335	-484	-491
Liaoning	-364	-206	-484	-583
Ningxia	-600	-404	-411	-485
Qinghai	107	115	115	109
Shaanxi	-3279	-2304	-2033	-3381
Shandong	-5343	-2744	-1817	-5161
Shanghai	-73	-64	-63	-79
Shanxi	-3445	-2270	-1721	-3646
Sichuan	-1181	-981	-784	-1727
Tianjin	-232	266	-354	-289
Xinjiang	2351	2513	829	2441
Yunnan	-1108	-1792	-1620	-1950
Zhejiang	-620	-758	-738	-732

References

(1) Esfahani, A.N., Moghaddam, N.B., Maleki, A., Nazemi, A. The knowledge map of energy security. Energy Rep. 2021,7,3570-89.

(2) Arderne, C., Zorn, C., Nicolas, C., Koks, E.E. Predictive mapping of the global power system using open data. Sci Data. 2020,7,19.

(3) Liu, J., Hull, V., Godfray, H.C.J., Tilman, D., Gleick, P., Hoff, H., Pahl-Wostl, C., Xu, Z., Chung, M.G., Sun, J., Li, S. Nexus approaches to global sustainable development. Nat Sustain. 2018,1,466-76.

(4) Huang, S.-W., Chung, Y.-F., Wu, T.-H. Analyzing the relationship between energy security performance and decoupling of economic growth from CO2 emissions for OECD countries. Renew Sustain Energy Rev. 2021,152.

(5) Chini, C.M., Djehdian, L.A., Lubega, W.N., Stillwell, A.S. Virtual water transfers of the US electric grid. Nat Energy. 2018,3,1115-23.

(6) Chu, S., Majumdar, A. Opportunities and challenges for a sustainable energy future. Nature. 2012,488,294-303.

(7) Xu, S. Governing the energy–water nexus in China: An analysis from the perspective of the science–policy interface. J Environ Manage. 2020,272,111119.

(8) Jin, Y., Behrens, P., Tukker, A., Scherer, L. The energy-water nexus of China's interprovincial and seasonal electric power transmission. Appl Energy. 2021,286,116493.

(9) Bleischwitz, R., Spataru, C., VanDeveer, S.D., Obersteiner, M., van der Voet, E., Johnson, C., Andrews-Speed, P., Boersma, T., Hoff, H., van Vuuren, D.P. Resource nexus perspectives towards the United Nations Sustainable Development Goals. Nat Sustain. 2018,1,737-43.

(10) Chini, C.M., Peer, R.A.M. The traded water footprint of global energy from 2010 to 2018. Sci Data. 2021,8,7.

(11) Zhou, Y., Ma, M., Gao, P., Xu, Q., Bi, J., Naren, T. Managing water resources from the energy - water nexus perspective under a changing climate: A case study of Jiangsu province, China. Energy Policy. 2019,126,380-90.

(12) Li, N., Chen, W. Energy-water nexus in China's energy bases: From the Paris agreement to the Well Below 2 Degrees target. Energy. 2019,166,277-86.

(13) Jin, Y., Tang, X., Feng, C., Höök, M. Energy and water conservation synergy in China: 2007–2012. Resour Conserv Recycl. 2017,127,206-15.

(14) Feng, C., Tang, X., Jin, Y., Höök, M. The role of energy-water nexus in water conservation at regional levels in China. J Clean Prod. 2019,210,298-308.

(15) Wang, S., Liu, Y., Chen, B. Multiregional input–output and ecological network analyses for regional energy–water nexus within China. Appl Energy. 2018,227,353-64.

(16) Jin, Y., Behrens, P., Tukker, A., Scherer, L. Water use of electricity technologies: A global meta-analysis. Renew Sustain Energy Rev. 2019,115,109391.

(17) Magagna, D., Medarac, H., Hidalgo González, I. Projected fresh water use from the European energy sector : disaggregated fresh water withdrawal and consumption in the EU up to 2050. Publications Office, European Commission. 2018.

(18) Ganguli, P., Kumar, D., Ganguly, A.R. US Power Production at Risk from Water Stress in a Changing Climate. Sci Rep. 2017,7,11983.

(19) Howells, M., Hermann, S., Welsch, M., Bazilian, M., Segerström, R., Alfstad, T., Gielen, D., Rogner, H., Fischer, G., van Velthuizen, H., Wiberg, D., Young, C., Roehrl, R.A., Mueller, A., Steduto, P., Ramma, I. Integrated analysis of climate change, land-use, energy and water strategies. Nature Climate Change. 2013,3,621-6.

(20) Webster, M., Donohoo, P., Palmintier, B. Water–CO2 trade-offs in electricity generation planning. Nature Climate Change. 2013,3,1029-32.

(21) International Energy Agency. World Energy Outlook 2021. 2021.

(22) Chai, L., Liao, X.W., Yang, L., Yan, X.L. Assessing life cycle water use and pollution of coal-fired power generation in China using input-output analysis. Applied Energy. 2018,231,951-8.

(23) Zhang, C., Zhong, L., Wang, J. Decoupling between water use and thermoelectric power generation growth in China. Nat Energy. 2018,3,792-9.

(24) Scherer, L., Pfister, S. Global water footprint assessment of hydropower. Renew Energy. 2016,99,711-20.

(25) Chu, S., Majumdar, A. Opportunities and challenges for a sustainable energy future. Nature. 2012,488,294.

(26) van Vliet, M.T.H., Wiberg, D., Leduc, S., Riahi, K. Power-generation system vulnerability and adaptation to changes in climate and water resources. Nature Climate Change. 2016,6,375.

(27) Verones, F., Hellweg, S., Antón, A., Azevedo, L.B., Chaudhary, A., Cosme, N., Cucurachi, S., Baan, L., Dong, Y., Fantke, P., Golsteijn, L., Hauschild, M., Heijungs, R., Jolliet, O., Juraske, R., Larsen, H., Laurent, A., Mutel, C.L., Margni, M., Núñez, M., Owsianiak, M., Pfister, S., Ponsioen, T., Preiss, P., Rosenbaum, R.K., Roy, P.O., Sala, S., Steinmann, Z., Zelm, R., Van Dingenen, R., Vieira, M., Huijbregts, M.A.J. LC-IMPACT: A regionalized life cycle damage assessment method J Ind Ecol. 2020,24,1201-19.

(28) D'Odorico, P. The global food–energy–water nexus. Rev Geophys. 2018,56.

(29) Dorber, M., Mattson, K.R., Sandlund, O.T., May, R., Verones, F. Quantifying net water consumption of Norwegian hydropower reservoirs and related aquatic biodiversity impacts in Life Cycle Assessment. Environ Impact Assess Rev. 2019,76,36-46.

(30) Chini, C.M., Stillwell, A.S. The changing virtual water trade network of the European electric grid. Appl Energy. 2020,260,114151.

(31) World Nuclear Association. Electricity Transmission Systems. 2020.

(32) Dang, Q., Lin, X., Konar, M. Agricultural virtual water flows within the United States. Water Resources Research. 2015,51,973-86.

(33) Xia, W., Chen, X., Song, C., Pérez-Carrera, A. Driving factors of virtual water in international grain trade: A study for belt and road countries. Agric Water Manage. 2022,262.

(34) Allan, J.A. Fortunately there are substitutes for water otherwise our hydro-political futures would be impossible. . Priorities for Water Resources Allocation and Management. 1993.

(35) Zhao, X., Liu, J., Liu, Q., Tillotson, M.R., Guan, D., Hubacek, K. Physical and virtual water transfers for regional water stress alleviation in China. Proc Natl Acad Sci U S A. 2015,112,1031.

(36) Zhao, X., Li, Y.P., Yang, H., Liu, W.F., Tillotson, M.R., Guan, D., Yi, Y., Wang, H. Measuring scarce water saving from interregional virtual water flows in China. Environ Res Lett. 2018,13,054012.

(37) International Energy Agency. Key World Energy Statistics 2021. 2021.

(38) CoalSwarm. Global Coal Plant Tracker. 2019.

(39) Zhang, C., Anadon, L.D. Life Cycle Water Use of Energy Production and Its Environmental Impacts in China. Environ Sci Technol. 2013,47,14459-67.

(40) Lv, H., Yang, L., Zhou, J., Zhang, X., Wu, W., Li, Y., Jiang, D. Water resource

synergy management in response to climate change in China: From the perspective of urban metabolism. Resour Conserv Recycl. 2020,163.

(41) Liang, X., Li, J., Guo, G., Li, S., Gong, Q. Evaluation for water resource system efficiency and influencing factors in western China: A two-stage network DEA-Tobit model. J Clean Prod. 2021,328.

(42) Liao, X., Hall, J.W., Eyre, N. Water use in China's thermoelectric power sector. Global Environmental Change. 2016,41,142-52.

(43) Yang, Y., Qu, S., Cai, B., Liang, S., Wang, Z., Wang, J., Xu, M. Mapping global carbon footprint in China. Nat Commun. 2020,11,2237.

(44) The State Council of the People's Republic of China. The speech made by Xi Jinping at the UN Biodiversity Summit. 2020.

(45) Mallapaty, S. How China could be carbon neutral by mid-century. Nature. 2020,586,482-3.

(46) Pöyry (Beijing) Engineering and Consulting Company Limited. The impacts of China's commitment to achieve carbon neutrality. 2020.

(47) Zhao, Z. Power generation companies are unveiling carbon reduction plans. China Energy News. 2021.

(48) China Electricity Council. Annual development report of China's power industry. 2020.

(49) Wu, C.B., Guan, P.B., Zhong, L.N., Lv, J., Hu, X.F., Huang, G.H., Li, C.C. An optimized low-carbon production planning model for power industry in coal-dependent regions - A case study of Shandong, China. Energy. 2020,192.

(50) The State Council of the People's Republic of China. Biodiversity Conservation in China. 2021.

(51) China Electricity Council. Annual compilation of statistics of power industry. 2018.

(52) Webster, M., Donohoo, P., Palmintier, B. Water-CO2 trade-offs in electricity generation planning. Nat Clim Change. 2013,3,1029-32.

(53) Behrens, P., van Vliet, M.T.H., Nanninga, T., Walsh, B., Rodrigues, J.F.D. Climate change and the vulnerability of electricity generation to water stress in the European Union. Nat Energy. 2017,2,17114.

(54) Hejazi, M.I., Voisin, N., Liu, L., Bramer, L.M., Fortin, D.C., Hathaway, J.E., Huang, M., Kyle, P., Leung, L.R., Li, H.-Y., Liu, Y., Patel, P.L., Pulsipher, T.C.,

Rice, J.S., Tesfa, T.K., Vernon, C.R., Zhou, Y. 21st century United States emissions mitigation could increase water stress more than the climate change it is mitigating. Proc Natl Acad Sci USA. 2015,112,10635.

(55) Peng, W., Wagner, F., Ramana, M.V., Zhai, H., Small, M.J., Dalin, C., Zhang, X., Mauzerall, D.L. Managing China's coal power plants to address multiple environmental objectives. Nat Sustain. 2018,1,693-701.

(56) Mouratiadou, I., Biewald, A., Pehl, M., Bonsch, M., Baumstark, L., Klein, D., Popp, A., Luderer, G., Kriegler, E. The impact of climate change mitigation on water demand for energy and food: An integrated analysis based on the Shared Socioeconomic Pathways. Environ Sci Policy. 2016,64,48-58.

(57) Beal, C.M., Archibald, I., Huntley, M.E., Greene, C.H., Johnson, Z.I. Integrating Algae with Bioenergy Carbon Capture and Storage (ABECCS) Increases Sustainability. Earths Future. 2018,6,524-42.

(58) Gerbens-Leenes, W., Hoekstra, A.Y., van der Meer, T.H. The water footprint of bioenergy. Proc Natl Acad Sci USA. 2009,106,10219-23.

(59) Hoekstra, A.Y. A critique on the water-scarcity weighted water footprint in LCA. Ecol Indic. 2016,66,564-73.

(60) Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M. The water footprint assessment manual: setting the global standard 2011.

(61) Raptis, C.E., Boucher, J.M., Pfister, S. Assessing the environmental impacts of freshwater thermal pollution from global power generation in LCA. Sci Total Environ. 2017,580,1014-26.

(62) Boulay, A.-M., Hoekstra, A.Y., Vionnet, S. Complementarities of Water-Focused Life Cycle Assessment and Water Footprint Assessment. Environ Sci Technol. 2013,47,11926-7.

(63) Macknick, J., Newmark, R., Heath, G., Hallett, K.C. Operational water consumption and withdrawal factors for electricity generating technologies: a review of existing literature. Environ Res Lett. 2012,7.

(64) Fthenakis, V., Kim, H.C. Life-cycle uses of water in US electricity generation. Renew Sustain Energy Rev. 2010,14,2039-48.

(65) Meldrum, J., Nettles-Anderson, S., Heath, G., Macknick, J. Life cycle water use for electricity generation: a review and harmonization of literature estimates. Environ Res Lett. 2013,8.

(66) Mekonnen, M.M., Gerbens-Leenes, P.W., Hoekstra, A.Y. The consumptive

water footprint of electricity and heat: a global assessment. Environ Sci-Wat Res. 2015,1,285-97.

(67) Pfister, S., Saner, D., Koehler, A. The environmental relevance of freshwater consumption in global power production. Int J of Life Cycle Ass. 2011,16,580-91.

(68) Spang, E.S., Moomaw, W.R., Gallagher, K.S., Kirshen, P.H., Marks, D.H. The water consumption of energy production: an international comparison. Environ Res Lett. 2014,9.

(69) Peer, R.A.M., Sanders, K.T. The water consequences of a transitioning US power sector. Appl Energy. 2018,210,613-22.

(70) Srinivasan, S., Kholod, N., Chaturvedi, V., Ghosh, P.P., Mathur, R., Clarke, L., Evans, M., Hejazi, M., Kanudia, A., Koti, P.N., Liu, B., Parikh, K.S., Ali, M.S., Sharma, K. Water for electricity in India: A multi-model study of future challenges and linkages to climate change mitigation. Appl Energy. 2018,210,673-84.

(71) Fulton, J., Cooley, H. The water footprint of California's energy system, 1990-2012. Environ Sci Technol. 2015,49,3314-21.

(72) Dodder, R.S., Barnwell, J.T., Yelverton, W.H. Scenarios for Low Carbon and Low Water Electric Power Plant Operations: Implications for Upstream Water Use. Environ Sci Technol. 2016,50,11460-70.

(73) Moher, D., Shamseer, L., Clarke, M., Ghersi, D., Liberati, A., Petticrew, M., Shekelle, P., Stewart, L.A., Group, P.-P. Preferred reporting items for systematic review and meta-analysis protocols (PRISMA-P) 2015 statement. Syst Rev. 2015,4,1.

(74) Pfister, S., Boulay, A.-M., Berger, M., Hadjikakou, M., Motoshita, M., Hess, T., Ridoutt, B., Weinzettel, J., Scherer, L., Döll, P., Manzardo, A., Núñez, M., Verones, F., Humbert, S., Buxmann, K., Harding, K., Benini, L., Oki, T., Finkbeiner, M., Henderson, A. Understanding the LCA and ISO water footprint: A response to Hoekstra (2016) "A critique on the water-scarcity weighted water footprint in LCA". Ecol Indic. 2017,72,352-9.

(75) Ali, B., Kumar, A. Development of water demand coefficients for power generation from renewable energy technologies. Energy Convers Manag. 2017,143,470-81.

(76) Carrillo, A.M.R., Frei, C. Water: A key resource in energy production. Energy Policy. 2009,37,4303-12.

(77) Liao, X.W., Hall, J.W., Eyre, N. Water use in China's thermoelectric power sector. Glob Environ Change. 2016,41,142-52.

(78) Tursi, A. A review on biomass: importance, chemistry, classification, and conversion. Biofuel Research Journal. 2019,6,962-79.

(79) Cai, B., Zhang, B., Bi, J., Zhang, W. Energy's thirst for water in China. Environ Sci Technol. 2014,48,11760-8.

(80) Eyer, J., Wichman, C.J. Does water scarcity shift the electricity generation mix toward fossil fuels? Empirical evidence from the United States. J Environ Econ Manag. 2018,87,224-41.

(81) Gerbens-Leenes, P.W., Hoekstra, A.Y., van der Meer, T. The water footprint of energy from biomass: A quantitative assessment and consequences of an increasing share of bio-energy in energy supply. Ecol Econ. 2009,68,1052-60.

(82) Xin, L., Feng, K., Siu, Y.L., Hubacek, K. Challenges faced when energy meets water: CO2 and water implications of power generation in inner Mongolia of China. Renew Sustain Energy Rev. 2015,45,419-30.

(83) Feng, K., Hubacek, K., Siu, Y.L., Li, X. The energy and water nexus in Chinese electricity production: A hybrid life cycle analysis. Renew Sustain Energy Rev. 2014,39,342-55.

(84) Clark, C.E., Harto, C.B., Schroeder, J.N., Martino, L.E., Horner, R.M. Life Cycle Water Consumption and Water Resource Assessment for Utility-Scale Geothermal Systems: An In-Depth Analysis of Historical and Forthcoming EGS Projects. United States: U. S. Department of Energy; 2013.

(85) Singh, S., Kumar, A. Development of water requirement factors for biomass conversion pathway. Bioresour Technol. 2011,102,1316-28.

(86) Dominguez-Faus, R., Powers, S.E., Burken, J.G., Alvarez, P.J. The Water Footprint of Biofuels: A Drink or Drive Issue. Environ Sci Technol. 2009,43,3005–10.

(87) Gerbens-Leenes, W., Hoekstra, A.Y. The water footprint of biofuel-based transport. Energy Environ Sci. 2011,4,2658-68.

(88) Mann, M.K., Spath, P.L. Life Cycle Assessment of a Biomass Gasification Combined-Cycle System. U. S.: National Renewable Energy Laboratory; 1997.

(89) Chang, Y., Huang, R.Z., Ries, R.J., Masanet, E. Life-cycle comparison of greenhouse gas emissions and water consumption for coal and shale gas fired power generation in China. Energy. 2015,86,335-43.

(90) Ali, B., Kumar, A. Development of life cycle water-demand coefficients for coal-based power generation technologies. Energy Convers Manag. 2015,90,247-

60.

(91) Ou, Y., Zhai, H.B., Rubin, E.S. Life cycle water use of coal- and natural-gas-fired power plants with and without carbon capture and storage. Int J Greenh Gas Con. 2016,44,249-61.

(92) Ali, B., Kumar, A. Development of life cycle water footprints for gas-fired power generation technologies. Energy Convers Manag. 2016,110,386-96.

(93) Poinssot, C., Bourg, S., Ouvrier, N., Combernoux, N., Rostaing, C., Vargas-Gonzalez, M., Bruno, J. Assessment of the environmental footprint of nuclear energy systems. Comparison between closed and open fuel cycles. Energy. 2014,69,199-211.

(94) Demeke, T.A., Marence, M., Mynett, A.E. Evaporation from reservoirs and the hydropower water footprint. International Conference and Exhibition on Water Storage and Hydropower Development for Africa. Ethiopia2013.

(95) Liu, J.G., Zhao, D.D., Gerbens-Leenes, P.W., Guan, D.B. China's rising hydropower demand challenges water sector. Sci Rep. 2015,5.

(96) Bakken, T.H., Modahl, I.S., Engeland, K., Raadal, H.L., Arnoy, S. The life-cycle water footprint of two hydropower projects in Norway. J Clean Prod. 2016,113,241-50.

(97) Herath, I., Deurer, M., Horne, D., Singh, R., Clothier, B. The water footprint of hydroelectricity: a methodological comparison from a case study in New Zealand. J Clean Prod. 2011,19,1582-9.

(98) Yuan, X., Lu, Y., Bi, X.J., Yuan, C., He, K.W., Dun, Y. A New Water Footprint Calculation Method for Hydroelectricity.(in Chinese). China Rural Water Hydropower. 2018,165-8.

(99) Peer, R.A.M., Sanders, K.T. Characterizing cooling water source and usage patterns across US thermoelectric power plants: a comprehensive assessment of self-reported cooling water data. Environ Res Lett. 2016,11,124030.

(100) Gao, J.J., Zhao, P., Zhang, H.W., Mao, G.Z., Wang, Y. Operational Water Withdrawal and Consumption Factors for Electricity Generation Technology in China-A Literature Review. Sustainability. 2018,10.

(101) Grubert, E.A., Beach, F.C., Webber, M.E. Can switching fuels save water? A life cycle quantification of freshwater consumption for Texas coal- and natural gas-fired electricity. Environ Res Lett. 2012,7.

(102) Williams, E.D., E., S.J. <BP-ESC-water-handbook.pdf>. BP International

Ltd; 2013.

(103) Diehl, T.H., Harris, M.A. Withdrawal and Consumption of Water by Thermoelectric Power Plants in the United States, 2010. U.S. Geological Survey; 2014.

(104) Bukhary, S., Ahmad, S., Batista, J. Analyzing land and water requirements for solar deployment in the Southwestern United States. Renew Sustain Energy Rev. 2018,82,3288-305.

(105) Stillwell, A.S., King, C.W., Webber, M.E., Duncan, I.J., Hardberger, A. Energy-Water Nexus in Texas. United States: The University of Texas at Austin; 2009.

(106) Mishra, G.S., Glassley, W.E., Yeh, S. Realizing the geothermal electricity potential-water use and consequences. Environ Res Lett. 2011,6.

(107) Wu, X.D., Ji, X., Li, C., Xia, X.H., Chen, G.Q. Water footprint of thermal power in China: Implications from the high amount of industrial water use by plant infrastructure of coal-fired generation system. Energy Policy. 2019,132,452-61.

(108) Byers, E.A., Hall, J.W., Amezaga, J.M. Electricity generation and cooling water use: UK pathways to 2050. Global Environmental Change. 2014,25,16-30.

(109) Ali, B. The cost of conserved water for coal power generation with carbon capture and storage in Alberta, Canada. Energy Convers Manag. 2018,158,387-99.

(110) Talati, S., Zhai, H.B., Kyle, G.P., Morgan, M.G., Patel, P., Liu, L. Consumptive Water Use from Electricity Generation in the Southwest under Alternative Climate, Technology, and Policy Futures. Environ Sci Technol. 2016,50,12095-104.

(111) Sharma, N., Mahapatra, S.S. A preliminary analysis of increase in water use with carbon capture and storage for Indian coal-fired power plants. Environmental Technology & Innovation. 2018,9,51-62.

(112) Chandel, M.K., Pratson, L.F., Jackson, R.B. The potential impacts of climate-change policy on freshwater use in thermoelectric power generation. Energy Policy. 2011,39,6234-42.

(113) Smith, P., Davis, S.J., Creutzig, F., Fuss, S., Minx, J., Gabrielle, B., Kato, E., Jackson, R.B., Cowie, A., Kriegler, E., van Vuuren, D.P., Rogelj, J., Ciais, P., Milne, J., Canadell, J.G., McCollum, D., Peters, G., Andrew, R., Krey, V., Shrestha, G., Friedlingstein, P., Gasser, T., Grübler, A., Heidug, W.K., Jonas, M., Jones, C.D., Kraxner, F., Littleton, E., Lowe, J., Moreira, J.R., Nakicenovic, N., Obersteiner, M., Patwardhan, A., Rogner, M., Rubin, E., Sharifi, A., Torvanger, A., Yamagata, Y.,

Edmonds, J., Yongsung, C. Biophysical and economic limits to negative CO2 emissions. Nat Clim Change. 2015,6,42.

(114) Ren, K., Tang, X., Jin, Y., Wang, J., Feng, C., Höök, M. Bi-objective optimization of water management in shale gas exploration with uncertainty: A case study from Sichuan, China. Resour Conserv Recycl. 2019,143,226-35.

(115) Guo, M., Lu, X., Nielsen, C.P., McElroy, M.B., Shi, W., Chen, Y., Xu, Y. Prospects for shale gas production in China: Implications for water demand. Renew Sustain Energy Rev. 2016,66,742-50.

(116) Clark, C.E., Horner, R.M., Harto, C.B. Life cycle water consumption for shale gas and conventional natural gas. Environ Sci Technol. 2013,47,11829-36.

(117) U.S. Energy Information Administration. Shale gas production. https://www.eia. gov/dnav/ng/ng_prod_shalegas_s1_a.htm; 2018, Accessed date: 20 May 2019.

(118) Liao, X., Chai, L., Jiang, Y., Ji, J., Zhao, X. Inter-provincial electricity transmissions' co-benefit of national water savings in China. J Clean Prod. 2019,229,350-7.

(119) Ehtiwesh, I.A.S., Da Silva, F.N., Sousa, A.C.M. Deployment of parabolic trough concentrated solar power plants in North Africa - a case study for Libya. International Journal of Green Energy. 2019,16,72-85.

(120) U.S. Energy Information Administration. Utility scale facility net generation from solar thermal. https://www.eia.gov/electricity/annual/html/epa_03_22.html. 2018.

(121) Wu, Z.Y., Hou, A.P., Chang, C., Huang, X., Shi, D.Q., Wang, Z.F. Environmental impacts of large-scale CSP plants in northwestern China. Environmental Science-Processes & Impacts. 2014,16,2432-41.

(122) The World Bank Group. Direct normal irradiation. https://globalsolaratlas.info/ downloads/china; 2017, Accessed date: 23 May 2019.

(123) International Energy Agency. Water-Energy Nexus. 2017.

(124) McMahon, J.E., Price, S.K. Water and Energy Interactions. Annu Rev Environ Resour. 2011,36,163-91.

(125) Guo, Y., Cao, Y.Q., Xian, T., Zhang, C.R. Brief analysis on water consumption index of inland nuclear power station. Water Resour Hydropower Eng. 2012,43,119-22.

(126) The food and agriculture organization. Country codes/names

http://www.fao.org/ countryprofiles/iso3list/en/; 2019, Accessed date: 22 May 2019.

(127) Li, L., Shan, Y., Lei, Y., Wu, S., Yu, X., Lin, X., Chen, Y. Decoupling of economic growth and emissions in China's cities: A case study of the Central Plains urban agglomeration. Appl Energy. 2019,244,36-45.

(128) Xie, P., Yang, F., Mu, Z., Gao, S. Influencing factors of the decoupling relationship between CO2 emission and economic development in China's power industry. Energy. 2020,209,118341.

(129) Chen, J., Wang, P., Cui, L., Huang, S., Song, M. Decomposition and decoupling analysis of CO2 emissions in OECD. Appl Energy. 2018,231,937-50.

(130) Scherer, L., de Koning, A., Tukker, A. BRIC and MINT countries' environmental impacts rising despite alleviative consumption patterns. Sci Total Environ. 2019,665,52-60.

(131) Liao, X.W., Zhao, X., Hall, J.W., Guan, D.B. Categorising virtual water transfers through China's electric power sector. Appl Energy. 2018,226,252-60.

(132) Yang, J., Chen, B. Energy-water nexus of wind power generation systems. Appl Energy. 2016,169,1-13.

(133) Coelho, C.D., da Silva, D.D., Sediyama, G.C., Moreira, M.C., Pereira, S.B., Lana, A.M.Q. Comparison of the water footprint of two hydropower plants in the Tocantins River Basin of Brazil. J Clean Prod. 2017,153,164-75.

(134) World Nuclear Association. Cooling power plants. 2019.

(135) Xu, X.H., Vignarooban, K., Xu, B., Hsu, K., Kannan, A.M. Prospects and problems of concentrating solar power technologies for power generation in the desert regions. Renew Sustain Energy Rev. 2016,53,1106-31.

(136) Zhai, H., Rubin, E.S., Versteeg, P.L. Water use at pulverized coal power plants with postcombustion carbon capture and storage. Environ Sci Technol. 2011,45,2479-85.

(137) Bosnjakovic, M., Tadijanovic, V. ENVIRONMENT IMPACT OF A CONCENTRATED SOLAR POWER PLANT. Tehnicki Glasnik-Technical Journal. 2019,13,68-74.

(138) Zhai, H., Rubin, E.S. Performance and cost of wet and dry cooling systems for pulverized coal power plants with and without carbon capture and storage. Energy Policy. 2010,38,5653-60.

(139) Zhang, C., Anadon, L.D., Mo, H.P., Zhao, Z.N., Liu, Z. Water-Carbon Trade-

off in China's Coal Power Industry. Environ Sci Technol. 2014,48,11082-9.

(140) Hogeboom, R.J., Knook, L., Hoekstra, A.Y. The blue water footprint of the world's artificial reservoirs for hydroelectricity, irrigation, residential and industrial water supply, flood protection, fishing and recreation. Adv in Water Resour. 2018,113,285-94.

(141) Zhao, D.D., Liu, J.G. A new approach to assessing the water footprint of hydroelectric power based on allocation of water footprints among reservoir ecosystem services. Phys Chem Earth. 2015,79-82,40-6.

(142) Mekonnen, M.M., Hoekstra, A.Y. The blue water footprint of electricity from hydropower. Hydrol Earth Syst Sci. 2012,16,179-87.

(143) Xie, X., Jiang, X., Zhang, T., Huang, Z. Regional water footprints assessment for hydroelectricity generation in China. Renew Energy. 2019,138,316-25.

(144) Lu, Y., He, D.M., He, K.W., Yuan, X. Analysis and calculation of water consumption of hydropower plants in Lancang-Mekong River basin.(in Chinese). Adv Water Sci. 2018,29,415-23.

(145) Ding, X., Tian, W., Chen, Q., Wei, G. Policies on water resources assessment of coastal nuclear power plants in China. Energy Policy. 2019,128,170-8.

(146) Operating Experience with Nuclear Power Stations in Member States. 2018.

(147) Crawford, R.H., Bontinck, P.-A., Stephan, A., Wiedmann, T., Yu, M. Hybrid life cycle inventory methods – A review. J Clean Prod. 2018,172,1273-88.

(148) Wolfram, P., Wiedmann, T., Diesendorf, M. Carbon footprint scenarios for renewable electricity in Australia. J Clean Prod. 2016,124,236-45.

(149) Lenzen, M. A guide for compiling inventories in hybrid life-cycle assessments: some Australian results. J Clean Prod. 2002,10,545-72.

(150) Suh, S. Functions, commodities and environmental impacts in an ecological–economic model. Ecol Econ. 2004,48,451-67.

(151) Joshi, S. Product Environmental Life-Cycle Assessment Using Input-Output Techniques. J Ind Ecol. 1999,3,95-120.

(152) Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinée, J., Heijungs, R., Hellweg, S., Koehler, A., Pennington, D., Suh, S. Recent developments in Life Cycle Assessment. J Environ Manag. 2009,91,1-21.

(153) Hertwich, E.G., Gibon, T., Bouman, E.A., Arvesen, A., Suh, S., Heath, G.A., Bergesen, J.D., Ramirez, A., Vega, M.I., Shi, L. Integrated life-cycle assessment of

electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. Proc Natl Acad Sci USA. 2015,112,6277-82.

(154) Zafrilla, J.E., Cadarso, M.-Á., Monsalve, F., de la Rúa, C. How Carbon-Friendly Is Nuclear Energy? A Hybrid MRIO-LCA Model of a Spanish Facility. Environ Sci Technol. 2014,48,14103-11.

(155) Bush, R., Jacques, D.A., Scott, K., Barrett, J. The carbon payback of micro-generation: An integrated hybrid input-output approach. Appl Energy. 2014,119,85-98.

(156) Wiedmann, T.O., Suh, S., Feng, K., Lenzen, M., Acquaye, A., Scott, K., Barrett, J.R. Application of Hybrid Life Cycle Approaches to Emerging Energy Technologies-The Case of Wind Power in the UK. Environ Sci Technol. 2011,45,5900-7.

(157) Suh, S., Huppes, G. Methods for Life Cycle Inventory of a product. J Clean Prod. 2005,13,687-97.

(158) Wu, X.D., Chen, G.Q. Energy and water nexus in power generation: The surprisingly high amount of industrial water use induced by solar power infrastructure in China. Appl Energy. 2017,195,125-36.

(159) Gu, A., Teng, F., Lv, Z. Exploring the nexus between water saving and energy conservation: Insights from industry sector during the 12th Five-Year Plan period in China. Renew Sustain Energy Rev. 2016,59,28-38.

(160) Tang, X., Jin, Y., Feng, C., McLellan, B.C. Optimizing the energy and water conservation synergy in China: 2007–2012. J Clean Prod. 2018,175,8-17.

(161) Li, X., Feng, K.S., Siu, Y.L., Hubacek, K. Energy-water nexus of wind power in China: The balancing act between CO2 emissions and water consumption. Energy Policy. 2012,45,440-8.

(162) Weber, C.L., Jaramillo, P., Marriott, J., Samaras, C. Life Cycle Assessment and Grid Electricity: What Do We Know and What Can We Know? Environ Sci Technol. 2010,44,1895-901.

(163) Bakken, T.H., Killingtveit, Å., Engeland, K., Alfredsen, K., Harby, A. Water consumption from hydropower plants – review of published estimates and an assessment of the concept. Hydrol Earth Syst Sci. 2013,17,3983-4000.

(164) Bakken, T.H., Modahl, I.S., Raadal, H.L., Bustos, A.A., Arnoy, S. Allocation of water consumption in multipurpose reservoirs. Water Policy. 2016,18,932-47.

(165) Mathioudakis, V., Gerbens-Leenes, P.W., Van der Meer, T.H., Hoekstra, A.Y.

The water footprint of second-generation bioenergy: A comparison of biomass feedstocks and conversion techniques. J Clean Prod. 2017,148,571-82.

(166) Pfister, S., Scherer, L. Uncertainty analysis of the environmental sustainability of biofuels. Energy, Sustainability and Society. 2015,5,30.

(167) Zhang, C., Zhong, L.J., Fu, X.T., Zhao, Z.N. Managing Scarce Water Resources in China's Coal Power Industry. Environ Manag. 2016,57,1188-203.

(168) Zhang, C., Zhong, L., Fu, X., Wang, J., Wu, Z. Revealing Water Stress by the Thermal Power Industry in China Based on a High Spatial Resolution Water Withdrawal and Consumption Inventory. Environ Sci Technol. 2016,50,1642-52.

(169) Ma, X.T., Yang, D.L., Shen, X.X., Zhai, Y.J., Zhang, R.R., Hong, J.L. How much water is required for coal power generation: An analysis of gray and blue water footprints. Sci Total Environ. 2018,636,547-57.

(170) The Water footprint assessment manual: Setting the global standard. 2011.

(171) Gao, X., Zhao, Y., Lu, S., Chen, Q., An, T., Han, X., Zhuo, L. Impact of coal power production on sustainable water resources management in the coal-fired power energy bases of Northern China. Appl Energy. 2019,250,821-33.

(172) Zhang, C., Zhong, L., Liang, S., Sanders, K.T., Wang, J., Xu, M. Virtual scarce water embodied in inter-provincial electricity transmission in China. Appl Energy. 2017,187,438-48.

(173) Mertens, J., Prieur-Vernat, A., Corbisier, D., Favrot, E., Boon, G. Water footprinting of electricity generated by combined cycle gas turbines using different cooling technologies: a practitioner's experience. J Clean Prod. 2015,86,201-8.

(174) Hoekstra, A.Y., Mekonnen, M.M., Chapagain, A.K., Mathews, R.E., Richter, B.D. Global Monthly Water Scarcity: Blue Water Footprints versus Blue Water Availability. PLoS One. 2012,7,e32688.

(175) Pfister, S., Koehler, A., Hellweg, S. Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environ Sci Technol. 2009,43,4098-104.

(176) Boulay, A.-M., Bulle, C., Bayart, J.-B., Deschênes, L., Margni, M. Regional Characterization of Freshwater Use in LCA: Modeling Direct Impacts on Human Health. Environ Sci Technol. 2011,45,8948-57.

(177) Boulay, A.-M., Bare, J., Benini, L., Berger, M., Lathuillière, M.J., Manzardo, A., Margni, M., Motoshita, M., Núñez, M., Pastor, A.V., Ridoutt, B., Oki, T., Worbe, S., Pfister, S. The WULCA consensus characterization model for water scarcity footprints: assessing impacts of water consumption based on available

water remaining (AWARE). Int J Life Cycle Ass. 2018,23,368-78.

(178) U.S. Energy Information Administration. Electric power annual. https://www.eia.gov/electricity/annual/; 2018, Accessed date: 29 May 2019.

(179) Nouri, N., Balali, F., Nasiri, A., Seifoddini, H., Otieno, W. Water withdrawal and consumption reduction for electrical energy generation systems. Appl Energy. 2019,248,196-206.

(180) Sood, A., Smakhtin, V. Global hydrological models: a review. Hydrological Sciences Journal. 2015,60,549-65.

(181) Li, X., Liu, J., Zheng, C., Han, G., Hoff, H. Energy for water utilization in China and policy implications for integrated planning. International Journal of Water Resources Development. 2016,32,477-94.

(182) Qin, Y., Curmi, E., Kopec, G.M., Allwood, J.M., Richards, K.S. China's energy-water nexus – assessment of the energy sector's compliance with the "3 Red Lines" industrial water policy. Energy Policy. 2015,82,131-43.

(183) Lu, Y., Jenkins, A., Ferrier, R.C., Bailey, M., Gordon, I.J., Song, S., Huang, J., Jia, S., Zhang, F., Liu, X., Feng, Z., Zhang, Z. Addressing China's grand challenge of achieving food security while ensuring environmental sustainability. Sci Adv. 2015,1,e1400039.

(184) Aghbashlo, M., Tabatabaei, M., Soltanian, S., Ghanavati, H. Biopower and biofertilizer production from organic municipal solid waste: An exergoenvironmental analysis. Renew Energy. 2019,143,64-76.

(185) Rocha, D.H.D., Silva, R.J. Exergoenvironmental analysis of a ultra-supercritical coal-fired power plant. J Clean Prod. 2019,231,671-82.

(186) Aghbashlo, M., Tabatabaei, M., Soltanian, S., Ghanavati, H., Dadak, A. Comprehensive exergoeconomic analysis of a municipal solid waste digestion plant equipped with a biogas genset. Waste Manag. 2019,87,485-98.

(187) Wang, J., Li, S., Zhang, G., Yang, Y. Performance investigation of a solar-assisted hybrid combined cooling, heating and power system based on energy, exergy, exergo-economic and exergo-environmental analyses. Energy Conversion and Management. 2019,196,227-41.

(188) Rosen, M.A. Environmental sustainability tools in the biofuel industry. Biofuel Research Journal. 2018,5,751-2.

(189) van Vliet, M.T.H., Flörke, M., Wada, Y. Quality matters for water scarcity. Nat Geosci. 2017,10,800.

(190) Li, D., Zhao, J., Govindaraju, R.S. Water benefits sharing under transboundary cooperation in the Lancang-Mekong River Basin. J Hydrol. 2019,577,123989.

(191) Xu, R., Hu, H., Tian, F., Li, C., Khan, M.Y.A. Projected climate change impacts on future streamflow of the Yarlung Tsangpo-Brahmaputra River. Global Planet Change. 2019,175,144-59.

(192) Center for Strategic Studies of CNA. Water Resource Competition in the Brahmaputra River Basin: China, India, and Bangladesh. 2016.

(193) Yang, Y.C.E., Wi, S., Ray, P.A., Brown, C.M., Khalil, A.F. The future nexus of the Brahmaputra River Basin: Climate, water, energy and food trajectories. Global Environmental Change. 2016,37,16-30.

(194) The CPC Central Committee's proposals for formulating the 14th Five-Year Plan (2021-2025) for National Economic and Social Development and the Long-Range Objectives Through the Year 2035. 2020.

(195) Rosa, L., Reimer, J.A., Went, M.S., D'Odorico, P. Hydrological limits to carbon capture and storage. Nature Sustainability. 2020.

(196) Mallapaty, S. How China could be carbon neutral by mid-century. Nature. 2020,482-3.

(197) Lee, U., Han, J., Elgowainy, A., Wang, M. Regional water consumption for hydro and thermal electricity generation in the United States. Appl Energy. 2018,210,661-72.

(198) Macknick, J., Newmark, R., Heath, G., Hallett, K.C. Operational water consumption and withdrawal factors for electricity generating technologies: a review of existing literature. Environ Res Lett. 2012,7,045802.

(199) Hofste, R., S. , Kuzma, S.W., Sutanudjaja, E.H. AQUEDUCT 3.0: UPDATED DECISION-RELEVANT GLOBAL WATER RISK INDICATORS. World Resources Institute. 2019.

(200) Zhang, X., Liu, J., Tang, Y., Zhao, X., Yang, H., Gerbens-Leenes, P.W., van Vliet, M.T.H., Yan, J. China's coal-fired power plants impose pressure on water resources. J Clean Prod. 2017,161,1171-9.

(201) National Bureau of Statistics of China. Water Supply and Water Use. 2019.

(202) Liao, X., Zhao, X., Hall, J.W., Guan, D. Categorising virtual water transfers through China's electric power sector. Appl Energy. 2018,226,252-60.

(203) Bakken, T.H., Killingtveit, Å., Alfredsen, K. The Water Footprint of

Hydropower Production—State of the Art and Methodological Challenges. Global Challenges. 2017,1,1600018.

(204) Liu, J., Zhao, D., Gerbens-Leenes, P.W., Guan, D. China’s rising hydropower demand challenges water sector. Sci Rep. 2015,5,11446.

(205) Zhu, X., Guo, R., Chen, B., Zhang, J., Hayat, T., Alsaedi, A. Embodiment of virtual water of power generation in the electric power system in China. Appl Energy. 2015,151,345-54.

(206) Wang, C., Wang, R., Hertwich, E., Liu, Y., Tong, F. Water scarcity risks mitigated or aggravated by the inter-regional electricity transmission across China. Appl Energy. 2019,238,413-22.

(207) He, G., Zhao, Y., Jiang, S., Zhu, Y., Li, H., Wang, L. Impact of virtual water transfer among electric sub-grids on China's water sustainable developments in 2016, 2030, and 2050. Journal of Cleaner Production. 2019,239,118056.

(208) Zhang, C., He, G., Zhang, Q., Liang, S., Zipper, S.C., Guo, R., Zhao, X., Zhong, L., Wang, J. The evolution of virtual water flows in China's electricity transmission network and its driving forces. Journal of Cleaner Production. 2020,242,118336.

(209) Zhang, Y., Hou, S., Chen, S., Long, H., Liu, J., Wang, J. Tracking flows and network dynamics of virtual water in electricity transmission across China. Renewable and Sustainable Energy Reviews. 2020,110475.

(210) Zhu, Y., Ke, J., Wang, J., Liu, H., Jiang, S., Blum, H., Zhao, Y., He, G., Meng, Y., Su, J. Water transfer and losses embodied in the West–East electricity transmission project in China. Appl Energy. 2020,275,115152.

(211) Guo, R., Zhu, X., Chen, B., Yue, Y. Ecological network analysis of the virtual water network within China’s electric power system during 2007–2012. Appl Energy. 2016,168,110-21.

(212) Liao, X., Zhao, X., Liu, W., Li, R., Wang, X., Wang, W., Tillotson, M.R. Comparing water footprint and water scarcity footprint of energy demand in China’s six megacities. Appl Energy. 2020,269,115137.

(213) Zhang, Y., Fang, J., Wang, S., Yao, H. Energy-water nexus in electricity trade network: A case study of interprovincial electricity trade in China. Appl Energy. 2020,257,113685.

(214) Zhang, Y., Wang, J., Zhang, L., Liu, J., Zheng, H., Fang, J., Hou, S., Chen, S. Optimization of China’s electric power sector targeting water stress and carbon emissions. Appl Energy. 2020,271,115221.

(215) Platts Energy InfoStore Utility Data Institute. World Electric Power Plants Database. 2018.

(216) Lehner, B., Liermann, C.R., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P., Endejan, M., Frenken, K., Magome, J., Nilsson, C., Robertson, J.C., Rödel, R., Sindorf, N., Wisser, D. High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. Frontiers in Ecology and the Environment. 2011,9,494-502.

(217) China Electricity Council. Materials of national energy efficiency benchmarking competition for thermal power units. 2018.

(218) National Bureau of Statistics of China. Regional power production. 2019.

(219) Editorial Board of China Power Yearbook. China Electric Power Yearbook. 2008-2018.

(220) Qu, S., Liang, S., Xu, M. CO2 Emissions Embodied in Interprovincial Electricity Transmissions in China. Environ Sci Technol. 2017,51,10893-902.

(221) Professional Knowledge Service System for Energy. Inter-provincial power transmission. 2019.

(222) China Electricity Council. Power Industry Statistical Information System. 2019.

(223) EARTHDATA. FLDAS Noah Land Surface Model L4 Global Monthly Anomaly 0.1 x 0.1 degree. 2017.

(224) Gassert, F., Luck, M., Landis, M., Reig, P., Shiao, T. Aqueduct Global Maps 2.1: Constructing Decision-Relevant Global Water Risk Indicators. Washington D.C. United States.: World Resources Institute; 2014.

(225) Ministry of Water Resources of the People's Republic of China. Water Resources Bulletin of China in 2020. 2021.

(226) Cai, B., Zhang, W., Hubacek, K., Feng, K., Li, Z., Liu, Y., Liu, Y. Drivers of virtual water flows on regional water scarcity in China. J Clean Prod. 2019,207,1112-22.

(227) Scherer, L., Pfister, S. Dealing with uncertainty in water scarcity footprints. Environ Res Lett. 2016,11,054008.

(228) Huang, J., Ridoutt, B.G., Sun, Z., Lan, K., Thorp, K.R., Wang, X., Yin, X., Huang, J., Chen, F., Scherer, L. Balancing food production within the planetary water boundary. Journal of Cleaner Production. 2020,253,119900.

(229) Pfister, S., Bayer, P. Monthly water stress: spatially and temporally explicit consumptive water footprint of global crop production. Journal of Cleaner Production. 2014,73,52-62.

(230) Zhou, F., Bo, Y., Ciais, P., Dumas, P., Tang, Q., Wang, X., Liu, J., Zheng, C., Polcher, J., Yin, Z., Guimberteau, M., Peng, S., Ottle, C., Zhao, X., Zhao, J., Tan, Q., Chen, L., Shen, H., Yang, H., Piao, S., Wang, H., Wada, Y. Deceleration of China's human water use and its key drivers. 2020,117,7702-11.

(231) National Development and Reform Commission. Requirements on the Planning and Construction of Coal Power Plants. 2004.

(232) National Energy Administration. Introduction to the grid-connected operation of renewable energy in 2018. 2019.

(233) Sharifzadeh, M., Hien, R.K.T., Shah, N. China's roadmap to low-carbon electricity and water: Disentangling greenhouse gas (GHG) emissions from electricity-water nexus via renewable wind and solar power generation, and carbon capture and storage. Appl Energy. 2019,235,31-42.

(234) China Electricity Council. Benchmarking management of energy efficiency of thermal power plants and benchmarking of energy efficiency of thermal power units. 2019.

(235) National Energy Administration. List of Twelve Electricity Transmission Corridors for Air Pollution Control to Be Approved by the National Energy Administration. 2014.

(236) Peng, W., Yuan, J.H., Zhao, Y., Lin, M.Y., Zhang, Q., Victor, D.G., Mauzerall, D.L. Air quality and climate benefits of long-distance electricity transmission in China. Environ Res Lett. 2017,12,10.

(237) Ministry of Natural Resources. National Seawater Utilization Report. 2020.

(238) Department of Energy. Transforming the Nation's Electricity System: The Second Installment ofthe Quadrennial Energy Review. 2017.

(239) EUREAU. EUREAU Statistics Overview on Water and Wastewater in Europe. 2009.

(240) Vassolo, S., Döll, P. Global-scale gridded estimates of thermoelectric power and manufacturing water use. Water Resources Research. 2005,41.

(241) van Vliet, M.T.H., Yearsley, J.R., Ludwig, F., Vögele, S., Lettenmaier, D.P., Kabat, P. Vulnerability of US and European electricity supply to climate change. Nature Climate Change. 2012,2,676.

(242) Peer, R.A.M., Grubert, E., Sanders, K.T. A regional assessment of the water embedded in the US electricity system. Environ Res Lett. 2019,14,084014.

(243) Standardization Administration. Environmental quality standards for surface water. 2002.

(244) Chabouni, N., Belarbi, Y., Benhassine, W. Electricity load dynamics, temperature and seasonality Nexus in Algeria. Energy. 2020,200,117513.

(245) Conevska, A., Urpelainen, J. Weathering electricity demand? Seasonal variation in electricity consumption among off-grid households in rural India. Energy Res Social Sci. 2020,65,101444.

(246) Do, L.P.C., Lin, K.-H., Molnár, P. Electricity consumption modelling: A case of Germany. Economic Modelling. 2016,55,92-101.

(247) Ferraz de Andrade Santos, J.A., de Jong, P., Alves da Costa, C., Torres, E.A. Combining wind and solar energy sources: Potential for hybrid power generation in Brazil. Utilities Policy. 2020,67,101084.

(248) Zhang, Q., Chen, W. Modeling China's interprovincial electricity transmission under low carbon transition. Applied Energy. 2020,279,115571.

(249) China electric power yearbook. 2018.

(250) Lèbre, É., Stringer, M., Svobodova, K., Owen, J.R., Kemp, D., Côte, C., Arratia-Solar, A., Valenta, R.K. The social and environmental complexities of extracting energy transition metals. Nat Commun. 2020,11,4823.

(251) Popescu, V.D., Munshaw, R.G., Shackelford, N., Montesino Pouzols, F., Dubman, E., Gibeau, P., Horne, M., Moilanen, A., Palen, W.J. Quantifying biodiversity trade-offs in the face of widespread renewable and unconventional energy development. Scientific Reports. 2020,10,7603.

(252) Wang, J., Zhong, H., Yang, Z., Wang, M., Kammen, D.M., Liu, Z., Ma, Z., Xia, Q., Kang, C. Exploring the trade-offs between electric heating policy and carbon mitigation in China. Nat Commun. 2020,11,6054.

(253) Mason, N., Ward, M., Watson, J.E.M., Venter, O., Runting, R.K. Global opportunities and challenges for transboundary conservation. Nat Ecol Evol. 2020,4,694-701.

(254) Harrison, P.A., Berry, P.M., Simpson, G., Haslett, J.R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, N., Geertsema, W., Lommelen, E., Meiresonne, L., Turkelboom, F. Linkages between biodiversity attributes and ecosystem services: A systematic review. Ecosyst Serv. 2014,9,191-

203.

(255) Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., Geschke, A. International trade drives biodiversity threats in developing nations. Nature. 2012,486,109.

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

(257) Holland, R.A., Scott, K.A., Flörke, M., Brown, G., Ewers, R.M., Farmer, E., Kapos, V., Muggeridge, A., Scharlemann, J.P.W., Taylor, G., Barrett, J., Eigenbrod, F. Global impacts of energy demand on the freshwater resources of nations. Proc Natl Acad Sci U S A. 2015,112,E6707.

(258) Vörösmarty, C.J., Rodríguez Osuna, V., Cak, A.D., Bhaduri, A., Bunn, S.E., Corsi, F., Gastelumendi, J., Green, P., Harrison, I., Lawford, R., Marcotullio, P.J., McClain, M., McDonald, R., McIntyre, P., Palmer, M., Robarts, R.D., Szöllösi-Nagy, A., Tessler, Z., Uhlenbrook, S. Ecosystem-based water security and the Sustainable Development Goals (SDGs). Ecohydrol Hydrobiol. 2018,18,317-33.

(259) Miara, A., Tarr, C., Spellman, R., Vörösmarty, C.J., Macknick, J.E. The power of efficiency: Optimizing environmental and social benefits through demand-side-management. Energy. 2014,76,502-12.

(260) Van Vliet, M.T., Wiberg, D., Leduc, S., Riahi, K. Power-generation system vulnerability and adaptation to changes in climate and water resources. Nat Clim Change. 2016,6,375.

(261) Lohrmann, A., Farfan, J., Caldera, U., Lohrmann, C., Breyer, C. Global scenarios for significant water use reduction in thermal power plants based on cooling water demand estimation using satellite imagery. Nat Energy. 2019,4,1040–8.

(262) Bakken, T.H., Killingtveit, A., Engeland, K., Alfredsen, K., Harby, A. Water consumption from hydropower plants - review of published estimates and an assessment of the concept. Hydrology and Earth System Sciences. 2013,17,3983-4000.

(263) Grubert, E.A. Water consumption from hydroelectricity in the United States. Adv Water Resour. 2016,96,88-94.

(264) National Bureau of Statistics of China. Electricity Balance Sheet. 2019.

(265) Madden, N., Lewis, A., Davis, M. Thermal effluent from the power sector: an analysis of once-through cooling system impacts on surface water temperature. Environ Res Lett. 2013,8,035006.

(266) Logan, L.H., Stillwell, A.S. Probabilistic assessment of aquatic species risk from thermoelectric power plant effluent: Incorporating biology into the energy-water nexus. Appl Energy. 2018,210,434-50.

(267) Raptis, C.E., Pfister, S. Global freshwater thermal emissions from steam-electric power plants with once-through cooling systems. Energy. 2016,97,46-57.

(268) Yang, L.P., Wang, Y.J., Liang, H., Wang, Y.H., Cui, Y.X., Chen, Y.F. Influence of thermal drainage from power plants on environment. Sci Technol Innov Herald. 2018,124-7.

(269) Xin-yu, W., Yi-chuan, W., Kun, Z., Yu-qin, D., Xiao-wei, X., Zhao-rong, S. Review of Impact Assessments of Thermal Discharges from Power Plants on Aquatic Biota. Journal of Hydroecology. 2018,39,1-10.

(270) Raptis, C.E., van Vliet, M.T.H., Pfister, S. Global thermal pollution of rivers from thermoelectric power plants. Environ Res Lett. 2016,11,9.

(271) Pfister, S., Suh, S. Environmental impacts of thermal emissions to freshwater: Spatially explicit fate and effect modeling for life cycle assessment and water footprinting. Int J Life Cycle Assess. 2015,20,927.

(272) Cheng, Y., Jiang, Y., Qu, J., Xue, Z., Wu, K. Heat effect of warm water discharge from power plant on fish: with different discharge arrangements. Thermal Power Generation. 2017,46,101-6.

(273) Wiedmann, T., Lenzen, M. Environmental and social footprints of international trade. Nat Geosci. 2018,11,314-21.

(274) Xenopoulos, M.A., Lodge, D.M., Alcamo, J., Märker, M., Schulze, K., Van Vuuren, D.P. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Global Change Biol. 2005,11,1557-64.

(275) Grosberg, R.K., Vermeij, G.J., Wainwright, P.C. Biodiversity in water and on land. Curr Biol. 2012,22,R900-3.

(276) Hanafiah, M.M., Xenopoulos, M.A., Pfister, S., Leuven, R.S.E.W., Huijbregts, M.A.J. Characterization Factors for Water Consumption and Greenhouse Gas Emissions Based on Freshwater Fish Species Extinction. Environ Sci Technol. 2011,45,5272-8.

(277) Tapio, P. Towards a theory of decoupling: degrees of decoupling in the EU

and the case of road traffic in Finland between 1970 and 2001. Transport Policy. 2005,12,137-51.

(278) Ang, B., Zhang, F., Choi, K. Factorizing changes in energy and environmental indicators through decomposition. Energy. 1998,23,489-95.

(279) Tang, X., Jin, Y., McLellan, B.C., Wang, J., Li, S. China's coal consumption declining—Impermanent or permanent? Resour Conserv Recycl. 2018,129,307-13.

(280) Ang, B.W. LMDI decomposition approach: A guide for implementation. Energy Policy. 2015,86,233-8.

(281) Ministry of Water Resources of the People's Republic of China. Lengths of major rivers in China. 2003.

(282) Ministry of Water Resources of the People's Republic of China. Bulletin of river sediment in China. 2020.

(283) Chinese Hydraulic Engineering Society. China Water Resources. 2016.

(284) Ministry of Water Resources of the People's Republic of China. China Water Statistical Yearbook. 2018.

(285) International Energy Agency. World Energy Outlook 2020. 2020.

(286) Standardization Administration. Norm of water intake Part 1: Electric power production. 2012.

(287) National Energy Administration. Water saving guideline for thermal power plant. 2018.

(288) Pan, L., Liu, Z., Zhang, B. Comprehensive analysis and related measures on current situation ofwater saving of thermal power generation in China. Electric Power (Chinese).50,158-63.

(289) Xu, H., Lei, Y., Lu, C., Xie, Y. Comparative analysis of direct and indirect air cooling system for thermal power unit selection. Air Pollution Control (Chinese). 2014,101-4.

(290) Kuipers, K.J.J., Hellweg, S., Verones, F. Potential Consequences of Regional Species Loss for Global Species Richness: A Quantitative Approach for Estimating Global Extinction Probabilities. Environ Sci Technol. 2019,53,4728-38.

(291) Byers, E.A., Hall, J.W., Amezaga, J.M. Electricity generation and cooling water use: UK pathways to 2050. Glob Environ Change. 2014,25,16-30.

(292) Rosa, L., Reimer, J.A., Went, M.S., D'Odorico, P. Hydrological limits to carbon capture and storage. Nat Sustain. 2020,658–66.

(293) Kati, V., Kassara, C., Vrontisi, Z., Moustakas, A. The biodiversity-wind energy-land use nexus in a global biodiversity hotspot. Sci Total Environ. 2021,768,144471.

(294) Wu, X., Shao, L., Chen, G., Han, M., Chi, Y., Yang, Q., Alhodaly, M., Wakeel, M. Unveiling land footprint of solar power: A pilot solar tower project in China. J Environ Manage. 2021,280,111741.

(295) Barbarossa, V., Schmitt, R.J.P., Huijbregts, M.A.J., Zarfl, C., King, H., Schipper, A.M. Impacts of current and future large dams on the geographic range connectivity of freshwater fish worldwide. Proc Natl Acad Sci U S A. 2020,117,3648-55.

(296) Heino, J., Paavola, R., Virtanen, R., Muotka, T. Searching for biodiversity indicators in running waters: do bryophytes, macroinvertebrates, and fish show congruent diversity patterns? Biodiversity & Conservation. 2005,14,415-28.

(297) Tendall, D.M., Hellweg, S., Pfister, S., Huijbregts, M.A.J., Gaillard, G. Impacts of River Water Consumption on Aquatic Biodiversity in Life Cycle Assessment—A Proposed Method, and a Case Study for Europe. Environ Sci Technol. 2014,48,3236-44.

(298) Sun, Z., Zhang, L., Duan, Z. Number and scale of hydropower station projects in China. China Water Resources. 2013,12-3.

(299) van Vliet, M.T.H., Yearsley, J.R., Franssen, W.H.P., Ludwig, F., Haddeland, I., Lettenmaier, D.P., Kabat, P. Coupled daily streamflow and water temperature modelling in large river basins. Hydrol Earth Syst Sci. 2012,16,4303-21.

(300) Miara, A., Macknick, J.E., Vörösmarty, C.J., Tidwell, V.C., Newmark, R., Fekete, B. Climate and water resource change impacts and adaptation potential for US power supply. Nature Climate Change. 2017,7,793.

(301) Zheng, X., Wang, C., Cai, W., Kummu, M., Varis, O. The vulnerability of thermoelectric power generation to water scarcity in China: Current status and future scenarios for power planning and climate change. Appl Energy. 2016,171,444-55.

(302) Liu, L., Hejazi, M., Li, H., Forman, B., Zhang, X. Vulnerability of US thermoelectric power generation to climate change when incorporating state-level environmental regulations. Nat Energy. 2017,2,17109.

(303) Zhang, C., Anadon, L.D., Mo, H., Zhao, Z., Liu, Z. Water–carbon trade-off in China’s coal power industry. Environ Sci Technol. 2014,48.

(304) Tang, B., Wu, Y., Yu, B., Li, R., Wang, X. Co-current analysis among

electricity-water-carbon for the power sector in China. Sci Total Environ. 2020,745,141005.

(305) Ministry of Science and Technology of the People's Republic of China. Roadmap for carbon capture, utilization and storage technology development in China (2019). 2019.

(306) Wu, X.D., Guo, J.L., Chen, G.Q. The striking amount of carbon emissions by the construction stage of coal-fired power generation system in China. Energy Policy. 2018,117,358-69.

(307) Fan, J.-L., Shen, S., Xu, M., Yang, Y., Yang, L., Zhang, X. Cost-benefit comparison of carbon capture, utilization, and storage retrofitted to different thermal power plants in China based on real options approach. Adv Clim Change Res. 2020.

(308) Liu, L., Hejazi, M., Iyer, G., Forman, B.A. Implications of water constraints on electricity capacity expansion in the United States. Nat Sustain. 2019,2,206-13.

(309) Bartos, M.D., Chester, M.V. Impacts of climate change on electric power supply in the Western United States. Nature Climate Change. 2015,5,748.

(310) Fan, J.-L., Wei, S., Yang, L., Wang, H., Zhong, P., Zhang, X. Comparison of the LCOE between coal-fired power plants with CCS and main low-carbon generation technologies: Evidence from China. Energy. 2019,176,143-55.

(311) Zhu, Y., Chen, M., Yang, Q., Alshwaikh, M.J.M., Zhou, H., Li, J., Liu, Z., Zhao, H., Zheng, C., Bartocci, P., Fantozzi, F. Life cycle water consumption for oxyfuel combustion power generation with carbon capture and storage. J Clean Prod. 2021,281.

(312) Zhao, X., Liao, X., Zhang, C., Zhang, X., Mao, G., Zhang, S., Tillotson, M.R. Unveiling the dynamic of water-electricity conflict within and beyond megacity boundary. J Environ Manage. 2021,286,112259.

(313) Sutanudjaja, E.H., van Beek, R., Wanders, N., Wada, Y., Bosmans, J.H.C., Drost, N., van der Ent, R.J., de Graaf, I.E.M., Hoch, J.M., de Jong, K., Karssenberg, D., López López, P., Peßenteiner, S., Schmitz, O., Straatsma, M.W., Vannametee, E., Wisser, D., Bierkens, M.F.P. PCR-GLOBWB 2: a 5 arcmin global hydrological and water resources model. Geosci Model Dev. 2018,11,2429-53.

(314) Ward, P.J., Winsemius, H.C., Kuzma, S., Bierkens, M.F.P. Aqueduct Floods Methdology. World Resources Institute. 2020.

(315) Han, Q., Tan, G., Fu, X., Yang, H., Li, X. Calculation methods of river environmental flow and their applications. Engineering Journal of Wuhan

University. 2018,51,189-97.

(316) MartinaFlörke. Domestic and industrial water uses of the past 60 years as a mirror of socio-economic development: A global simulation study. Global Environmental Change. 2013,23,144-56.

(317) Zeke H, P., G.P. Emissions – the ‘business as usual’ story is misleading. Nature. 2020,618-20.

(318) Rosa, L., Chiarelli, D.D., Rulli, M.C., Dell’Angelo, J., D’Odorico, P. Global agricultural economic water scarcity. Sci Adv. 2020,6,eaaz6031.

(319) Koch, H., Vögele, S. Dynamic modelling of water demand, water availability and adaptation strategies for power plants to global change. Ecol Econ. 2009,68,2031-9.

(320) Wang, Y. Vulnerability of existing and planned coal-fired power plants in developing Asia to changes in climate and water resources. Energy Environ Sci. 2019,12.

(321) Morris, J., Paltsev, S., Ku, A.Y. Impacts of China's emissions trading schemes on deployment of power generation with carbon capture and storage. Energy Econ. 2019,81,848-58.

(322) Hu, B., Zhai, H. The cost of carbon capture and storage for coal-fired power plants in China. Int J Greenhouse Gas Control. 2017,65,23-31.

(323) Meldrum, J., Nettles-Anderson, S., Heath, G., Macknick, J. Life cycle water use for electricity generation: a review and harmonization of literature estimates. Environ Res Lett. 2013,8,015031.

(324) Larsen, M.A.D., Petrovic, S., Engström, R.E., Drews, M., Liersch, S., Karlsson, K.B., Howells, M. Challenges of data availability: Analysing the water-energy nexus in electricity generation. Energy Strategy Reviews. 2019,26.

(325) National Energy Administration. Notice of the National Energy Administration on the Issue of the Risk Early Warning for 2023 Coal Power Planning and Construction 2020.

(326) Institute of Geographic Sciences and Natural Resources Research. Thirsty coal: Coal power base development and water resources research. 2012.

(327) Yang, J., Huang, X., Tang, Q. Satellite-derived river width and its spatiotemporal patterns in China during 1990–2015. Remote Sensing of Environment. 2020,247.

(328) Cui, R.Y., Hultman, N., Cui, D., McJeon, H., Yu, S., Edwards, M.R., Sen, A.,

Song, K., Bowman, C., Clarke, L., Kang, J., Lou, J., Yang, F., Yuan, J., Zhang, W., Zhu, M. A plant-by-plant strategy for high-ambition coal power phaseout in China. Nat Commun. 2021,12,1468.

(329) Huang, P.F., Wang, X.J., Zhang, Y., Wang, S.H., Wang, X.L., LI, D.Y. The State of the Art of Desalination Development and Research on Exploration Scheme in China. Journal of Salt Science and Chemical Industry. 2020,49,6-10.

(330) Zhang, Y., Li, J., Tian, Y., Deng, Y., Xie, K. Virtual water flow associated with interprovincial coal transfer in China: Impacts and suggestions for mitigation. J Clean Prod. 2021.

(331) The State Council of the People's Republic of China. The CPC Central Committee's proposals for formulating the 14th Five-Year Plan (2021-2025) for National Economic and Social Development and the Long-Range Objectives Through the Year 2035. 2020.

(332) Wen, W., Feng, C., Zhou, H., Zhang, L., Wu, X., Qi, J., Yang, X., Liang, Y. Critical provincial transmission sectors for carbon dioxide emissions in China. Renew Sustain Energy Rev. 2021,149.

(333) Zhao, X., Ma, X., Chen, B., Shang, Y., Song, M. Challenges toward carbon neutrality in China: Strategies and countermeasures. Resour Conserv Recycl. 2022,176.

(334) Fan, J.-L., Wei, S., Zhang, X., Yang, L. A comparison of the regional investment benefits of CCS retrofitting of coal-fired power plants and renewable power generation projects in China. Int J Greenhouse Gas Control. 2020,92.

(335) Schwalm, C.R., Glendon, S., Duffy, P.B. RCP8.5 tracks cumulative CO2 emissions. Proc Natl Acad Sci U S A. 2020,117,19656-7.

(336) National Energy Administration. Supervision reports on the investment results of ten typical power grid projects, such as Zhejiang-Fujian ultra-high voltage. 2018.

(337) Xu, W., Xie, Y., Cai, Y., Ji, L., Wang, B., Yang, Z. Environmentally-extended input-output and ecological network analysis for Energy-Water-CO2 metabolic system in China. Sci Total Environ. 2021,758,143931.

(338) Meng, F., Wang, D., Meng, X., Li, H., Liu, G., Yuan, Q., Hu, Y., Zhang, Y. Mapping urban energy–water–land nexus within a multiscale economy: A case study of four megacities in China. Energy. 2022,239.

(339) Feng, C., Qu, S., Jin, Y., Tang, X., Liang, S., Chiu, A.S.F., Xu, M. Uncovering urban food-energy-water nexus based on physical input-output

analysis: The case of the Detroit Metropolitan Area. Appl Energy. 2019,252.

(340) Notice of restricted fishing range and duration in key waters of the Yangtze River Basin. 2020.

(341) Ramjeawon, T. Life cycle assessment of electricity generation from bagasse in Mauritius. J Clean Prod. 2008,16,1727-34.

(342) Wang, C.Y., Li, P.G., Liu, Y. Investigation of water-energy-emission nexus of air pollution control of the coal-fired power industry: A case study of Beijing-Tianjin-Hebei region, China. Energy Policy. 2018,115,291-301.

(343) Yuan, J.H., Lei, Q., Xiong, M.P., Guo, J.S., Zhao, C.H. Scenario-Based Analysis on Water Resources Implication of Coal Power in Western China. Sustainability. 2014,6,7155-80.

(344) Burkhardt, J.J., Heath, G.A., Turchi, C.S. Life Cycle Assessment of a Parabolic Trough Concentrating Solar Power Plant and the Impacts of Key Design Alternatives. Environ Sci Technol. 2011,45,2457-64.

(345) Frisvold, G.B., Marquez, T. Water Requirements for Large-Scale Solar Energy Projects in the West. J Contemp Water Res Educ. 2013,151,106-16.

(346) Klein, S.J.W. Multi-Criteria Decision Analysis of Concentrated Solar Power with Thermal Energy Storage and Dry Cooling. Environ Sci Technol. 2013,47,13925-33.

(347) Klein, S.J.W., Rubin, E.S. Life cycle assessment of greenhouse gas emissions, water and land use for concentrated solar power plants with different energy backup systems. Energy Policy. 2013,63,935-50.

(348) Schroeder, J.N., Harto, C.B., Clark, C.E. Federal policy documentation and geothermal water consumption: Policy gaps and needs. Energy Policy. 2015,84,58-68.

(349) Schroeder, J.N., Harto, C.B., Horner, R.M., Clark, C.E. Geothermal Water Use: Life Cycle Water Consumption, Water Resource Assessment, and Water Policy Framework. United States: U. S. Department of Energy; 2014.

(350) Clark, C.E., Harto, C.B., Schroeder, J.N., Martino, L.E., Horner, R.M. Life Cycle Water Consumption and Water Resource Assessment for Utility-Scale Geothermal Systems: An In-Depth Analysis of Historical and Forthcoming EGS Projects. United States: U. S. Department of Energy; 2013a.

(351) Clark, C.E., Harto, C.B., Sullivan, J.L., Wang, M.Q. Water use in the development and operation of geothermal power plants. United States: U. S.

Department of Energy; 2011.

(352) Kotb, S.A., Abdelaal, M.M.Z. Analysis of the impact of introduction of nuclear power plants on energy characteristics and environment in Egypt. Electr Eng. 2018,100,285-92.

(353) Chen, Q.W., Qi, B.Q., Jiang, Q., Jin, D. Nuclear power development and water resources management. J Hohai Univ. 2010,4.

(354) Mielke, E. Water Consumption of Energy Resource Extraction, Processing, and Conversion. 2010.

(355) Peng, B. Analyzing the water use of biomass power generation project.(in Chinese). ZHI HUAI 2014,42-4.

(356) Clark, C.E., Horner, R.M., Harto, C.B. Life Cycle Water Consumption for Shale Gas and Conventional Natural Gas. Environ Sci Technol. 2013b,47,11829-36.

(357) Liao, X., Hall, J.W. Drivers of water use in China's electric power sector from 2000 to 2015. Environ Res Lett. 2018,13.

(358) Zhang, C., Zhong, L.J., Fu, X.T., Zhao, Z.N. Managing Scarce Water Resources in China's Coal Power Industry. Environ Manag. 2016b,57,1188-203.

(359) Bakken, T.H., Modahl, I.S., Raadal, H.L., Bustos, A.A., Arnoy, S. Allocation of water consumption in multipurpose reservoirs. Water Policy. 2016b,18,932-47.

(360) Yang, Q., Liang, J., Li, J., Yang, H., Chen, H. Life cycle water use of a biomass-based pyrolysis polygeneration system in China. Appl Energy. 2018,224,469-80.

(361) Lin, B., He, J. Is biomass power a good choice for governments in China? Renew Sustain Energy Rev. 2017,73,1218-30.

(362) Wu, Y., Sun, X., Lu, Z., Zhou, J., Xu, C. Optimal site selection of straw biomass power plant under 2-dimension uncertain linguistic environment. J Clean Prod. 2019,212,1179-92.

(363) McDonald, R.I., Olden, J.D., Opperman, J.J., Miller, W.M., Fargione, J., Revenga, C., Higgins, J.V., Powell, J. Energy, Water and Fish: Biodiversity Impacts of Energy-Sector Water Demand in the United States Depend on Efficiency and Policy Measures. PLoS One. 2012,7.

(364) Harte, J., El-Gasseir, M. Energy and Water. Science. 1978,199,623.

(365) Kania, J.J. Economics of coal transport by slurry pipeline versus unit train: A

case study. Energy Economics. 1984,6,131-8.

(366) Zhang, C., Anadon, L.D., Mo, H., Zhao, Z., Liu, Z. Water−Carbon Trade-off in China's Coal Power Industry. Environ Sci Technol. 2014,48,11082-9.

(367) Committee of Yearbook of China water resources. Yearbook of China water resources. 2014.

(368) Tibet academy of agricultural and animal husbandry sciences. Water resources of Tibet. 2020.

(369) Yan, S., Zeng, D., Liu, J., Wang, W., Xie, X. A Soft-sensor Method of Reheat Steam Flow Based on Simplified Heat-balance Equation. Chin Soc for Elec Eng. 2011,31,114-9.

(370) Cheng, F., Fan, S., Zhao, J. Improve Thermal Efficiency by Stabilizing Steam Parameters of Power Plant Boiler. Energy and Energy Conversion. 2013,71-4.

(371) Wanders, N., van Vliet, M.T.H., Wada, Y., Bierkens, M.F.P., van Beek, L.P.H.R. Global monthly water temperature dataset, derived from dynamical 1-D water-energy routing model (DynWat) at 10 km spatial resolution (1.1). 2018.

(372) Wei, X., Xia Binqiang, Yuzhu, W. Measuring Method of Hydroelectric Generator Efficiency Based on Infrared Thermal Imaging Technology. China Water Power & Electrification. 2018,6,12-8.

(373) Yan, Y., Hao, J. Consumption measurement and efficiency calculation for large-sized hydraulic generators based on calorimetric method. Hunnan Electric Power. 2014,34,8-11.

(374) Pfister, S., Suh, S. Environmental impacts of thermal emissions to freshwater: spatially explicit fate and effect modeling for life cycle assessment and water footprinting. The International Journal of Life Cycle Assessment. 2015,20,927-36.

(375) Pfister, S., Koehler, A., Hellweg, S. Assessing the Environmental Impacts of Freshwater Consumption in LCA. Environ Sci Technol. 2009,43,4098.

(376) Resource and Environment Science and Data Center. China's land use type. 2020.

Summary

Electric power production is a major driver of global water stress. With China's rapid increase in energy access and build-out of an electrified energy system, it is now the world's largest electricity producer and is seeing increasing levels of energy-related water stress. The electric power sector has become China's second-largest water user, after irrigation. This thesis investigates the water use of power production and its impacts on water stress across China. Given that the production-related impacts are often transferred via regional power transmission, this thesis also assesses the water embedded in China's national transmission system. The future availability of water for energy generation is set to be threatened not only by the increasing water use of all sectors but also by changes in the climate, raising both research and policy concerns. This thesis therefore puts forward the following overarching research question:

What are the impacts and challenges of water use of electric power production in China?

To answer this question, the thesis addresses four subquestions in four chapters.

First, a global meta-analysis investigates the question: *What are the water requirements of different electricity technologies and what is the availability of regionally specific data? (research subquestion 1)*

To further assess the water-electricity nexus, the remaining three subquestions specifically concern the impacts of power production on water resources and the impacts of water stress on power production:

How much water is required for power production in China and how much water is virtually transferred via power transmission? (research subquestion 2)

What are the impacts of power production on freshwater biodiversity in China? (research subquestion 3)

What are the changes in water stress and the consequent impacts on power production in the future, and how might future carbon capture and storage (CCS) requirements exacerbate water issues in China? (research subquestion 4)

To answer the first subquestion, Chapter 2 reviewed the literature on the water

requirements of power production and investigated the characteristics of water use and uncertainties in assessment. There are large differences in water use estimates across power types. Photovoltaics, wind power, and run-of-river hydropower consume relatively little water; concentrated solar power and geothermal power consume intermediate volumes of water; woody and herbaceous biomass and reservoir hydropower may consume considerable water resources. Fossil fuels consume very large amounts of water. However, water use can vary greatly across power plants of the same type, depending on many factors, such as the type of water use, operational efficiency, plant location, and so on. These impacts will change dramatically in the future, contingent upon the climate mitigation strategies chosen. While climate change mitigation via solar and wind power would reduce water stress in the power system, the retrofitting of carbon capture and storage to fossil plants would lead to increases in water stress. For example, the operational water consumption of thermal power plants increases by up to 81% if carbon capture and storage (CCS) is added. Uncertainties arose from inconsistent methodological choices and system boundary definitions among studies. The chapter also highlighted the key points that need to be improved in assessments. For example, clarity on water use type (consumption vs. withdrawal) and water sources (e.g., seawater vs. freshwater) is needed for future research. The review provided methodological and data support for answering the remaining questions.

Chapter 3 answered the second subquestion by assessing the water use of power production in China from the perspective of both water consumption and water withdrawal at the power plant level. China's power production withdrew 62.7 billion m^3 of freshwater in 2017, of which 13 billion m^3 was consumed. Overall, 6.2 billion m^3 of freshwater withdrawal and 2.1 billion m^3 of water consumption were virtually traded through the transmission system, with large variations throughout the year. A counterfactual scenario where a region does not import power but satisfies the local demand by producing power itself showed that if transmission does not take place, freshwater withdrawal increases but consumption is reduced. This was because, compared with the east of China, the west generally had a larger water consumption factor but a lower withdrawal factor. Water stress was more equally distributed across provinces through power transmission. This chapter provided an international perspective in terms of the application of methods and results. The methods can be applied to other countries if sufficient data on the power system and water use are available. The results for China, as one of the major energy users worldwide, can make an important contribution to a database of global energy-related water use.

Chapter 4 assessed the impacts on freshwater biodiversity caused by water use for power production in China, in light of the third subquestion. This included the consumption of freshwater and the thermal emissions to freshwater. The total biodiversity loss caused by water consumption and thermal pollution due to China's electricity generation increased by 45% during 2008-2017, while the biodiversity loss caused per unit of electricity generation decreased by 23%. Biodiversity loss from thermal pollution was 60% higher than that driven by freshwater consumption. Electricity transmission resulted in the shifting of biodiversity impacts across regions. The results showed that 15% of total biodiversity loss was embedded in transmission networks. In terms of electric power system drivers of biodiversity loss, the total generation was the main driving factor of the increase in loss (rather than shifts in generation type, for example). This chapter proposed a framework for assessing the freshwater biodiversity impacts of power production, which can be incorporated in electricity and energy planning to reduce the impacts on ecosystems.

Chapter 5 answered the fourth subquestion by exploring the water vulnerability of China's thermoelectric power fleet under climate change by developing a hydrology-electricity modelling framework at a monthly time step and a 5-arcmin spatial resolution of the river network. The results showed that 120-176 GW of capacity will be exposed to water scarcity for at least one additional month per year in the 2030s. In the absence of carbon capture and storage (CCS), the national usable capacity of thermoelectric power will increase slightly, due to slight mitigation of water stress in northern regions, where many plants are located. However, the addition of CCS – which requires more water – would significantly exacerbate water vulnerability, leading to further reductions of 7.4-7.7% in usable capacity. Testing several adaptation options for vulnerability mitigation revealed that early retirement of power plants was most effective, because this significantly reduced water requirements. Interregional power transmission also played an important mitigating role by shifting power production from water-scarce regions to water-abundant regions. The results demonstrated the importance of incorporating climate and water-scarcity changes in electricity planning. It is also important to take account of competition for water between the power sector and other users, and adaptation strategies from the perspective of both individual plants and the power system as a whole.

Based on the above studies, this thesis found several answers to the overarching research question. First, there were large differences in water use of electricity technologies and there was inconsistency in the methods and data used in previous studies. This can lead to substantial uncertainties in water use assessments. In China, large volumes of water were required by the energy system because of large-scale thermal and hydropower production. Due to virtual water transfer via power transmission, water stress was more equally distributed across the country. In addition, China saw increasing freshwater biodiversity loss caused by freshwater consumption and thermal emissions of power production. Power production faces challenges if its water demand cannot be met. In the future, thermal power plants in China would see significant reductions in usable capacity if retrofitted with CCS.

There are many areas on which future research can focus. Specifically, data availability could be improved because data for the two systems – energy and water – are not easily accessible, which limits the scope and transparency of studies. Further, research models of the water-electricity nexus could be improved in terms of several aspects, including water, electricity, and climate simulations. Additionally, the interaction of water demand from governments, industries, and households needs to be further quantified for a deeper understanding of the water-electricity nexus.

To conclude, the new knowledge generated in this thesis advances the understanding of: 1) the water requirements of various types of electricity technologies; 2) the impacts of power production on water resources and the related biodiversity systems; and 3) the impacts of water stress on power production and adaptation. Overall, this thesis provides insights into the impacts and challenges of water use of electric power generation, yields methodological and data support for connecting water and power systems both theoretically and in practice, and offers suggestions for policymakers on how to mitigate energy-water conflicts and support further research.

Samenvatting

De productie van elektrische energie is een belangrijke oorzaak van waterstress op globale schaal. Door de snelle toename het gebruik van energie in China en de opbouw van een voor een belangrijk deel op elektriciteit gebaseerd energiesysteem is het land nu de grootste elektriciteitsproducent ter wereld. Hierdoor neemt de waterstress als gevolg van energiegebruik toe. De elektriciteitssector is na irrigatie de grootste gebruiker van water in China geworden. Deze dissertatie onderzoekt het watergebruik voor elektriciteitsproductie en de gevolgen daarvan voor de waterstress in China. Aangezien elektriciteit vaak via transmissie tussen regio's wordt getransporteerd, vinden de effecten van de productie van elektriciteit vaak op andere locaties plaats dan waar elektriciteit wordt gebruikt. Daarom analyseert deze dissertatie ook welke rol China's elektriciteitstransmissiesysteem speelt in het 'virtueel transport' van waterstress tussen provincies. De beschikbaarheid van water voor energieopwekking zal in de toekomst worden beperkt door het toenemende watergebruik van alle sectoren, maar ook door veranderingen in het klimaat, wat diverse onderzoeks- en beleidsvragen oproept. Daarom wordt in dit proefschrift de volgende overkoepelende onderzoeksvraag gesteld:

Wat zijn de problemen en milieugevolgen gerelateerd aan watergebruik voor de productie van elektrische energie in China?

Daartoe worden in dit proefschrift in vier hoofdstukken vier deelvragen beantwoord.

Ten eerste wordt in een globale meta-analyse ingegaan op de vraag: wat is de waterbehoefte van verschillende technologieën voor het opwekken van elektriciteit, en zijn er specifiek per regio gegevens beschikbaar over die behoefte?

Om de nexus tussen water en elektriciteit verder te begrijpen, richt dit proefschrift zich hiernaast op nog een aantal vragen rond de effecten van elektriciteitsproductie op watergebruik en de effecten van waterstress op de elektriciteitsproductie. Deze vragen zijn:

Hoeveel water is er nodig voor de elektriciteitsproductie in China, en hoeveel 'virtueel' transport van water vindt plaats via de transmissie van elektriciteit? (deelvraag 2)

Wat zijn de effecten van energieproductie op de zoetwaterbiodiversiteit in China?

(deelvraag 3)

Wat zijn de veranderingen in de waterstress en de daaruit voortvloeiende gevolgen voor de elektriciteitsproductie in de toekomst en welke invloed heeft de toekomstige behoefte aan koolstofafvang en opslag ('Carbon Capture and Storage', CCS) op waterstress in China? (deelvraag 4)

Om de eerste deelvraag te beantwoorden, is in hoofdstuk 2 een literatuuranalyse verricht ten aanzien van watergebruik bij elektriciteitsproductie. De analyse ging ook in op de kenmerken van het watergebruik en onzekerheden in de omvang daarvan. Er zijn grote verschillen in watergebruik over de hele levenscyclus per kWh elektriciteitsproductie tussen verschillende productietechnieken. Fotovoltaïsche energie, windenergie en waterkracht uit doorstroomwaterkrachtcentrales gebruiken relatief weinig water; geconcentreerde zonne-energie en geothermische energie gebruiken een gemiddelde hoeveelheid water; terwijl het opwekken van elektriciteit met biomassa en waterkrachtcentrales gevoed door stuwmeren een hoog gebruik van water kent. Ook het opwekken van elektriciteit met fossiele brandstoffen kent een aanzienlijk watergebruik. Het watergebruik kan echter sterk variëren tussen elektriciteitscentrales van hetzelfde type. Factoren als het gebruik van het type water, de operationele efficiëntie, de locatie van de centrale, enz. spelen daarbij een belangrijke rol. Het patroon van watergebruik zal in de toekomst drastisch veranderen. Dit hangt samen met de ambitie om elektriciteitsproductie koolstofneutraal te maken. Klimaatmitigatie door de inzet van zonne- en windenergie kan waterstress door elektriciteitsproductie sterk verminderen. Maar het achteraf inbouwen van CCS in centrales die fossiele energie gebruiken leidt juist tot een toename van de waterstress. Het operationele waterverbruik van thermische centrales kan bijvoorbeeld tot 81% toenemen als CCS wordt toegepast. Onzekerheden in data over watergebruik zijn vaak het gevolg van inconsistente methodologische keuzes en verschillen in definitie in systeemgrenzen tussen verschillende studies. In het hoofdstuk worden dan ook de belangrijkste punten belicht die in analyses moeten worden verbeterd. Voor toekomstig onderzoek is het van belang specifiek te zijn ten aanzien van bijvoorbeeld het type water (bv. zeewater vs. zoet water) en het type watergebruik (onttrekking: hoeveelheid water die b.v. voor koeling uit een rivier wordt onttrokken, maar grotendeels weer kan worden teruggevoerd; versus consumptie of verbruik: de hoeveelheid water die echt verbruikt wordt en niet meer kan worden teruggevoerd, b.v. door verdamping in koeltorens). Het review leverde methodologische inzichten en gegevens op de behulpzaam waren bij de resterende deelvragen.

In hoofdstuk 3 wordt de tweede deelvraag beantwoord. Het hoofdstuk analyseert op het niveau van individuele elektriciteitscentrales in China zowel de wateronttrekking als het waterverbruik. De elektriciteitsproductie in China onttrok in 2017 62,7 miljard m^3 zoet water, waarvan 13 miljard m^3 werd verbruikt. In totaal werd 6,2 miljard m^3 onttrokken zoet water en 2,1 miljard m^3 verbruikt zoet water via het transmissiesysteem voor elektriciteit virtueel tussen provincies getransporteerd, met grote schommelingen gedurende het jaar. Een fictief scenario waarin elke regio zelf stroom voor eigen behoefte produceert en waarin geen import of export van stroom plaatsvindt, liet zien dat de totale onttrekking van zoet water zou toenemen maar de consumptie zou afnemen. Dit komt omdat het westen van China, in vergelijking met het oosten van het land, over het algemeen een grotere waterverbruiksfactor had, maar een lagere onttrekkingsfactor. De waterstress, die samenhangt met verbruik, was dankzij import en export van elektriciteit via het transmissienet gelijkmatiger verdeeld over de provincies. De analysemethode kan ook op andere landen worden toegepast als er voldoende gegevens over het elektriciteitssysteem en het watergebruik beschikbaar zijn. De resultaten voor China, als een van de grootste energiegebruikers ter wereld, kunnen een basis vormen voor een globale database ten aanzien van energiegerelateerd watergebruik.

In hoofdstuk 4 zijn met het oog op de derde deelvraag de effecten van watergebruik voor de elektriciteitsproductie in China op de zoetwaterbiodiversiteit beoordeeld. Hierbij werden het verbruik van zoet water en de thermische emissies naar zoet water meegenomen. Het totale biodiversiteitsverlies door waterverbruik en thermische verontreiniging van de elektriciteitsproductie in China is in de periode 2008-2017 met 45% toegenomen, terwijl het biodiversiteitsverlies per eenheid elektriciteitsproductie met 23% is afgenomen. Het biodiversiteitsverlies door thermische vervuiling was 60% groter dan dat door het verbruik van zoet water. Elektriciteitstransmissie resulteerde in een verschuiving van de biodiversiteitseffecten tussen de regio's. Uit de resultaten bleek dat 15% van het totale biodiversiteitsverlies virtueel via transmissie van elektriciteit was verplaatst. De groei van het verlies aan biodiversiteit tussen 2008 en 2017 werd vooral veroorzaakt door een verhoging van de totale opwekking van elektriciteit. Verschuivingen tussen het type opwekking had veel minder invloed. Kort gezegd biedt dit hoofdstuk een kader om de gevolgen van elektriciteitsopwekking voor de biodiversiteit in zoet water door te rekenen, dat gebruikt kan worden om bij het opstellen van plannen voor toekomstige elektriciteitsopwekking de gevolgen voor ecosystemen te beperken.

Hoofdstuk 5 beantwoordde de vierde deelvraag. Het hoofdstuk onderzocht de kwetsbaarheid van de thermische elektriciteitsopwekking in China wanneer klimaatverandering plaatsvindt. Voor dit doel werd een hydrologisch-elektriciteitsmodel ontwikkeld met een ruimtelijke resolutie van het rivierennetwerk van 5 boogminuten en dat per maand de waterbeschikbaarheid analyseerde. De resultaten toonden aan dat 120-176 GW aan capaciteit zal worden blootgesteld aan waterschaarste gedurende ten minste één extra maand per jaar na 2030. Indien geen koolstofafvang en -opslag (CCS) wordt geïnstalleerd, zal de nationale bruikbare capaciteit van thermo-elektrische centrales licht toenemen. De toevoeging van CCS - waarvoor meer water nodig is - zou de kwetsbaarheid voor waterstress echter aanzienlijk vergroten, wat zou leiden tot een verdere vermindering van de bruikbare capaciteit met 7,4-7,7%. Deze kwetsbaarheid kan worden verminderd door een aantal maatregelen. Hiervan is vervroegde uitdienstneming van fossiele elektriciteitscentrales het doeltreffendst, aangezien dit de behoefte aan (koel)water aanzienlijk vermindert. Interregionaal elektriciteitstransport speelt ook een belangrijke mitigerende rol, omdat dit het mogelijk maakt de elektriciteitsproductie te verschuiven van regio's met waterschaarste naar regio's die over voldoende water beschikken. Uit de resultaten blijkt dat het belangrijk is om bij de elektriciteitsplanning rekening te houden met klimaatverandering en de invloed hiervan op waterschaarste. Het is ook belangrijk rekening te houden met de groeiende vraag naar water van zowel de elektriciteitssector als andere gebruikers, en oplossingen ten aanzien van waterschaarste te zoeken vanuit zowel het perspectief van individuele centrales als het elektriciteitssysteem als geheel.

Op basis van de bovenstaande analyses zijn in deze dissertatie de volgende antwoorden gevonden op de overkoepelende onderzoeksvraag. Ten eerste bleken er grote verschillen te bestaan in het watergebruik van technologieën voor elektriciteitsopwekking en was er inconsistentie in de methoden en gegevens die in eerdere studies werden gebruikt. Dit kan leiden tot grote onzekerheden in analyses van watergebruik. In China zijn grote hoeveelheden water nodig in het energiesysteem omdat veel elektriciteit wordt opgewekt met fossiele energie en met waterkracht. De virtuele wateroverdracht via het transmissiesysteem voor elektriciteit bleek te leiden tot een gelijkmatiger verdeling van de waterstress over het land. Bovendien bleek in China een toenemend verlies aan zoetwaterbiodiversiteit op te treden als gevolg van het zoetwaterverbruik en de thermische emissies van de elektriciteitsproductie. De elektriciteitsproductie staat voor grote uitdagingen wanneer niet aan de vraag naar water kan worden voldaan. In de toekomst zou de bruikbare capaciteit van thermische centrales in China aanzienlijk afnemen indien zij met CCS worden uitgerust.

Er zijn veel gebieden waarop toekomstig onderzoek zich kan toespitsen. Met name de beschikbaarheid van gegevens kan worden verbeterd. Op dit moment zijn veel gegevens voor de twee systemen - energie en water – niet altijd makkelijk toegankelijk, wat de mogelijkheden voor onderzoek en de transparantie daarvan beperkt. Voorts kunnen modellen die zijn gericht op het onderzoeken van de water-elektriciteitsnexus op verschillende punten worden verbeterd, onder meer voor wat betreft het gebruik van water, de opwekking van elektriciteit en klimaatsimulaties. Bovendien moet de vraag naar water van overheden, industrieën en huishoudens verder worden gekwantificeerd om de implicaties hiervan voor de water-elektriciteitsnexus beter te begrijpen.

Concluderend kan worden gesteld dat de kennis die in dit proefschrift bijeen is gebracht, leidt tot een beter begrip van: 1) de waterbehoefte van verschillende technologieën voor elektriciteitsopwekking; 2) de effecten van elektriciteitsproductie op watervoorraden en de daarmee samenhangende effecten op biodiversiteit; en, 3) de effecten van waterstress op elektriciteitsproductie. Samenvattend geeft deze dissertatie inzicht in de gevolgen van watergebruik bij elektriciteitsopwekking en omgekeerd, biedt het ondersteuning ten aanzien van data en methoden voor het analyseren van de relatie tussen water- en elektriciteitssystemen, en biedt het suggesties voor beleidsmakers over hoe problemen in de energie-waternexus op te lossen en dit met verder onderzoek te ondersteunen.

Acknowledgements

Time flies, and it is like a dream that my PhD journey is about to come to an end. The experience of living and studying in the CML family has been both enjoyable and interesting. I expanded my horizons and acquired a lot of new knowledge and skills during the journey. I would like to express profound thanks to my co-supervisors, Dr. Paul Behrens and Dr. Laura Scherer. They spent much time and energy on every word of the thesis. Their patient guidance, valuable suggestions, encouragement, and trust helped me to complete this thesis successfully. I would like to thank my main supervisor (*promotor*), Prof. Arnold Tukker, for giving me the opportunity to work at CML. His feedback was always very useful to my study. I would also like to thank my MSc supervisor, Xu Tang, a professor at the China University of Petroleum-Beijing, who helped me to take the first step in this research field and gave me the confidence to take the further step of undertaking a PhD study. I wish to thank the China Scholarship Council (CSC) for their financial support (grant no. 201806440027). I also wish to express my sincere gratitude to the Doctorate Committee, and to thank them for reviewing my thesis and giving comments.

During my PhD, I encountered difficulties and challenges in my life that I had never experienced before. Leiden witnessed my first step in many aspects: the first time of renting an apartment, cooking, and facing loneliness. I thank my officemates, Glenn Aguilar Hernandez, Carlos Siguenza Sanchez, and Qi Chen, for being so helpful. They are all very kind. Special thanks to my Chinese friends, Yujia Zhai, Zhongxiao Sun, Qi Yu, Juan Wu, Di Dong, Jianhong Zhou, Xining Yang, Xiaoyang Zhong, Yanan Liang, and Zhenyang Chen.

I would like to thank everyone who accompanied me on the PhD journey. I wish to express my sincere appreciation to my parents and family for their encouragement, unconditional love, and support for every decision I have ever made.

Curriculum Vitae

Yi Jin was born in 1992 in Suzhou, Jiangsu province, China. He graduated from Wuxian Senior High School in 2011. He majored in Energy Economics and obtained his bachelor's degree from Jiangsu University in 2015. Specializing in Management Science and Engineering, he obtained his master’s degree from the China University of Petroleum-Beijing. He joined the Department of Industrial Ecology, Institute of Environmental Sciences, Leiden University in 2018, with the support of the China Scholarship Council. His research topic is the nexus between energy and water systems.

Publication List

Jin, Y., Behrens, P., Tukker, A., Scherer, L. Biodiversity loss from freshwater use for China's electricity generation. *Environmental Science & Technology.* 2022, 56, 5, 3277-3287.

Jin, Y., Scherer, L., Sutanudjaja, E.H., Tukker, A., Behrens, P. Climate change and CCS increase the water vulnerability of China's thermoelectric power fleet. *Energy*. 2022,245,123339.

Jin, Y., Behrens, P., Tukker, A., Scherer, L. The energy-water nexus of China's interprovincial and seasonal electric power transmission. *Applied Energy*. 2021,286,116493.

Jin, Y., Behrens, P., Tukker, A., Scherer, L. Water use of electricity technologies: A global meta-analysis. *Renewable and Sustainable Energy Reviews*. 2019,115,109391.