# Decoupling climate impact of wastewater infrastructure and water stress alleviation across 300 cities in China is challenging yet plausible by 2030

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**Article**

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# Decoupling climate impact of wastewater infrastructure and water stress alleviation across 300 cities in China is challenging yet plausible by 2030

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## Abstract

Urban wastewater treatment and reuse are crucial for achieving water sustainability. Yet, pathways of realizing water–climate synergies in the planning of wastewater infrastructure remain unclear. In this paper, we examine the nexus of urban water stress and climate impact resulting from the expansion of wastewater infrastructures across over 300 cities in China. We demonstrate how the effect of alleviating urban water stress from wastewater treatment and reclaimed water reuse has been highly uneven across cities, costing the country a 183% increase in life-cycle greenhouse gas emissions during 2006–2015. Decoupling the climate impact of wastewater infrastructure from significant water stress is plausible by 2030, potentially leading to up to 35% reduction of greenhouse gas emissions. However, given the growing demand for water reuse and investment in the diffusion of low-carbon technologies at each life-cycle stage of wastewater systems, this decoupling would be a challenging endeavor.

**Keywords**: wastewater treatment; water reuse; urban water stress; life-cycle climate impact; water–climate nexus

# INTRODUCTION

Well-functioning wastewater treatment systems are crucial infrastructure for improving water sustainability and accessibility by replenishing water supplies and reusing reclaimed water[1–4]. However, given the increasingly higher standards for water quality and stringent environmental regulations, wastewater treatment and reuse have led to a significant growth in energy use and greenhouse gas (GHG) emissions[2,5–7]. In a rapidly urbanizing society with a mounting demand for freshwater, it is difficult to expand the wastewater infrastructure of cities to reduce local water stress (water stress means that the demand for water exceeds the sustainable supply of water) while avoiding additional climatic impacts[8–10]. Previous studies have revealed that the GHG emissions related to wastewater treatment systems would be more significant if the life-cycle contributions from upstream energy use, construction of infrastructure, on-site energy use and treatment processes, and the sludge disposal stage were included[11–14]. Improving the accounting and planning of urban wastewater infrastructure throughout their life cycles therefore, can be essential in achieving a positive synergy between two key Sustainable Development Goals (SDGs), i.e., *Clean Water and Sanitation (SDG 6)* and *Climate Action (SDG 13)*[15].

Over 300 cities in China experience water stress to varying degrees. This is because the growth rate of water demand during urbanization frequently exceeds the provision of freshwater resources which are in turn increasingly under stress due to climate change[8,16,17]. Meanwhile, to match the growing need for waste water treatment plant (WWTP), construction of urban WWTPs in China have increased from 976 plants in 2006 to over 6, 276 in 2015. By contrast, water reuse in China remains in its infancy, only accounting for ~10% of the total volume of treated wastewater in 2015[18]. This proportion is lower than that achieved approximately a decade ago by other advanced countries in water management, e.g., Germany (20%) and Israel (70%)[19]. To further address the threat of water shortages and to promote water availability and equity, in 2015, the Chinese government initiated a series of water management policies. First, these policies were designated to raise the country's effluent quality standard to match the quality for potable surface waters, e.g., lakes, rivers, and reservoirs[20]. Furthermore, they set a path of increasing the water reuse rate in cities through new investments in infrastructure, especially in rapidly-growing cities and regions facing severe water stress[21].

While China aims to improve water availability and quality for its population by building more wastewater treatment infrastructure, it also aims to achieve the goal of peak carbon emissions by 2030 and carbon neutrality by 2060. In this light, the projected expansion of wastewater infrastructure in China have raised concerns over their potential to exacerbate climate change challenges[11,14,22]. Studies have revealed costly direct and indirect GHG emissions resulting from the energy requirements of powering the systems, producing and transporting the input material, and biochemical processes for wastewater treatment plants[11,13,14,23]. Furthermore, alleviating water shortages by reusing larger amounts of water could result in more energy- and carbon-intensive resource inputs[7,10,24,25]. Despite these studies, it is unclear to what extent advanced treatment procedures and technologies can offset the climate impact of WWTPs throughout their life cycles while alleviating urban water stress[26–30].

In this study, we unravel the water-climate nexus from the perspective of the life-cycle of wastewater infrastructure. Towards this end, we quantify the life-cycle climate impacts, i.e., direct and indirect GHG emissions, of all urban WWTPs using an input–output-based hybrid life-cycle analysis (IO-LCA) from 2006 to 2015. We map water stress alleviation in over 300 cities across China by matching detailed plant-level treatment and reuse data with city-scale water resource and consumption data. Through this mapping, we demonstrate the extent in which water stress alleviation of WWTPs was coupled with climate impacts. To identify pathways of water-climate synergies by 2030, we examine the potential of enhancing water stress alleviation while mitigating climate impacts under various scenarios. These scenarios consider the diffusion and integration of low-carbon technologies using a newly proposed indicator termed as the additional climate impact for water stress alleviation (CIWSA; the magnitude of life-cycle climate impact divided by the change of water stress related to WWTPs). Our results reveal a very challenging yet plausible scenario to decouple water stress alleviation from an increase in climate impacts of wastewater infrastructure by 2030. This scenario necessitates a wider implementation of existing low-carbon technologies throughout the life cycles of wastewater infrastructure across China's cities.

# RESULTS

## Uneven climate impacts of wastewater infrastructures across cities

The mapping of climate impacts attributable to urban WWTPs in China (Fig. 1) reveal, from a life-cycle perspective, that the GHG emissions (represented by $CO_2$e) increased by 183% during 2006–2015 (from 45 to 129 Mt $CO_2$e). This was mainly as result of WWTPs expanding by 543% (from 976 to 6, 276 plants) and the amount of wastewater treated rising by 309% (Figs. S1 and S2). Although there was a slowdown in the construction of new treatment plants during 2011–2015, the total life-cycle climate impact increased by 103% during the 2006–2010 period (Fig. S3a). The combination of upstream and downstream activities such as plant construction

(35%, including raw material extraction) and sludge disposal (22%) showed a bigger contribution to the total climate impact of WWTPs than on-site wastewater treatment (40%) in 2015 (Fig. S3a). In addition, although a much smaller climate impact was generated at the water reuse stage (with a share of 3% in 2015), owing to the rapid increase of investment in water recycling infrastructure, this amount was double the amount in 2006. Despite the total increase, the climate impact intensity of urban WWTPs (represented by the life-cycle GHG emissions per unit of wastewater treated) in China has decreased by 61% on average (with a marked reduction from 3.1 kg $CO_2e/m^3$ in 2006 to 1.2 kg $CO_2e/m^3$ in 2015) (Fig. S3b). The climate impact intensity of water reuse has also shown an overall trend of rapid decrease (by 28%) during the same time period. By comparison, the climate impact intensity of sludge disposal has decreased slowly from 2006 to 2015 (by 21%). The lower impact intensities were attributable not only to an increase in the operational load rates of plants that better matched their designed capacities, but also to the adoption of more energy-efficient technologies resulting from stricter environmental and climate policies.

The challenge in reducing WWTP-related climate impact was quite unevenly shared, i.e., it was highly concentrated in fast-developing and densely-populated cities (Fig. 1a). For example, Beijing, Shanghai, Shenzhen, Chongqing, and Chengdu were among the biggest and fastest-growing emitters of WWTP-related GHGs in 2015, accounting for 1.7–3.2 Mt $CO_2e$ – almost double the amount in 2006. However, the major contributing factors to emissions varied greatly across cities (Fig. S4). For Shenzhen and Chengdu, direct and indirect emissions of $CH_4$, $N_2O$, and $CO_2$ from wastewater treatment played the major role in emissions (accounting for 51–55% of the total impact in 2015). For Beijing, indirect $CO_2$ emissions related to the building of water reuse facilities was an important source of its climate impact (53% in 2015), whereas for Chongqing and Shanghai, $CH_4$, $N_2O$, and $CO_2$ emissions related to sludge disposal (34–38%) were the main contributors to their climatic impact.

The alleviation of water stress in cities due to WWTPs has notably increased, at the national average, over time, i.e., from 2.1% in 2006 to 5.7% in 2015 (Fig. 1b). We find the decrease of water stress levels has been mainly contributed by 1) the enlarged urban wastewater treatment capacity, 2) the increased flow of very well treated wastewater that replenishes surface water, and 3) the expanded scale of reclaimed water reuse within cities. In 2006, only a few cities in the North, Northwest, and Southwest of China benefited notably from the water stress alleviation, which was mainly due to the reuse of water from urban WWTPs rather than from wastewater replenishment (Fig. S5). Since 2010, the alleviation effect has spread much more widely among other cities. For the two arid cities of Tangshan and Beijing, local water stress was reduced by respectively 12% and 18% in 2010 by increasing water reuse for landscape, industrial, and municipal applications. Wastewater replenishment has also played an important role in alleviating water stress, as more urban WWTPs reached a higher standard of discharge water quality. For instance, water stress in two eastern cities, Suzhou and Wuxi decreased respectively by 20% and 19%, due to the considerable amounts of reclaimed water and treated wastewater that replenished the nearby Taihu Lake. In 2015, more than 15 cities, mostly located in North and Western China, showed a major progress towards their water stress alleviation. As a result, two northwestern cities, Urumchi and Datong downgraded their water stress status from "extremely high" to "high" [31].

For each unit of water stress alleviated by WWTPs, some of the northwestern cities of China caused the largest climatic impact during 2006–2015. For example, in 2015, Kashgar (CIWSA: 7.5 Mt $CO_2e$) and Yili (4.1 Mt $CO_2e$) (both located in Xinjiang province), had 6 to 10 times higher climatic impact from water stress alleviation than the national average. This indicates a huge potential in further improving treatment technologies and reusing more water towards achieving water-climate synergy in these cities. Although China's northern cities had a significantly high total climatic impact due to the larger proportion of reclaimed water as total wastewater treated, in 2015 these cities only registered a CIWSA value half as large of cities in the south (Fig. 1c). This was significant as cities in the south have higher water availabilities.

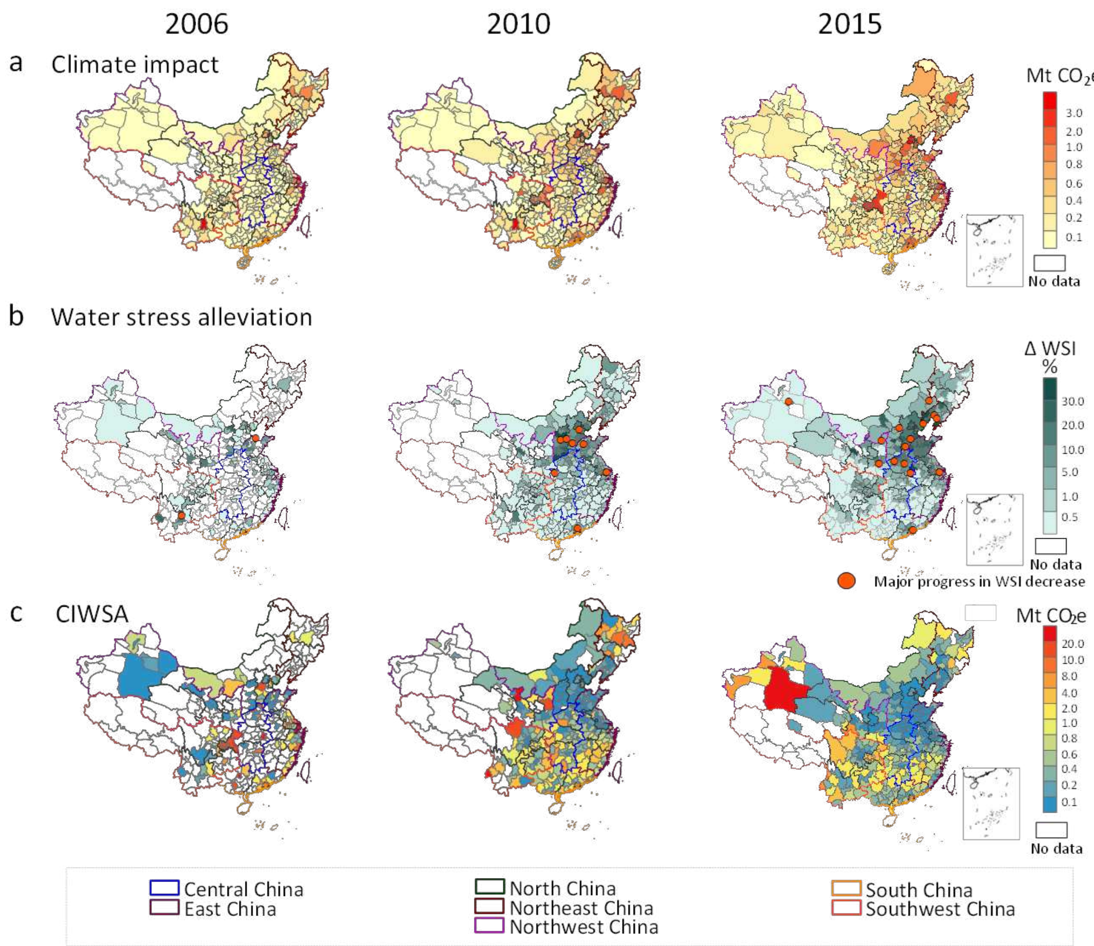

**Fig. 1 Variations in the life-cycle climate impacts of urban WWTPs and their respective water stress alleviation in 300 cities across China during 2006–2015. a)** City-scale climate impact induced by urban WWTPs in 2006, 2010, and 2015. **b)** City-scale water stress alleviation owing to wastewater treatment, quantified by the difference of Water Stress Index (WSI) with and without current urban WWTPs in 2006, 2010, and 2015 (see details in Fig. S6). Climate impact is represented by $CO_2$ equivalence ($CO_2e$), which is the sum of direct and indirect $CO_2$, $CH_4$, and $N_2O$ emissions weighted by their relative global warming risks in 100 years considering climate change feedback[32]. **c)** additional climate impact for water stress alleviation (CIWSA) is defined by the ratio of the life-cycle climate impact of WWTPs to the respective variation of water stress in each city. "Major progress in WSI decrease" indicate the situation that cities (orange dots) have downgraded their water stress status (e.g. from "extremely high" to "high").

## Influence of technological choices on the water–climate nexus

Wastewater treatment technologies have a profound influence on achieving the synergy of water stress alleviation and climate impact mitigation (Fig. 2). We investigated the most common WWTP technologies adopted in China, which include physicochemical (PC) treatment, anaerobic treatment (ANA), oxidation ditches (OD), anaerobic–anoxic–oxic/anoxic oxic ($A^2O$/AO) treatment, sequencing batch reactors (SBRs), microbial membranes reactors (MBRs), and other technologies, e.g., high–gradient magnetic separation and gas–liquid exchange.

Our results reveal that the $A^2O$/AO, PC, and OD technologies made the biggest contributions to water stress alleviation in 2015, accounting for 41%, 12%, and 18%, by respectively treating 42%, 11%, and 18% of the total wastewater volume. These three technologies also contributed the most to water replenishment, i.e., $A^2O$/AO (43%), PC (10%), and OD (18%), and provided the largest proportions of reclaimed water nationally. Overall, 66% of treated wastewater was replenished as renewable water, contributing to 72% of the water stress alleviation. However, reclaimed water comprised only approximately 6% of the total treated wastewater, implying the possibility of further increases in the future, e.g., by reusing treated wastewater discharged into the sea (7% of total discharge) as cooling water for coastal power plants.

From a life-cycle perspective, the greatest climate impact was from plants employing $A^2O$/AO, PC, and OD technologies, which accounted respectively for 33%, 16%, and 14% of the total. These technologies produced the biggest share of indirect $CO_2$ emissions (with $A^2O$/AO, PC, and OD technologies accounting for 61% in total). In terms of intensity, $A^2O$/AO, OD, and SBR technologies generally showed the lowest GHG emissions per unit of wastewater treated (between 1.0 and 1.2 kg $CO_2e/m^3$). As treatment technologies largely determine the climate impact intensities of WWTPs, distinctive strategies of the technology mix are required to achieve a synergy between water stress alleviation and climate impact mitigation in cities. For example, due to the large climatic impact resulting from their infrastructure, cities of Shanxi Province have the highest GHG intensities. Therefore, in Shanxi and other similar provinces, $A^2O$/AO and SBR technologies could be prioritized in decarbonizing their wastewater sectors (Fig. S7).

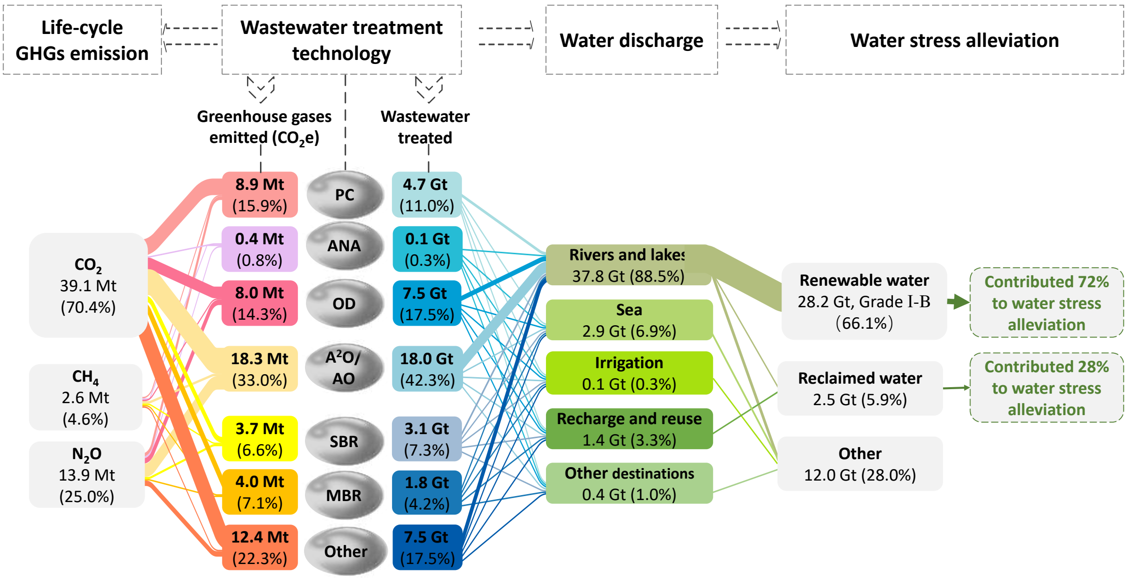


**Fig. 2 Sankey diagram revealing the nexus of water stress alleviation and climate impact mediated by wastewater treatment technologies in China in 2015**. The width of the connecting lines between the boxes is proportional to the magnitude of treated water or GHG emission. PC: Physicochemical treatment; ANA: Anaerobic treatment; OD: Oxidation Ditch; $A^2O$/AO: Anaerobic-Anoxic-Oxic/Anoxic Oxic; SBR: Sequencing Batch Reactor; MBR: Microbial Membranes Reactors; Other (other treatment technologies include, for example, High gradient magnetic separation and Gas-liquid exchange)

## Decoupling of wastewater climate impacts and water stress alleviation by 2030

China's wastewater sector faces significant challenges of climate impact mitigation in the next decade as the volume of wastewater and its reuse continues to increase (Fig. 3). In the Business-As-Usual (BAU) Scenario, i.e., if all technologies remain unchanged, to meet the mounting demand of urban wastewater treatment, the total climate impact of WWTPs increases from 129 Mt $CO_2$e in 2015 to an average of 205 Mt $CO_2$e by 2030 (within the range of 184–224 Mt $CO_2$e, considering the model uncertainties in Table S4; Fig. 3a). In the Water Reuse Enhancement (WRE) Scenario, the volume of reclaimed water increases by 430% between 2015–2030, therefore, increasing the total climate impact to an average of 224 Mt $CO_2$e (202–

244 Mt $CO_2e$) by 2030. In the WRE scenario, due to the new construction of water reuse facilities and their energy usage during their operation, the climate impact would be 10% higher than the BAU Scenario.

Adopting currently available low-carbon technologies throughout the life-cycle stage of urban WWTP, i.e., namely in wastewater treatment, sludge disposal, and water reuse, can open up opportunities for the decoupling of future climate impact and water stress alleviation. Technological diffusion, as reflected within each of the WRE+LCW, WRE+LCW+LCS, and WRE+LCW+LCS+LCR Scenarios, could reduce the life-cycle climate impacts by respectively 213 Mt $CO_2e$ (195–231 Mt $CO_2e$), 155 Mt $CO_2e$ (143–166 Mt $CO_2e$) and 146 Mt $CO_2e$ (135–157 Mt $CO_2e$) by 2030 (Fig. 3a). With the widest integration of technologies, i.e, within the WRE+LCW+LCS+LCR Scenario, the climate impact would be up to 35% less than that of the WRE Scenario by 2030. Furthermore, within the above scenario, a decrease of emissions would most likely occur between 2025 and 2030 –although the total amount in 2030 would still be greater than that of 2015. Among all stages, by 2030, implementing new technologies in the sludge disposal stage has the highest potential for climate impact reduction (76%); followed by, wastewater treatment (14%); and finally, water reuse (10%).

Another significant challenge that needs to be addressed is the increasingly uneven distribution of the future climate impacts of WWTPs among cities (Fig. 3b). Cities in the more developed region of Eastern China, due to their rapidly-growing need for wastewater and sludge treatment during urbanization, would be most accountable for the total WWTPs-related climate impacts. Even under the scenario with the widest integration of technologies, urban WWTPs in Eastern China would emit 35 Mt $CO_2e$ (32–38 Mt $CO_2e$) in 2030, compromising 24% of total emissions, while Northern, Central, and Southwestern China compromise respectively 20%, 17%, and 11%, of total emissions. Under this scenario, by 2030, the climate impacts of WWTPs in Southern and Eastern China would be respectively 9% and 2% lower than 2015; this is while, by 2030, the climate impacts in other regions would still be 10–43% higher than 2015 levels. Cities in Eastern and Southern China are expected to have the highest potential in reducing climate impacts related to WWTPs. Within these regions, the two megacities of Shanghai and Chongqing could slash respectively 45% and 42% of their total climate impacts by 2030. Given the low-carbon transition in various stages from plant construction to sludge disposal, these reductions can be achieved despite a rapid increase of water reuse infrastructure.

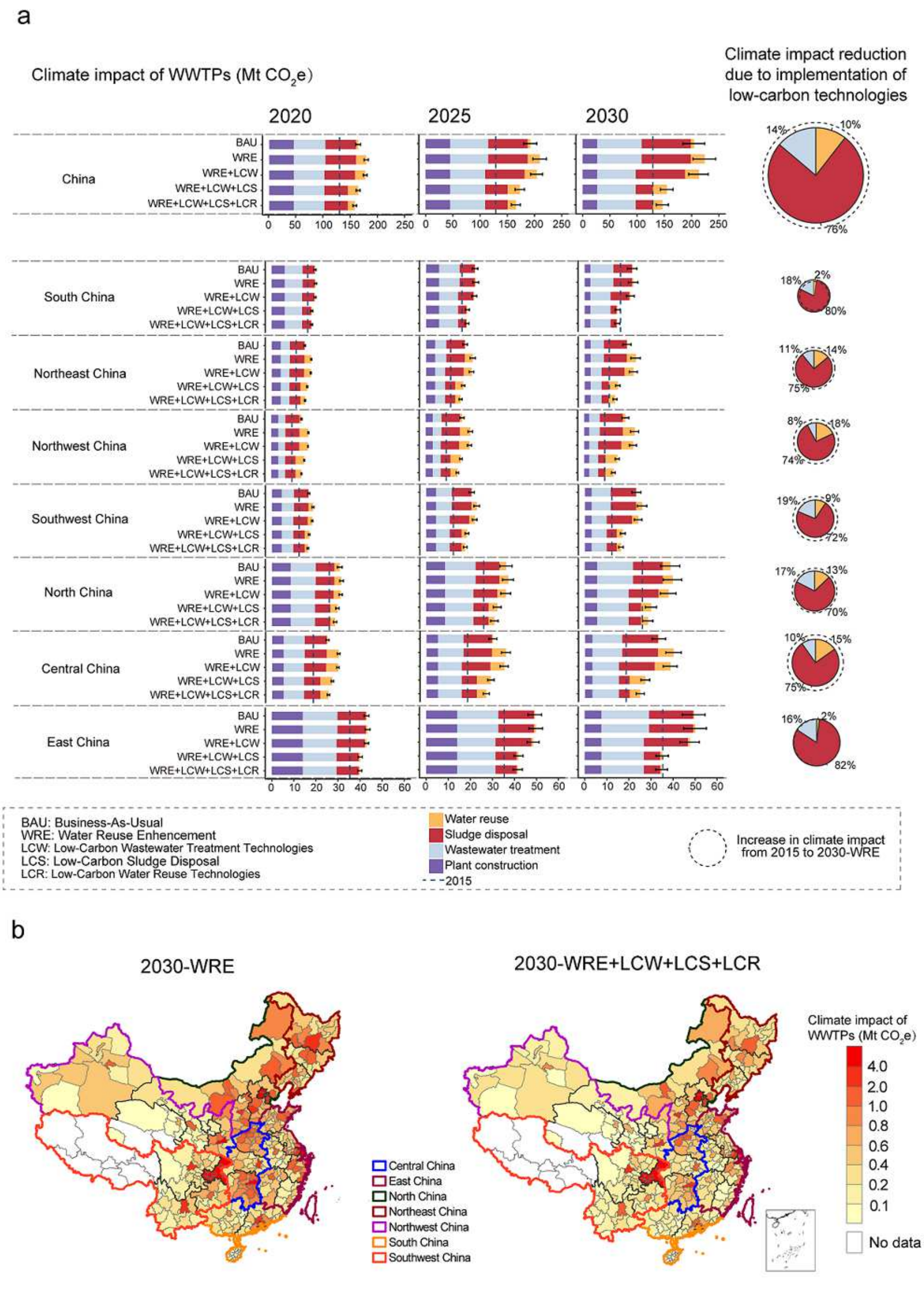


**Fig. 3 Future climate impacts of urban WWTPs in China under different scenarios**. **a**) Climate impact of plant construction, wastewater treatment, sludge disposal, and water reuse under different scenarios in different regions of China. The pie charts illustrate the climate impact reduction from the adoption of low-carbon technologies at the wastewater treatment, sludge disposal and water reuse stages. b) Climate impact of WWTPs in 2030 under the WRE Scenario and the WRE+LCW+LCS+LCR Scenario (see climate impacts of WWTPs in 2020, 2025 and 2030 under the BAU Scenario, the WRE Scenario and the WRE+LCW+LCS+LCR Scenario in Fig. S8).

The future increase of reused water will unleash the potential of water stress alleviation, whereby the WSI would further increase to 9.2% under the BAU Scenario or 12.1% under the WRE Scenario at national level by 2030 (Fig. 4a and Fig. S9). In 2030, 20 cities would experience a major decrease of their WSIs under the BAU Scenario, moving those cities towards a lower-water-stress status. Furthermore, the number of cities registering a major progress of water stress alleviation would increase to 30 under the WRE Scenario. The cities registering the largest progress in mitigating their water stress were in the Northeast, i.e., Panjin and Jinzhou, which would respectively decrease their WSI from 0.8 to 0.3 and from 0.5 to 0.3 and therefore improve their relatively high-water-stress to a less severe status.

Achieving water-climate synergy in the planning of urban WWTPs however, would not be an easy task. Although nationally, the climate impact per unit of water stress alleviated (CIWSA) in 2030 would be lower than 2015 under all future scenarios, stronger decoupling effects of climate impacts and water stress alleviated are only observed in scenarios where low-carbon technologies would be sufficiently adopted (Fig. 4b and Fig. S10). Under the WRE Scenario in 2030, the average CIWSA of all cities would be ~0.2 Mt $CO_2$e, which is notably lower than the BAU Scenario (~0.3 Mt $CO_2$e) and is only one third of the level in 2015. Under the scenario with the widest integration of technologies, the average CIWSA of all cities would dramatically drop to ~0.1 Mt $CO_2$e. Northeastern cities show the largest advancement in decoupling in 2030 (average CIWSA: 0.03 Mt $CO_2$e), decreasing by ~92% compared to that of 2015, followed by Southwestern and Southern cities with an average CIWSA of respectively 0.2 and 0.1 Mt $CO_2$e and respectively 83% and 82% lower than 2015 levels. Northwestern cities, with a CIWSA 2.4 times bigger than the national average in 2030, would continue to face the biggest challenge of reducing climate impacts while alleviating water stress. This calls for further efforts in improving technologies at all stages and also enlarging water reuse so that a stronger decoupling can be achieved.

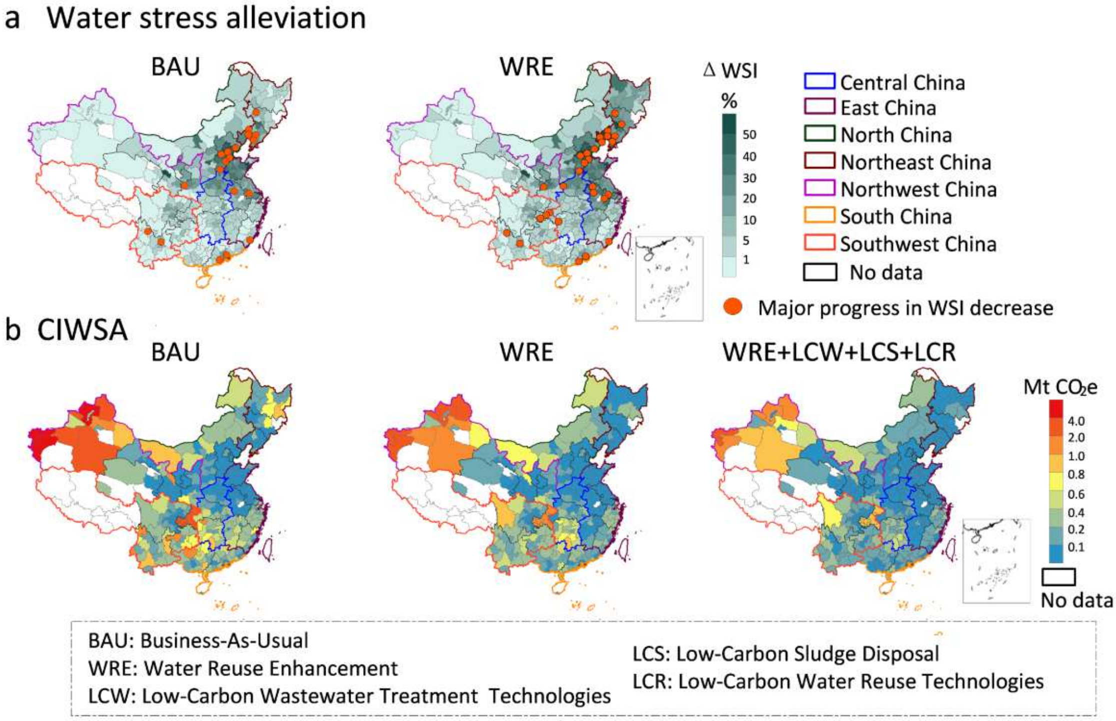


**Fig. 4 Water stress alleviation and additional climate impact (CIWSA) across cities of China in 2030 from WWTPs under different scenarios** (see CIWSA in 2020, 2025, and 2030 under different scenarios in Fig. S10).

# DISCUSSION

With the threat of future increasing water shortages and climate change, wastewater treatment and reuse will play a more critical role in the sustainable provisioning of water in cities, especially in those already suffering from severe water deficiency[1,3,33,34]. In China, the range of urban water availability is significant and often mismatched with the country's uneven regional economic and demographic growth –in the context of climate change this mismatch is widening even further. While wastewater treatment systems will help relieve water stress in the coming decades, their operation as well as their upstream and downstream processes and supporting infrastructures, will further impact the regional challenges of climate change[11,13,23,35]. Concurrent to the need for water sustainability, China has committed itself to achieve carbon-peak and neutrality goals; within this context, the wastewater treatment and water reuse industries face an unprecedented challenge of deep decarbonization.

Optimizing current technologies adopted in wastewater treatment systems in Northwest and South China, would lead to significant benefits in achieving water–climate synergies in these regions. Previously, most policies and practices in the operation of WWTPs were implemented with the intention of reducing electricity use while increasing their water treatment rates and effluent quality[33,36–39]. Recent discussions, however, have focused on how the supporting wastewater infrastructures may contribute towards achieving the national carbon peak and neutrality targets[40–42]. Our results reveal that the structural optimization of existing treatment technologies alone could realize a ~35% reduction of GHG emissions. Furthermore, arid cities located in the northwestern provinces, e.g., Xinjiang, and those in highly-populated provinces, e.g., Guangdong, have more urgent needs for reducing the climate impact per unit of water stress alleviated than cities located in the Northern regions of China. This means that these cities should introduce more efficient and affordable water treatment technologies[25,43] and ensure that these technologies can provide low- or net-zero-carbon solutions. In this avenue, treatment plants in these cities should uncover the potential of integrating more low-carbon applications into their current treatment systems, e.g., efficiently capturing $CH_4$ and heat from treatment processes and reusing them to offset emissions[42,44].

The unequal regional distribution of reclaimed water reuse and their climate impacts also have strong policy implications for decarbonizing wastewater systems. In this avenue, we find that the relatively water-deficient cities in the North and Northwest had large climate impacts in reusing more treated water from their WWTPs than other regions. Given the trend of urbanization and the changing climate, climatic impacts will increase over time. However, these regions also had the lowest CIWSA value, i.e., had the least impact on climate change for each unit of water stress alleviation, and showed significant progress in decreasing local water stress levels. Given the applications of more low-carbon technologies, water reuse in Southern cities will also have less climate-impact by 2030. The reuse rate of reclaimed water has been limited in China, i.e., <10% on average, much lower than the United Arab Emirates, Israel, and Cyprus, reaching respectively 59%, 87%, and 97%[45,46]. Although, not all Chinese cities could (or should) match the rates achieved by these countries, for those that are heavily reliant on distant and/or alternative sources of water, further development of their water reuse industry is vital for reducing the gap between the fast-growing urban water demand and the decreasing per capita water availability in an increasingly changing climate. This will potentially reduce the energy consumption and related GHG emissions in long-distance water transportation projects, e.g., the South-to-North Water Diversion Project[47]. However, the additional climate impact of fast-developing water reuse infrastructure may have to be partially offset by the decarbonization achieved in other processes of the wastewater systems, e.g., wastewater treatment and sludge disposal.

To support the synergistic mitigation of water stress and climate impacts, more investments in the wastewater treatment and water reuse industries are critical. According to our estimates, investments throughout the life-stages of the wastewater infrastructure of China might reach 361 billion CNY by 2030 (accounting for ~86% in wastewater treatment and ~14%

in water reuse) (Fig. S11a). This means that from 2015 to 2030, investments in wastewater treatment and water reuse will increase by 276%. A major challenge for more investments is that the current pricing for wastewater treatment and water reclamation in China is inadequate to compensate the capital investment and operational costs of wastewater treatment (Fig. S11b). Lower profits could reduce incentives for developing and adopting water-conserving and low-carbon technologies in the wastewater treatment and reuse companies. In this light, our results reveal that although the investment per unit of reused water may increase slightly due to the construction of new infrastructures, a 49% reduction of investment per unit of wastewater treated could be expected over the next decade (Fig. S11c). As high-tech WWTPs employing more low-carbon technologies become more affordable for cities at different development stages[25], investments in wastewater treatment and water reuse would be more profitable and climate-friendly nationwide and be embraced by both the business owners of the treatment plants and urban consumers of water demanding climate change action.

# METHODS

## Analytical framework and system boundaries

We quantified and tracked the city-scale water stress alleviation and climate impact induced by urban WWTPs over 2006–2030 using a consistent analytical framework (Fig. 5). To assess the alleviation effect of urban water stress, the framework included two key factors related to the operation of WWTPs that affects local water availabilities: 1) the treated wastewater discharged as replenishment to natural water bodies, and 2) the reuse of reclaimed water after treatment in plants. The contributions of both factors to the change of urban water stress over time were quantified. On the other hand, to capture the full climate impact of WWTPs, we included life-cycle stages such as infrastructure construction (including material extraction), plant operation (treatment process) and maintenance, water recycling and reuse, and sludge disposal. Here, we accounted for three types of GHG emissions, i.e., $CO_2$, $CH_4$, and $N_2O$, which were identified as the main contributors to WWTP-related climate impacts[11,14,22,23]. In this system boundary, we accounted for the $CH_4$ and $N_2O$ emitted from on-site treatment processes and the downstream sludge disposal, as well as the $CO_2$ emissions driven by energy use and resource inputs in all stages. The analytical processes in the three main life-cycle stages were as follows:

(1) WWTP infrastructure construction. This included GHG emissions embodied in the upstream extraction of raw materials, and the supply of fuels/electricity and other products used for WWTP construction. To reflect the change of construction technologies over time, we

assessed the construction-related climate impact of the WWTPs based on the year when the facilities were built. According to historical records, approximately 10% of urban WWTPs in China were built during 2000–2003, 20% during 2004–2006, and the remaining 70% during 2007–2010. Based on the average of previous estimates[48,49], the life span of the infrastructure of Chinese WWTPs is generally assumed to be 20 years. This meant that by 2010, 30% of the existing WWTPs were expected to operate 10 years more, while the remaining 70% could operate for another 20 years. Additionally, an increasing number of WWTPs were still under construction during 2011–2015. To reflect the above considerations, the climate impact caused by WWTP-related infrastructure was distributed evenly over their 20-year life cycle starting from the year of their construction.

(2) On-site wastewater treatment. Direct release of $CH_4$ and $N_2O$ from on-site wastewater treatment using various technologies was tracked and quantified. Generally, $CH_4$ is emitted during aerobic and anaerobic fermentation, while $N_2O$ is released in nitrification/de-nitrification processes (detailed analytical methods for these emissions are provided in *SI* Note 1). Indirect $CO_2$ emissions related to the resources and materials consumed in the operation and maintenance of WWTP systems were also included in this stage. Here, we focused on the on-site treatment within the plant while the collection and transportation of wastewater were excluded from beyond the scope of our research. Therefore, $CH_4$ emissions and the energy requirements related to sewer pipelines were excluded from the system boundary[11,13]. As only a very limited number of WWTPs in China have built-in $CH_4$ recovery equipment and even a smaller number of those have monitoring data, recovery of $CH_4$ as an energy source was not considered. Additionally, the discharge of treated wastewater (meeting certain water quality standard) as replenishment to the urban water cycle was analyzed for its contribution to alleviation of urban water stress.

(3) Water reuse and sludge disposal. The direct and indirect GHG emissions related to the energy (including electricity use) and resource inputs to water reuse and sludge disposal were included in this stage. Given the lack of separate detailed records on water reuse infrastructure, we used the plant-level investment data to compute the indirect emissions related to water reuse. Statistically, the four types of sludge disposal approaches applied most in China are land use, landfill, incineration, and recycling as building materials[26,50]. Direct GHG emissions from sludge disposal mainly include $CH_4$ emissions from landfill and $N_2O$ emissions from land use[51,52]. The recovery of $CH_4$ from landfills was excluded owing to its small scale and a lack of detailed on-site data nationally. To reveal how the water reuse industry could influence the water–climate nexus empowered by WWTPs, we also quantified the contribution of the reclaimed water reused to water stress alleviation across cities.

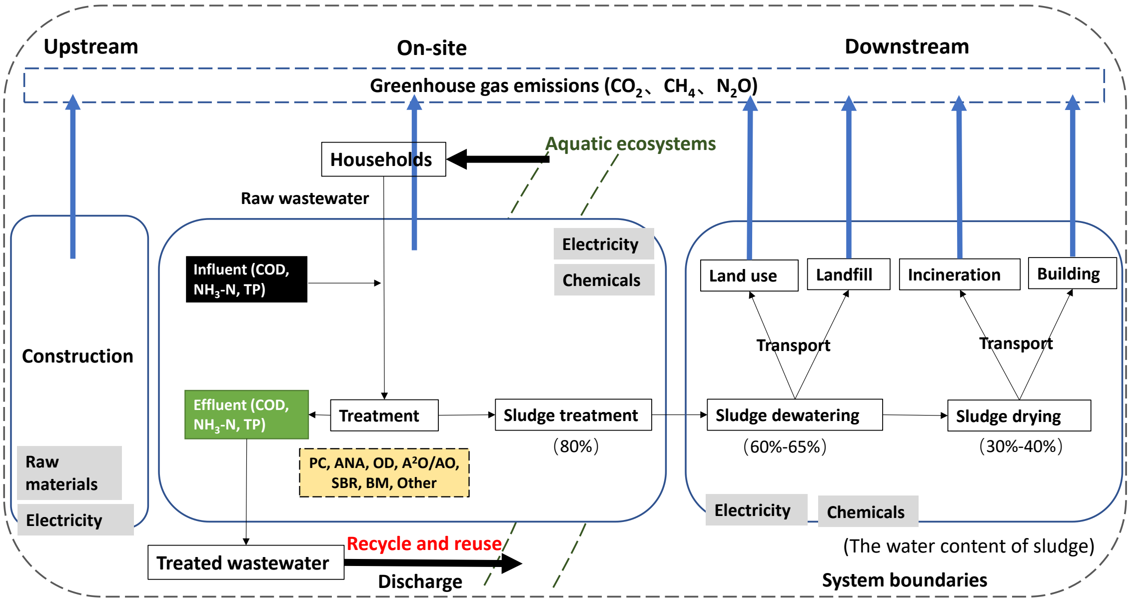


**Fig. 5 Analytical framework for water stress alleviation and GHG emissions related to urban WWTPs from a life cycle perspective.** Water quality indicators for effluent: COD: Chemical Oxygen Demand; $NH_3$-N: Ammonia Nitrogen; TP: Total Phosphorus.

## Accounting for life-cycle climate impact of wastewater infrastructure

We used IO-LCA to compute the climate impacts of WWTPs over different life-cycle stages. Compared with process LCA, the IO-LCA has the advantages of covering the full supply chains[53,54] related to wastewater infrastructure and of acquiring region-specific and time-sensitive emission factors in different life-cycle stages[11,55–57]. This makes the IO-LCA more suitable for the analysis of time-series climate impacts of WWTPs at the city scale, where energy and material input data of treatment plants are rare. Previous studies have developed various wastewater IO models for life-cycle environmental impact assessments[11,55,58,59].

Here, we developed a wastewater multi-regional input-output model (W-MRIO) that captured intermediate flows between urban wastewater systems and other socioeconomic sectors (Table S1). The model separated the sectors related to wastewater treatment systems, i.e., construction of WWTPs, on-site wastewater treatment and water reuse, and sludge disposal, from urban economies on both supply and demand sides. China's MRIO tables for different years were used to reflect the technical change of production over time. The W-MRIO model also distinguished seven types of treatment technologies used in wastewater treatment and reuse and four approaches for sludge disposal while evaluating their climate impacts. Accordingly, the W-MRIO table included four sections: (I) economic sectors providing goods and services, (II) wastewater-related construction, (III) wastewater treatment and reuse, and (IV) sludge disposal.

The direct GHG emission intensities of the sectors in all four sections were calculated as follows:

$$\boldsymbol{q}_{(n+3)\times 1} = \boldsymbol{Q}_{(n+3)\times 1} X_{(n+3)\times 1}^{-1} \qquad [1]$$

where $\boldsymbol{q}_{(n+3)\times 1}$ is the GHG emission intensity vector for economic sector (n), plant construction, wastewater treatment and reuse, and sludge disposal; $\boldsymbol{Q}_{(n+3)\times 1}$ represents the sum of $CO_2$, $CH_4$ (in $CO_2$e), and $N_2O$ (in $CO_2$e) emissions from the respective sectors; and $\boldsymbol{X}_{(n+3)\times 1}$ is the vector of total economic outputs of sectors.

Building on the developed W-MRIO model, the life-cycle climate impact ($CI$) of urban WWTPs and the contributions of different sections were calculated as follows:

$$CI = \begin{pmatrix} q_I \\ q_{II} \\ q_{III} \\ q_{IV} \end{pmatrix} \left( I - \begin{pmatrix} A_{I,I} & A_{I,II} & A_{I,III} & A_{I,IV} \\ 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 \\ 0 & 0 & A_{IV,III} & 0 \end{pmatrix} \right)^{-1} \begin{pmatrix} y_I \\ y_{II} \\ y_{III} \\ y_{IV} \end{pmatrix} \qquad [2]$$

where $q_I$, $q_{II}$, $q_{III}$, and $q_{IV}$ are the direct GHG emission intensity vectors for economic sectors (I), plant construction (II), wastewater treatment and reuse (III), and sludge disposal (IV), which are calculated as follows: $q_I = Q_I X_I^{-1}$, $q_{II} = Q_{II} X_{II}^{-1}$, $q_{III} = Q_{III} X_{III}^{-1}$, and $q_{IV} = Q_{IV} X_{IV}^{-1}$. Here, I is the identity matrix; $A_{I,I}$ is the input coefficient matrix, representing the amount of output from economic sector *i* required to produce one unit of output from economic sector *j*, which is given by $A_{I,I} = Z_{I,I} X_I^{-1}$, where $Z_{I,I}$ refers to the intermediate input flows in the economic system. Similarly, $A_{I,II}$, $A_{I,III}$, and $A_{I,IV}$ are the input coefficient matrixes, indicating the amount of output from economic sector *i* required to produce one unit of output respectively from construction (II), wastewater treatment and reuse (III), and sludge disposal (IV). Similarly, the input coefficient matrix $A_{IV,III}$ is defined by $A_{IV,III} = Z_{IV,III} X_{III}^{-1}$, where $Z_{IV,III}$ represents the disposal inputs of sludge produced by different treatment technologies. Finally, $y_I$, $y_{II}$, $y_{III}$, and $y_{IV}$ refer to the final demand vectors for the sectors in the respective sections (I, II, III, and IV) of the W-MRIO that drive the indirect GHG emissions related to WWTPs.

It is essential to articulate whether the climate cost of building wastewater treatment infrastructure decrease or increase through time while improving urban water sustainability. Therefore, we proposed an indicator termed the additional climate impact for water stress alleviation (*CIWSA*) to identify the decoupling between climate impact and change of water stress at the city scale, which is calculated by:

$$CIWSA_i = \frac{CI_i}{\Delta WSI_i} \quad [3]$$

where $CI_i$ refers to the life-cycle climate impact of city *i* attributable to all of its urban WWTPs; $\Delta WSI_i$ represents the decrease of urban water stress in city *i* owing to the treated wastewater replenishment and water reuse from its urban WWTPs (the calculation of urban water stress is described in the next section).

## Measurement of variations in urban water stress

The degree of water scarcity is usually measured by the Water Stress Index (WSI)[31,60,61], which is defined as the ratio of annual water withdrawals (for domestic, industrial, and agricultural uses) to the mean of available renewable water resources, i.e., surface blue water. The WSI indicator has been widely used to assess the status of water availability at the catchment or regional scales [16,62,63]. The regional WSIs within the intervals of 0–0.1, 0.1–0.2, 0.2–0.4, 0.4–0.8, and >0.8 represent respectively Low, Low-to-medium, Medium to high, High, and Extremely high levels of water stress[31]. Without the consideration of the effect of WWTPs, the urban water stress of city *i* was computed as follows:

$$WSI_i = \frac{WU_i}{RWR_i} \quad [4]$$

where *WU* is the quantity of water distributed to end users, including water lost in transmission, which is equal to water withdrawals plus net physical water imports; and *RWR* denotes the total amount of renewable water resources (surface blue water) available in a city[16]. In the absence of data on net physical water transfer at the city level, we calculated the local *RWR* by assuming that cities consume most of their required water from local sources.

To measure the alleviation effect attributable to WWTPs, we calculated the alternative water stress ($WSI'$) by incorporating two additional factors in the equation: 1) the replenishment of treated wastewater to the surface renewable water in cities that has met the strict discharge standard of effluent (here, this refers to China's I-B standard in the *Discharge Standards of Pollutants for Municipal Wastewater Treatment Plants* (GB 18918–2002)[20], and 2) the reuse of water reclaimed from WWTPs in various urban activities including industrial use, municipal water, and landscaping. Thus, $WSI'$ was used to measure the urban water stress of city *i* with the application of WWTPs:

$$WSI_i{}' = \frac{WU_i - RW_i}{RWR_i + RWW_i} \quad [5]$$

where *RW* refers to the volume of reclaimed water used in a city, and *RWW* refers to the volume of treated wastewater meeting the discharge standards, which thus can be treated as replenishment to the surface renewable water, e.g., lakes, rivers, and reservoirs.

$\Delta WSI$ refers to the relative variation between $WSI$ and $WSI^{'}$, which is calculated to represent the extent to which WWTPs have alleviated urban water stress and enhanced local water availability of city *i*:

$$\Delta WSI_i = \frac{WSI_i{'} - WSI_i}{WSI_i} \times 100\% \qquad [6]$$

## Scenario analysis

We simulated the future alleviation of water stress and climate impacts related to WWTPs by 2030 based on a set of scenarios that reflect the stepwise technological developments in their life-cycle stages including wastewater treatment, water reuse, and sludge disposal. Toward this end, we first projected future scales of urban wastewater treated in all the studied cities. Prior studies suggested that the amount of urban wastewater is most closely related to the population size and gross domestic product (GDP) of a city[64,65]. We developed an econometric model to estimate the amount of wastewater treated by 2030 for each city in China using a multifactor regression:

$$Wastewater_{it} = \beta_1 Population_{it} + \beta_2 GDP_{it} + \alpha_i + \gamma_t + \varepsilon_{it} \qquad [7]$$

where $Wastewater_{it}$ is the annual total amount of wastewater treated in city *i* in year *t*; $Population_{it}$ is the total population of city *i* in year *t*; $GDP_{it}$ is the GDP of city *i* in year *t*; $\alpha_i$ represents the city-level fixed effect, which controls the time-invariant city-level variables, such as food preferences and geographical location; $\gamma_t$ represents the year fixed effect, which is used to exclude the intervention of time-variant heterogeneity, such as technological improvement; and $\varepsilon_{it}$ represents a random error term. Standard errors are clustered at the city level.

Given the huge difference in the level of development among Chinese cities and the consequential differences in wastewater treatment scales, we constructed city-specific models with GDP per capita for level classification. We then estimated the future changes in treated wastewater amounts in the cities by associating the response coefficients of population and GDP (Table S2).

By combining treatment scale projection and technological transition, we set five policy scenarios to simulate alleviation on water stress and climate impact induced by WWTPs from 2015 (the benchmark year) to 2030 (Table S3). In the BAU Scenario, the volume of reclaimed water increased in proportion to the scale of wastewater treatment, and the distribution proportion among technologies remained the same as in 2015. In the other four scenarios, we applied stepwise improvement strategies building on the BAU Scenario. In the WRE Scenario, the volume of reclaimed water increased from 2.5 Gt in 2015 to 8.2 Gt in 2020 according to the *13th Five-year National Urban Sewage Treatment and Recycling Facilities Construction Plan*[26], which then was assumed to increase to 13.3 Gt simply in proportion to the scale of wastewater treatment in 2030. In the Low-Carbon Wastewater Treatment (WRE+LCW) Scenario or Low-Carbon Wastewater Treatment plus Sludge Disposal Integrated (WRE+LCW+LCS) Scenario, the technological mix for wastewater treatment or sludge disposal have seen improvements using more advanced technologies that have lower carbon intensities, i.e., a 4%–14% reduction in wastewater treatment and 15%–52% reduction in sludge disposal over the 2020–2030 period depending on the number of WWTPs that adopt these technologies across cities. Finally, in the Low-Carbon Wastewater Treatment plus Sludge Disposal and Water Reuse Integrated (WRE+LCW+LCS+LCR) Scenario, low-carbon water reuse technologies were applied more extensively nationwide, which leads to the further reduction of the climate impact intensity of water reuse by 32%[24]. Details of the settings and assumptions of the scenarios are provided in *SI* Note 2.

We also estimated the matched investments needed to support the operation of WWTPs that meet the treatment, reuse, and sludge disposal demand before 2030. In the BAU Scenario, the investments in wastewater treatment, water reuse, and sludge disposal were assumed to increase in proportion to the scale of wastewater treatment in cities. In the WRE Scenario, investments were estimated according to the *13th Five-year National Urban Sewage Treatment and Recycling Facilities Construction Plan,* increasing from 56 billion CNY in 2015 to 252 billion CNY in 2020; considering the development of different low-carbon technologies, and these include 77% for wastewater treatment, 15% for water reuse, and 8% for sludge disposal.

## Uncertainty analysis

The uncertainties introduced by the model and data used in assessing climate impact and water stress alleviation were quantitatively analyzed. First, we computed the uncertainty of the life-cycle climate impact result was introduced by supply chains in the W-MRIO model. The propagation of uncertainties along the modeling processes of IO-LCA was tracked using a standard deviation approach proposed in the literature[66-69].

The standard deviations of the historical climate impacts (*f*) of the WWTPs were calculated as follows:

$$\sigma_f = \sqrt{\sum_{i=1}^{n}\left(\frac{\partial f}{\partial q_i}\right)^2 var(q_i) + \sum_{i=1}^{n}\sum_{j=1}^{n}\left(\frac{\partial f}{\partial L_{ij}}\right)^2 var\left(L_{ij}\right) + \sum_{j=1}^{n}\left(\frac{\partial f}{\partial y_j}\right)^2 var\left(y_j\right)} \quad [8]$$

where uncertainty factors such as the elements ($i, j = 1 \ldots m$) of the Leontief Inverse matrix (**L**), final demand (**y**), and the direct emission intensity matrix (**q**) were included for computing the uncertainty of the climate impact of WWTPs with a 95% confidence interval (see detailed method in *SI* Note 3). Here, $\partial f/f$ was used to calculate the relative standard deviations for the climate impact of $CO_2$, $CH_4$ (in $CO_2e$), and $N_2O$ (in $CO_2e$). The analytical results of the climate impact uncertainty are presented in Fig. S12.

The uncertainties of future climate impacts and water stress alleviation that is influenced by the scale of the wastewater treated were also estimated. Although our predictions regarding the amounts of treated wastewater, reclaimed water, and sludge generated in 2020 were largely consistent with other estimates reported in the literature[70,71]; the econometric models of cities might introduce errors in projecting future climate impacts, water stress alleviation, and investments. We used the confidence intervals of population and GDP response coefficients to represent the upper and lower bounds of future wastewater scale, climate impacts, and water stress alleviation (Table S4). Additionally, these analytical results might be affected by other factors such as lack of information regarding the exact year of plant construction, the remaining number of years in service, and replacement of treatment and water reuse technologies in the next decade.

## Data

In this study, we used the plant-level WWTP monitoring data compiled from the China Environmental Statistics Database (CESD)[72]. This dataset covers all the major urban WWTPs operated in China during 2006–2015, comprising more than 6,000 urban WWTPs distributed across >300 cities in 30 mainland provincial regions (no available data for Tibet) in 2015 alone. The key plant-level information recorded includes geographic location; year of construction; wastewater treatment with different technologies, i.e., PC treatment, anaerobic treatment, oxidation ditches, $A^2O$/AO treatment, SBRs, and MBRs; approaches and scale of reclaimed water reuse, i.e., industrial use, municipal water, and landscaping; electricity consumption; discharge volume into various types of water body, i.e., rivers, lakes, reservoirs, and sea; amount of sludge disposed using different approaches, i.e., land use, landfill, incineration, and reuse as building material; concentrations of chemical oxygen demand; ammonia nitrogen; total phosphorus in influent and effluent; and annual operating load and expenditure.

The provincial-level $CO_2$ emission inventories by economic sector during 2006–2015 were derived from the China Emission Accounts and Datasets[73]. City-level data of water use, i.e., agriculture, industry, service, domestic, and ecological, and renewable water resources, i.e., surface water and groundwater, during 2006–2015 were obtained from provincial-level Water Resources Bulletin[74]. Data that were not available in the Water Resources Bulletin, i.e., Heilongjiang and Hainan provinces, were supplemented by Provincial Statistical Yearbooks. The population and GDP data of each city were derived from the China Statistical Yearbooks and City Statistical Yearbooks[75]. We collected and re-compiled China MRIO tables in 2007, 2010, 2012, and 2015[76–79] to match the required structure and format of the developed W-MRIO consisting of four different sections.

**Code availability.** Programming code for hybrid life-cycle model is available from the corresponding author on request.

**Data availability.** Sources of data used to perform this study are provided in Methods and Supplementary Information. Any further data that support the model of this study are available from the corresponding authors upon request.

## SUPPLEMENTAL INFORMATION

Supplementary information for this article is available:

Supplementary Notes 1 to 3

Figs. S1 to S12

Tables S1 to S5

Supplementary references

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## AUTHOR CONTRIBUTIONS

S.C., B.L. and B.C. designed the research; L.Z., S.C. and S.H. performed the research; S.C., L.Z., H.Y., A.K. and F.J. analyzed data; S.C., B.L., A.K., B.C., Z.L. and J.C.C wrote the paper; J.C.C reviewed and edited the manuscript.

## DECLARATION OF INTERESTS

The authors declare no competing interests.

## REFERENCES

1. Larsen, T. A., Hoffmann, S., Luthi, C., Truffer, B. & Maurer, M. Emerging solutions to the water challenges of an urbanizing world. *Science* 352, 928–933(2016).
2. UNESCO, UN-Water. United Nations World Water Development Report 2020: Water and Climate Change, Paris, UNESCO.
3. Jeffrey, P. et al. Water Reuse Europe Review 2018. https://www.water-reuse-europe.org/wp-content/uploads/2018/08/wre_review2018_final.pdf
4. Zhang, Y., Huo, J. & Zheng, X. Wastewater: China next water source. *Science* 374, 1332 (2021).
5. Rothausen, S. & Conway, D. Greenhouse-gas emissions from energy use in the water sector. *Nat. Clim. Change* 1, 210–219 (2011).
6. Zakkour, P. D., Gaterell, M. R., Griffin, P., Gochin, R. J. & Lester, J. N. Developing a sustainable energy strategy for a water utility. Part I. A review of the UK legislative framework. *J. Environ. Manage*. 66, 105–114 (2002).
7. Mo, W., Wang, R. & Zimmerman, J. B. Energy-water nexus analysis of enhanced water supply scenarios: a regional comparison of Tampa Bay, Florida, and San Diego, California. *Environ Sci Technol* 48, 5883-5891 (2014).
8. Hejazi, M. I. et al. 21st century United States emissions mitigation could increase water stress more than the climate change it is mitigating. *Proc. Natl. Acad. Sci.* 112, 10635-10640 (2015).
9. Stokes, J. R, Hendrickson T. P. & Horvath, A. Save Water To Save Carbon and Money: Developing Abatement Costs for Expanded Greenhouse Gas Reduction Portfolios. *Environ. Sci. Technol.* 48,13583-13591 (2014).
10. Parkinson, S., Krey, V., Huppmann, D., Kahil, T. & Riahi, K. Balancing clean water-climate change mitigation trade-offs. *Environ. Res. Lett.* 14, 014009 (2019).
11. Zhang, Q., Nakatani, J., Wang, T., Chai, C. & Moriguchi, Y. Hidden greenhouse gas emissions for water utilities in China's cities. *J. Cleaner Prod.* 162, 665-677 (2017).
12. Hendrickson, P. et al. Life-cycle energy use and greenhouse gas emissions of a building-scale

wastewater treatment and non-potable reuse system. *Environ. Sci. Technol.* 49, 10303-10311 (2015).

13. Zawartka, P., Burchart-Korol, D. & Blaut, A. model of carbon footprint assessment for the life cycle of the system of wastewater collection, transport and treatment. *Sci. Rep.* 10, 5799 (2020).
14. Wang, D. et al. Greenhouse gas emissions from municipal wastewater treatment facilities in China from 2006 to 2019. *Scientific Data* 9, 317, doi:10.1038/s41597-022-01439-7 (2022).
15. United Nations. *Sustainable Development Goals*, 2018. https://www.un.org/sustainabledevelopment/.
16. Zhao, X. et al. Physical and virtual water transfers for regional water stress alleviation in China. *Proc. Natl. Acad. Sci.* 112, 1031-1035 (2015).
17. Hejazi, M. I. et al. Integrated assessment of global water scarcity over the 21st century under multiple climate change mitigation policies. *Hydrol. Earth Syst. Sci.* 18, 2859-2883 (2014).
18. State Statistics Bureau. *China Environmental Statistical Yearbook on 2015*. China Statistics Press, Beijing (2016).
19. Bixio, D., Thoeye, C., Wintgens, T., Hochstrat, R., & Durham, B. Wastewater reclamation and reuse in the European Union and Israel: Status quo and future prospects. *Int. Rev. Environ. Strategies* 6, 251-268 (2006).
20. Ministry of Ecology and Environment. *The Discharge Standard of Pollutants for Municipal Wastewater Treatment Plant, GB18918-2002*. Beijing: The State Council of the People's Republic of China https://www.mee.gov.cn/gkml/hbb/bgth/201511/t20151111_316837.htm (2015).
21. Wang, J., Zhong, L. & Long, Y. "Baseline Water Stress: China." Technical Note. World Resources Institute, Beijing (2016). http://www.wri.org/publication/baseline-water-stress-china.
22. Yan, X. et al. Spatial and temporal distribution of greenhouse gas emissions from municipal wastewater treatment plants in China from 2005 to 2014. *Earth's Future* 7, 340–350 (2018).
23. Lundin, M., Bengtsson, M. & Molander, S. Life cycle assessment of wastewater systems: Influence of system boundaries and scale on calculated environmental loads. *Environ. Sci. Technol.* 34, 180-186 (2000).
24. Zeng, S. Y., Chen, X., Dong, X. & Liu, Y. Efficiency assessment of urban wastewater treatment plants in China: Considering greenhouse gas emissions. *Resour. Conserv. Recycl.* 120, 157-165 (2017).
25. Qu, J. et al. Emerging trends and prospects for municipal wastewater management in china. *ACS ES&T Engineering* 2, 323-336, doi:10.1021/acsestengg.1c00345 (2022).
26. Chai, C., Zhang, D., Yu, Y., Feng, Y. & Wong, M.S. Carbon footprint analyses of mainstream wastewater treatment technologies under different sludge treatment scenarios in China. *Water* 7, 918-938 (2015).
27. Wang, X. et al. Probabilistic evaluation of integrating resource recovery into wastewater

treatment to improve environmental sustainability. *Proc. Natl. Acad. Sci.* 112, 1630-1635 (2015).

28. Liu, L. et al. The importance of system configuration for distributed direct potable water reuse. *Nat. Sustain.* 3, 548–555 (2020).
29. Guo, T., Englehardt, J. & Wu, T. Review of cost versus scale: water and wastewater treatment and reuse processes. *Water Sci. Technol.* 69, 223–234 (2014).
30. Pryce, D., Memon, F. A. & Kapelan, Z. Life cycle analysis approach to comparing environmental impacts of alternative materials used in the construction of small wastewater treatment plants. *Environ. Adva.* 4, 100065 (2021).
31. Gassert F. P., Luo R. T., and Maddocks A. 2013. Aqueduct country and river basin rankings: a weighted aggregation of spatially distinct hydrological indicators. Working paper. Washington, DC: World Resources Institute (2013).
32. IPCC. *Climate Change 2014: Mitigation of Climate Change*. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Chang (2014).
33. Wang, X. et al. Evolving wastewater infrastructure paradigm to enhance harmony with nature. *Sci. Adv.* 4, eaaq0210 (2018).
34. Hsien, C., Low, J., Fuchen, S.C., & Han, T. W. Life cycle assessment of water supply in Singapore — A water-scarce urban city with multiple water sources. *Resour. Conserv. Recycl.* 151, 104476 (2019).
35. Huang, Y., Liu, S., Meng, F. & Smith, K. Increasing and diverging greenhouse gas emissions of urban wastewater treatment in china. https://doi.org/10.48550/arXiv.2202.11511 (2022).
36. Mo, W. & Zhang, Q. Energy-nutrients-water nexus: integrated resource recovery in municipal wastewater treatment plants. *J. Environ. Manage.* 127, 255-267 (2013).
37. Rieger, L., Takacs, I., & Siegrist, H. Improving Nutrient Removal While Reducing Energy Use at Three Swiss WWTPs Using Advanced Control. *Water Environ. Res.* 84, 170-188 (2012).
38. Rahman, S. M., Eckelman, M. J., Onnis-Hayden, A., & Gu, A. Z. Life-Cycle Assessment of Advanced Nutrient Removal Technologies for Wastewater Treatment. *Environ. Sci. Technol.* 50, 3020-3030 (2016).
39. Yu, C. et al. Managing nitrogen to restore water quality in China. *Nature* 567, 516–520 (2019).
40. Hao, X., Liu, R., & Xin, H. Evaluation of the potential for operating carbon neutral WWTPs in China. *Water Res*. 87 (2015).
41. Hao, X., Batstone, D., & Guest, S., J. Carbon neutrality: an ultimate goal towards sustainable wastewater treatment plants. *Water Res*. 87, 413-415 (2015).
42. Lu, L. et al. Wastewater treatment for carbon capture and utilization. *Nat. Sustain.* 1, 750-758 (2018).
43. Qu, J. et al. Municipal wastewater treatment in China: development history and future perspectives. *Front. Environ. Sci. Eng.* 13, 88 (2019).
44. National Development and Reform Commission of China. To help save energy, Biogas power

generation projects in Beijing have landed in Gaobeidian and Xiaohongmen. http://fgw.beijing.gov.cn/gzdt/fgzs/mtbdx/bzwlxw/202109/t20210909_2489670.htm (2021).

45. Water UK. *Net zero 2030 roadmap*. https://www.water.org.uk/routemap2030/wp-content/uploads/2020/11/Water-UK-Net-Zero-2030-Routemap.pdf (2020).
46. Kesari, K. K. et al. Wastewater treatment and reuse: A review of its applications and health implications. *Water Air & Soil Pollution* 232, 208 (2021).
47. Zhao, Y. et al. Energy reduction effect of the South-to-North Water Diversion Project in China. *Sci. Rep.* 7, 15956, doi:10.1038/s41598-017-16157-z (2017).
48. Zhang, Z. & Wilson, F. Life-cycle assessment of a sewage-treatment plant in South-East Asia. *J Chart. Inst. Water Env. Manage.* 14,51-56 (2000).
49. Yu, J. P. & Zheng, Z.G. Life cycle assessment system for the whole process of municipal wastewater treatment plants (in Chinses). *J. Civ. Archit.Environ. Eng*. 31 (2009).
50. Fu, X., & Zhong, L. Environment-Energy-Economic Benefit Assessment for Sludge-to-Energy: A Case Study of Capturing Methane from Sludge in Xiangyang, Hubei Province. World Resource Institute (2016).
51. Lombardi, L., Nocita, C., Bettazzi, E., Fibbi, D. & Carnevale, E. Environmental comparison of alternative treatments for sewage sludge: an Italian case study. *Waste Manag.* 69, 365–376, (2017)
52. Thomsen, M. & Lyck, E. Emission of $CH_4$ and $N_2O$ from Wastewater Treatment Plants (6B). National Environmental Research Institute, Denmark. 46 pp. – Research Notes from NERI no. 208 (2005). http://research-notes.dmu.dk
53. Wiedmann, T. A review of recent multi-region input–output models used for consumption-based emission and resource accounting. *Ecol. Econ.* 69, 211-222, doi:https://doi.org/10.1016/j.ecolecon.2009.08.026 (2009).
54. Davis, S. J., Peters, G. P. & Caldeira, K. The supply chain of CO2 emissions. Proc. Natl. Acad. Sci. U. S. A. 108, 18554-18559, doi:10.1073/pnas.1107409108 (2011).
55. Stokes, J. R., Hendrickson, T. P. & Horvath, A. Save water to save carbon and money: developing abatement costs for expanded greenhouse gas reduction portfolios. *Environ. Sci. Technol.* 48, 13583-13591, doi:10.1021/es503588e (2014).
56. Zheng, B., Huang, G., Liu, L., Guan, Y., & Zhai, M. Dynamic wastewater-induced research based on input-output analysis for Guangdong province, China. *Environ. Pollut.* 256, 113502.1-113502.11 (2020).
57. Corominas, L. et al. The application of life cycle assessment (LCA) to wastewater treatment: A best practice guide and critical review. *Water Res.* 184, 116058 (2020).
58. Lin, C. Hybrid input-output analysis of wastewater treatment and environmental impacts: a case study for the Tokyo metropolis. *Ecol. Econ.* 68, 2096-2105 (2009).
59. Chen, W., Oldfield, T. L., Patsios, S. I. & Holden, N. M. Hybrid life cycle assessment of agro-industrial wastewater valorisation. *Water Res* 170, 115275 (2020).

60. Vörösmarty, C. J., Green, P., Salisbury, J. & Lammers, R. B. Global water resources: Vulnerability from climate change and population growth. *Science* 289, 284-288 (2000).
61. Oki, T. & Kanae, S. Global hydrological cycles and world water resources. *Science*, 313, 1068-1072 (2006).
62. Zhang, C., Zhong, L. & Wang, J. Decoupling between water use and thermoelectric power generation growth in China. *Nat. Energy* 3, 792–799 (2018).
63. Lee, M. et al. Water-energy nexus for urban water systems: A comparative review on energy intensity and environmental impacts in relation to global water risks. *Appl. energy* 205, 589-601 (2017).
64. Boretti, A. & Rosa, L. Reassessing the projections of the World Water Development Report. *npj Clean Water* 2, 15 (2019).
65. Ding, L., Lv, Z., Han, M., Zhao, X., & Wang, W. Forecasting China's wastewater discharge using dynamic factors and mixed-frequency data. *Environ. Pollut.* 255, 113148 (2019).
66. Lenzen, M. Errors in Conventional and Input–Output-Based Life-Cycle Inventories. *J. Ind. Ecol.* 4, 127–148 (2001).
67. Lenzen, M., Wood, R. & Wiedmann, T. Uncertainty analysis for multi-regional input-output models: A case study of the UK's carbon footprint. *Economic Systems Research* 22, 43-63, doi:10.1080/09535311003661226 (2010).
68. Heijungs, R. & Lenzen, M. Error propagation methods for LCA—a comparison. *Int. J. Life Cycle Assess.* 19, 1445–1461 (2014)
69. Oita, A. et al. Substantial nitrogen pollution embedded in international trade. *Nat. Geosci.* 9, 111-115 (2016).
70. Lu, J. Y. Carbon footprint and reduction potential of Chinese wastewater treatment sector (in Chinese) (2019).
71. China Business Industry Research Institute. Review of the operation of China's sewage treatment industry in 2020 and forecast of its development trend in 2021. https://www.askci.com/news/chanye/20210114/1133091332584.shtml
72. Ministry of Environmental Protection of the People’s Republic of China (MEPPRC). Annual Statistic Report on Environment in China. China Environmental Science Press, 2006-2015
73. CEADs. China Emission Accounts and Datasets. http://www.ceads.net.
74. Department of Urban Surveys National Bureau of Statistics of China. https://data.cnki.net/yearbook/Single/N2022040095 (China Statistics Press, Beijing, 2021).
75. Ministry of Water Resources of the people’s Republic of China. http://www.mwr.gov.cn/
76. Liu, W. et al. Theory and Practice of Compiling China 30-Province Inter-Regional Input-Output Table of 2007. China Statistics Press, 2012.
77. Liu, W. D., Tang, Z., Chen, J. & Yang, B. China 30-Province Inter-Regional Input-Output Table of 2010. China Statistics Press, 2014.

78. Mi, Z. et al. Chinese $CO_2$ emission flows have reversed since the global financial crisis. *Nat. Commun.* 8, 1712, doi:10.1038/s41467-017-01820-w (2017).
79. Zheng et al. Regional determinants of China's consumption-based emissions in the economic transition. *Environ. Res. Lett.* 15, 074001 (2020).

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