Earth's Future

RESEARCH ARTICLE
10.1029/2021EF002601

Key Points:

- An analytical, generic framework was developed to assess wastewater treatment plants causing ecological risks in rivers under climate change
- Smaller streams will face higher ecological risks for almost all load classes of wastewater treatment plants in future climate
- Of the legally regulated effluent parameters for treated wastewater, ammonium-nitrogen concentration will pose the greatest ecological risk

Supporting Information:

Supporting Information may be found in the online version of this article.

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Citation:

Yang, S., Büttner, O., Kumar, R., Basso, S., & Borchardt, D. (2022). An analytical framework for determining the ecological risks of wastewater discharges in river networks under climate change. *Earth's Future*, *10*, e2021EF002601. https://doi.org/10.1029/2021EF002601

Received 14 DEC 2021
Accepted 21 JUL 2022

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An Analytical Framework for Determining the Ecological Risks of Wastewater Discharges in River Networks Under Climate Change

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Abstract Over the last decades, treatment of domestic wastewater promoted by environmental regulations have reduced human health risks and improved water quality. However, ecological risks caused by effluents of wastewater treatment plants (WWTPs) discharged into rivers still persist. Moreover, the evolution of these ecological risks in the future is intimately related to effects of changing climate, especially regarding streamflow in receiving rivers. Here, we present an analytical and transferable framework for assessing the ecological risks posed by WWTP-effluents at the catchment scale. The framework combines the size-class k of WWTPs, which is a load-proxy, with their outflows' location in river networks, represented by stream-order ω. We identify ecological risks by using three proxy indicators: the urban discharge fraction and the local-scale concentrations of each total phosphorous and ammonium-nitrogen discharged from WWTPs. About 3,200 WWTPs over three large catchments (Rhine, Elbe, and Weser) in Central Europe were analyzed by incorporating simulated streamflow for the most extreme projected climate change scenario. We found that WWTPs causing ecological risks in the future prevail in lower ω, across almost all k. Distinct patterns of ecological risks are identified in the k-ω framework for different indicators and catchments. We show, as climate changes, intensified risks are especially expected in lower ω receiving effluents of intermediate-k WWTPs. We discuss the implications of our findings for prioritizing WWTPs upgrading and urging updates on environmental regulations. Further discussions underline the feasibility of applying the framework to any geographical regions and highlight its potentials to help in achieving global long-term commitments on freshwater security.

1. Introduction

Human settlements require clean water in sufficient quantity with constant supply and at the same time produce continuous wastewater flows loaded with pathogens, organic and mineral matter, nutrients, and pollutants (Ceola et al., 2019; HDR, 2002). Exposure to household wastewater in living environments can directly endanger human health caused by waterborne pathogens and accelerating their spread (Naik & Stenstrom, 2012; Wolfe et al., 2018). Thus, collecting and draining wastewater from domestic sources were considered as key strategies to sanitizing residential areas, resulting in ~60% connection of the recent global population to sewer systems (WWAP, 2017). As the basic needs for human health have been satisfied, larger-scope environmental problems induced by untreated wastewater releases have been addressed and resolved to mitigate water quality impairment, protect aquatic ecosystem integrity, and further ensure sufficient freshwater availability (Arden & Jawitz, 2019; EEA, 2015; Viessman et al., 2009). Centralized wastewater treatment systems, such as municipal wastewater treatment plants (WWTPs), have played major and significant roles in accomplishing these complex and long-term societal and environmental challenges. Jones et al. (2021) estimate that as of 2015, about 52% of worldwide municipal and manufacturing wastewater is released into the environment after treatment processes, with values ranging from ~16% in Sub-Saharan Africa to ~86% in western Europe. Over the coming decades, construction of new WWTPs is expected to meet targets to halve untreated wastewater discharges by 2030 under the Sustainable Development Goal (SDG) Target 6.3 formulated by the United Nations (UN, 2015). In addition, the need for extended wastewater collection and treatment will be amplified due to continuous increase of human-produced wastewater followed by substantial growth of the global population and its demand for water, which is projected to increase of 50% by 2030 (UN, 2016).

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In the course of the creation of nationwide wastewater infrastructures over time, national-scale and international regulatory standards for effluents from WWTPs have been implemented over the recent decades, such as the EU Urban Waste Water Treatment Directive (UWWTD) (EEC, 1991) and the US Clean Water Act (USEPA, 1972). These regulations focused on end-of-pipe treatments, which do not guarantee that a good-status of the aquatic ecosystem is achieved in many receiving water bodies (Büttner et al., 2020). This is mainly attributed to the fact that the environmental regulations for WWTP-effluents are lacking a systematic and site-specific incorporation of the properties and sensitivities of receiving bodies, such as hydrological flow conditions and spatial interference of wastewater discharges (Freni et al., 2010). For example, despite remarkable reduction of nutrient loads discharged from European WWTPs in the recent few decades, point-source nutrient loads still pose local and regional scale problems regarding water quality and ecological impairments in central Europe (Büttner et al., 2020; Yang, Büttner, Kumar, et al., 2019). Besides the conventional pollutants for which release regulations are already established, newly emerging contaminants discharged from WWTPs without current-applied regulations have additionally jeopardized the health of river ecosystems downstream of WWTP outlets by increasing ecotoxicological risks, altering macroinvertebrate community composition, and losing species biodiversity (Gücker et al., 2006; Rice & Westerhoff, 2017; Weitere et al., 2021). To assess the environmental security of WWTP-effluents, a more comprehensive perspective, which includes their impacts on the water quality of receiving river courses, is needed. At the global scale, identifying and improving WWTPs causing significant ecological risks to receiving water bodies are indispensable requirements in successfully undertaking international commitments for protecting water quality and aquatic ecosystems with a long-term but legally binding schedule, such as the UN SDGs and the EU Water Framework Directive (WFD) (European Commission, 2000).

With the aforementioned emission to receiving water perspective, changing climate conditions should be taken into account for assessing ecological risks of treated wastewater discharged via WWTPs under expected global warming and changing hydrological regimes (Rakovec et al., 2022). Increasing trends of continuous heatwaves and hot weather have prompted greater water usage of people to cope with heat by more frequently washing and cooling private and public fields (EC & EEA, 2020; USEPA, 2015). Unprecedented heatwaves during the last few decades have sparked a record-breaking amount of daily water use in Europe (BBC, 2018; Lomax, 2021), North America (Glassman, 2018; Labbe, 2021), and Australia (Pearlman, 2018; Wells, 2014). Thus, regarding the emission, the adaptive behaviors of sweltering people are likely to augment the mass flux of water and pollutants discharged from WWTPs. In terms of the receiving water perspective, climate-related hydrological regimes of receiving rivers play crucial roles in determining the ecological risk status of WWTP emissions. A lower magnitude of river flow provides less dilution capacity, resulting in a higher fraction of WWTP-effluents in receiving streams and likely more direct exposure of people to anthropogenic pollutants in recreational places (Siddiqui et al., 2020; Yang, Büttner, Kumar, et al., 2019). In addition, the persistence of drier hydrological regimes results in prolonged residence time for solutes or particulate matters from WWTPs in river water columns. Consequently, undesirable ecological statuses such as excessive algal blooms or fish deaths can be more easily observed along rivers (Kamjunke et al., 2021; Ritz & Fischer, 2019; Sheldon et al., 2021). Such extreme meteorological and hydrological conditions are expected to be more frequent under climate change (Hari et al., 2020; Hulley et al., 2020; Stott, 2016), thus calling for implementation of WWTP-technology adaptation to changing climate that shall be accompanied by an ecological risk assessment of WWTP-effluents.

In this context, our aim is to develop an analytical, generic, and transferable framework to assess ecological risks of WWTP-effluents and inform improvement of environmental guidelines across river basins in any climate zone. We envision this framework to be applicable under future scenarios of growing population, increasing wastewater production and WWTP construction, climate changes and their cascading impacts on human usage of water resources, and hydrological regimes of receiving rivers. This study focuses on presenting the framework's fundamental principles, application, and implications under an extreme scenario for climate change. Cost-efficiency or socioeconomic analysis related to the implications of our results is outside the scope of this study. To this end, we selected the Rhine, the Elbe, and the Weser rivers in central Europe. The Rhine and the Elbe have similar drainage area but significantly contrasting properties regarding climate (Marx et al., 2018; Thober et al., 2018), hydrological responses (Ionita & Nagavciuc, 2020; Pfeiffer & Ionita, 2017), matter transport from land to river (Hoffmann et al., 2020; Li et al., 2020), and river ecosystem responses (Hardenbicker et al., 2014, 2016). They also display notably similar characteristics in the hierarchical scaling patterns of the coupled human-WWTPs distribution across river networks (Yang, Büttner, Jawitz, et al., 2019), see Section 2.1 for details of the study areas. Specifically, we employed a stream-ordering scheme that systematically characterizes the hierarchical

Table 1
Characteristics for the Study River Basins

Characteristic	Unit	River basin		
		Rhine	Elbe	Weser
Total basin area [a,b]	[km^2]	~185,000	~144,000	~46,000
Source [a,b]	[–]	Southeastern Swiss Alps	Northern Czech giant mountains	Central German highlands
Mouth [a,b]	[–]	North Sea in the Netherlands	North Sea at Cuxhaven (Germany)	North Sea at northern Bremen (Germany)
Mainstem length from source to mouth [a,b,c]	[km]	~1,300	~1,100	~700
Total population [a,b,c]	[million people]	~58	~25	~8.4
Spanning countries [a,b,f]	[–]	Germany (55); Swiss (18); France (13); The Netherlands (6); Austria; Belgium; Italy; Liechtenstein; Luxemburg	Germany (66); Czech Republic (33); Austria; Poland	Germany (100)
Climate zone [a,b]	[–]	Maritime in the west (dominated); Continental in the southeast	Maritime in the northwest; Continental in the southeast (dominated)	Maritime in the north; Continental in the southeast (almost equal)
Mean and range of annual precipitation [a,c,d]	[mm/year]	~950 (from 500 to 3,500)	~660 (from 450 to 1,600)	~780 (from 600 to 1,100)
Mean annual discharge [c,d,e]	[m^3/s]	~2,300 at Emmerich gauging station, close to the Dutch-German border	~860 at Cuxhaven gauging station, Germany	~370 at Intchede gauging station located at most downstream without tidal effect

[a]Uehlinger et al. (2009). [b]Pusch et al. (2009). [c]Yang et al. (2021). [d]Pfeiffer and Ionita (2017). [e]Hardenbicker et al. (2016). [f]Values in parentheses indicate % of each country constituting the total basin area. Note that the cases of >5% are presented here.

structure of river networks (Horton, 1945; Strahler, 1957), synthesized two independent geo-spatial data sets (i.e., WWTP discharge locations and their receiving river streams), and modeled streamflow data under a projected extreme climate to address the following questions:

1. How will the impact of WWTP-effluents on ecological risks in receiving rivers develop under climate change?
2. How would the impacts be *different* among large catchments in central Europe?
3. How can an *analytical*, *generic*, *and transferable framework* be formulated to facilitate a quantitative and spatially explicit risk assessment of adverse impacts in aquatic ecosystems at the catchment scale?

2. Data

2.1. Study River Basins

This study focuses on three large river basins in central Europe: the Rhine, the Elbe, and the Weser, with respective total drainage areas of ~185K, ~144K, and ~46K km^2 (Table 1), which were selected based on the following rationale. First, the fraction of people connected to WWTPs is overall high (>90%) in central European countries compared to other European regions (EEA, 2020b). Furthermore, WWTPs that apply the most stringent tertiary treatment processes serve at the highest rate (~77%) to people residing in central Europe (EEA, 2020b). Second, several countries located in the select three river basins have sufficient similarities in lifestyle and water resource usage patterns of residents (EurEup, 2020). The strong commonalities in the degree of urban sanitary infrastructure, the level of advanced technology utilized in WWTPs, and socio-economic conditions enable to minimize inherent uncertainty in international-scale metadata analyses. Finally, river water bodies in central Europe mostly characterized as continental and maritime climate zones are generally expected to experience the least changes in hydrologic conditions across Europe, simulated under projected global warming scenarios (e.g., Marx et al., 2018; Thober et al., 2018). Note that here we limited our analyses for the three catchments to drainage areas in Germany, Czech Republic, and the Netherlands, resulting in a coverage of 61%, 99%, and 100% (the Rhine, the Elbe, and the Weser, respectively; Table 1), because of constraints in the availability of WWTP metadata (see Section 2.2).

We used the freely available EU Hydro data set to indicate network configurations of the Rhine, the Elbe, and the Weser Rivers (EEA, 2020a). The EU Hydro river networks and the corresponding catchment boundaries are initially derived from EU Digital Elevation Models and verified through comparison to photo-interpreted river networks provided by all 39 countries involved in the European Environmental Agency (EEA). To characterize the structural hierarchy of the select river networks, we referred to the metric of Horton-Strahler stream-orders ω (Horton, 1945) included in the EU Hydro data set. All three catchments finally have seventh-order streams draining to the North Sea (Table 1). Furthermore, the EU Hydro data set builds the basis for graph theory based networks that are used to calculate cumulative effects.

2.2. Wastewater Treatment Plants

Member countries of the EU are mandatorily responsible for reporting all WWTPs with population equivalents (PE) >2,000 and their properties to the EU under the UWWTD (EEC, 1991). Note that WWTPs with PE $\leq$ 2,000 can be voluntarily reported. PE is a normal established proxy of total sanitary flows from people connected to each WWTP via sewer systems and non-sanitary flows, such as urban storm water runoff. PE magnitude is estimated by per-capita mean loading equivalents for biochemical oxygen demand or nutrients (nitrogen or phosphorus). For example, one PE is considered as an organic biodegradable load having a 5-day biochemical oxygen demand of 60 g per day (EEC, 1991). The EEA organizes the reporting process of WWTPs every 2 years and makes the data publicly available (EEA, 2019). We used the latest data set published in the year 2016.

Since there are no compatible standards to categorize WWTPs over Europe, all WWTPs analyzed here were classified as five size-classes k by following German regulations based on the PE magnitude: $k = 1$ for PE $\leq$ 1,000, $k = 2$ for 1,000 < PE $\leq$ 5,000, $k = 3$ for 5,000 < PE $\leq$ 10,000, $k = 4$ for 10,000 < PE $\leq$ 100,000, and $k = 5$ for PE > 100,000 (https://www.gesetze-im-internet.de/abwv/anhang_1.html). The discharge points of the WWTPs were defined by overlapping the geospatial data of WWTPs with the EU Hydro river network. Note that WWTPs in Switzerland and France covering ~31% of the Rhine catchment area were excluded from this study. Switzerland is not an EU member state and thus has no duty to report WWTP metadata. French WWTPs are reported with geospatial information but their nutrient output loads are missing. Overall, the total number of WWTPs analyzed for each catchment is ~1,820, ~900, and ~480 for the Rhine, the Elbe, and the Weser rivers, respectively (Figure 1). About 5% of total WWTPs have <2000 PE.

2.3. River Discharge Simulated Under a Climate Change Scenario

River discharge data are prerequisite to estimate the impacts of WWTP-effluents on the water quantity and quality of receiving river bodies at local and regional scales. We used daily river discharge simulated over the entire Europe for a given climate condition (Marx et al., 2018) using the 5 km-grid based mesoscale hydrological model (mHM; www.ufz.de/mhm) (Kumar et al., 2013; Samaniego et al., 2010). The reliability of the mHM framework and simulation results is well established at German, European, and global scales (Krysanova et al., 2017; Samaniego et al., 2019; Zink et al., 2017). Simulating the mHM requires hydro-metrological data inputs from the simulation results of a general circulation model (GCM) under a representative concentration pathway (RCP) reflecting the projected climate change conditions in the future. In this study, we employed the mHM results based on the GCM/RCP combination of GFDL-ESM2M/RCP8.5. Of three warming levels (1.5, 2, and 3K) compared to preindustrial conditions, we selected the 3K warming level to consider river discharge regimes resulting from the most severe expected climate change conditions. Sequentially, the simulated river discharge for the first 30-year period crossing the warming level of 3K (the years 2067–2096—*far future scenario*) was compared to that of the reference period (the years 1971–2000—*historical scenario*). Hydrological regimes for the selected periods were characterized (Text S1 and Figure S1 in Supporting Information S1). GFDL-ESM2M climate model outputs are part of the large-scale community driven Inter-Sectoral Impact Model Intercomparison Project (Warszawski et al., 2014), which has been part of several impact assessment studies (e.g., Marx et al., 2018; Samaniego et al., 2019; Thober et al., 2018; Warszawski et al., 2014). Climate model outputs were made available globally at a 0.5-degree spatial resolution with bias corrected using a trend preserving approach by Hempel et al. (2013). To further enable high-resolution hydrologic simulations, climate projection data sets were disaggregated to 5 km resolution using the external drift kriging (EDK) approach using terrain elevation as an external drift. Please refer to Marx et al. (2018), Thober et al. (2018), and Samaniego et al. (2019) to find

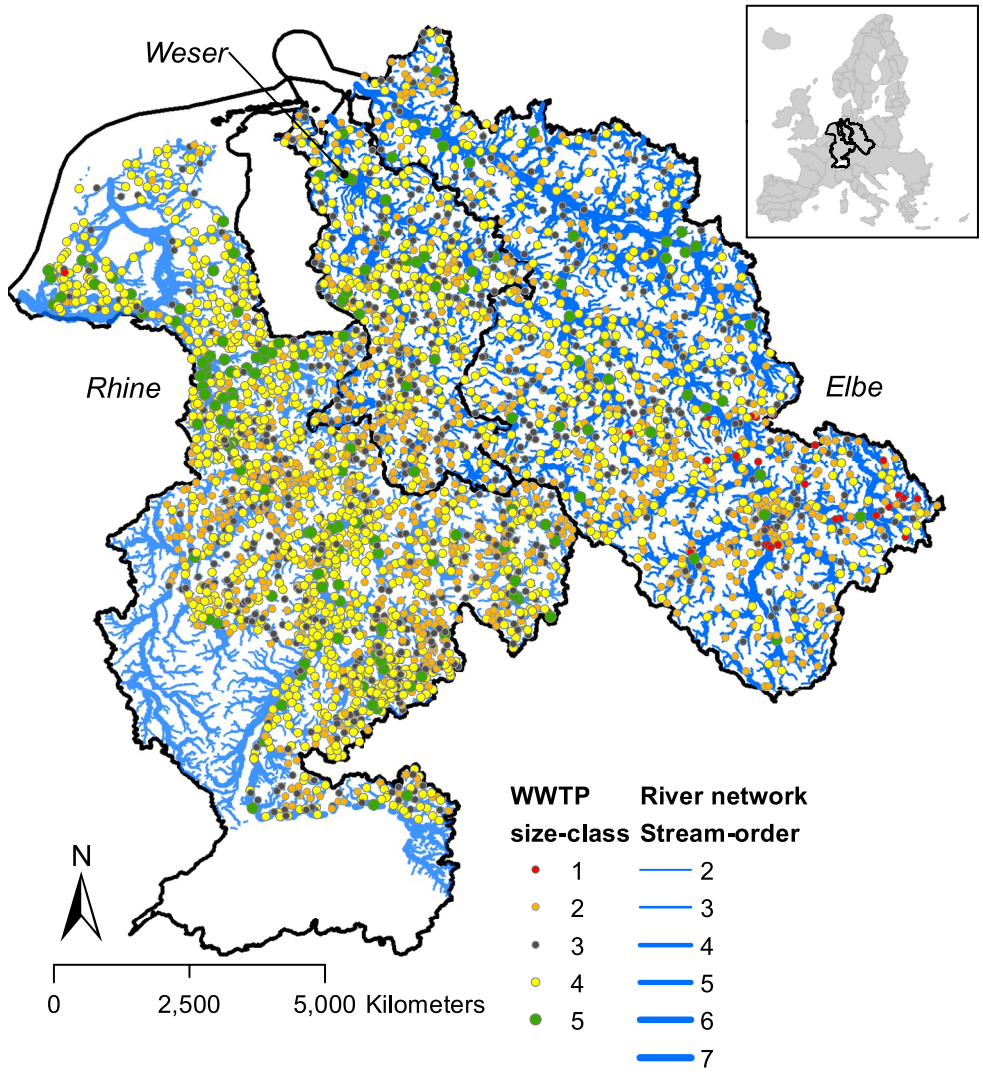

Figure 1. Spatial distribution of wastewater treatment plants (WWTPs) studied over the Rhine, the Elbe, and the Weser catchments in central Europe. The boundary of the three catchments is highlighted with a black solid line. The upper right panel shows the geographical location of the studied catchments in the European continent. Circle markers indicate WWTPs with five size-classes distinguished by different colors (red, orange, black, yellow, and green for size-classes from 1 to 5, sequentially). Blue lines are EU Hydro river networks of the three catchments. Thicker lines represent higher stream-order. Smallest streams with 1st-order are not shown in the map for efficient visualization.

details on the basic assumptions, approaches, and validations of the model simulations under projected climate change and to understand detailed measures of GCM downscaling and bias correction.

3. Methods

The data sets are analyzed to assess ecological risks of WWTP-effluents through the seven steps stated below. The following four sub-sections provide detailed descriptions of these steps (Step 1 in Section 3.1; Step 2 in Section 3.2 and Section 3.3; and Steps 3–7 in Section 3.4).

1. Step 1: Assign individual points of WWTP-effluents to receiving river body information (i.e., river basin, stream-order, and simulated river discharge data).
2. Step 2: Calculate the daily time series of three proxy indicators at individual points of WWTP discharge for a given 30-year period (i.e., historical or far future).

3. Step 3: Calculate the exceedance probability of each indicator value equaling the environmental threshold that corresponds to being at least *Good-Ecological-Status* of surface water bodies.
4. Step 4: Select a management threshold ($WWTP^*_{threshold}$; ranging from 0% to 100% in principle) to categorize each WWTP as Causing Ecological Risks (WWTP-CER) or not-Causing Ecological Risks (WWTP-not-CER).
5. Step 5: Compare the $WWTP^*_{threshold}$ with the exceedance probability result of Step 3 for individual WWTPs.
6. Step 6: If the $WWTP^*_{threshold}$ > the Step 3 result, classify the analyzed WWTP as WWTP-CER. Otherwise, classify it as WWTP-not-CER.
7. Step 7: Visualize the resultant binary classification into the two dimensional domain consisting of WWTP size-class k and stream-order ω. Finally, this results in the k-ω framework for assessing ecological risk of each proxy indicator for a given time period.

3.1. Assignment of WWTP-Effluent Points to Receiving River Information

A receiving stream and its discharge data should be matched with each of the WWTP discharge points to estimate the effects of WWTP-effluents on river bodies. Given that the EU Hydro river networks, simulated discharge data, and WWTP metadata are independent data sources, developing an algorithm was needed to make spatial alignment across multiple data sets with different dimensions (i.e., point, line, and grid) and spatial resolutions possible. For example, two or more stream lines with different stream-orders can fall within the same grid cell used for the mHM discharge simulations. In general, only one of these multiple stream-orders should be accepted as the correct one, as suggested by the order of magnitude of the mHM-simulated river discharge.

Therefore, we developed an algorithm to identify the most reasonable river discharge simulation data for a stream or river reach receiving effluents of a given WWTP. The key principle was to minimize the difference between stream-orders defined in the EU Hydro data set and those derived from the mHM while simultaneously maximizing the spatial adjacency of a WWTP location and an mHM-grid. Details are given in Text S2 in Supporting Information S1.

3.2. Urban Wastewater Discharge Fraction

We used the urban discharge fraction (UDF) as a proxy for estimating the impacts of treated wastewater discharged via WWTPs to their receiving river bodies at local scales (Yang, Büttner, Kumar, et al., 2019). UDF is defined as the ratio between the treated wastewater from WWTPs and the total water flux in the respective receiving river:

$$\mathrm{UDF}_{\mathrm{local},i}(t) = \frac{Q_{U,i}}{Q_{U,i} + Q_{R,i}(t)} \tag{1}$$

where $Q_{U,i}$ is the water flux discharged from the i-th WWTP, and $Q_{R,i}(t)$ indicates the time series of daily river discharge at a given location receiving the effluents of the i-th WWTP. Here, $Q_{U,i}$ is calculated as a sum of wastewater from households and small industries (Q_H), storm water (Q_{SW}), and sewer infiltration water (Q_{SIW}). Taking the last two terms into account together as $Q_{\mathrm{SW}} + Q_{\mathrm{SIW}} \sim Q_H$ (Büttner et al., 2020), the final expression of $Q_{U,i}$ is given as:

$$Q_{U,i} = 2 \cdot Q_{H,i} \tag{2}$$

where $Q_{H,i}$ is estimated as a product of the population equivalent served by the i-th WWTP and the annual mean daily water usage per-capita in Germany (123 l/day/person). Note that in this study, $Q_{U,i}$ is dealt with as a constant over time under the assumption that daily per-capita water usage of people living in the three catchments would only change for subseasonal time periods under far future climates. By referring to temporally varying $Q_{R,i}(t)$, $\mathrm{UDF}_{\mathrm{local},i}$ (t) is calculated with a daily time-step t for the i-th WWTP at the local scale under the two climate scenarios. The value of UDF ranges from 0% to 100%. A higher UDF value indicates a lower dilution capacity of the receiving river.

In addition to the aforementioned local-scale assessment, we also estimated the basin-scale UDF at each discharging point of WWTPs by reflecting accumulated effluents of all upstream WWTPs along river flow paths. A graph theory based network grounded on the EU Hydro river network data set was applied to identify upstream WWTPs for a given WWTP (i.e., WWTPs serving as nodes and river segments as edges). Indeed, the location of

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Table 2
Nutrient Removal Efficiency (ε) and Fraction of Ammonium-Nitrogen to Total Nitrogen Effluent Loads (γ) by Wastewater Treatment Plants Size-Classes

WWTP	Conservative scenario (ε = 10th and γ = 90th percentile)		
Size-class k[a]	$\varepsilon_{TP,k}$	$\varepsilon_{TN,k}$	γ_k
1	0.30	0.38	0.60
2	0.45	0.54	0.53
3	0.67	0.70	0.24
4	0.86	0.71	0.21
5	0.92	0.73	0.14

[a]Note that the estimates of ε and γ were based on their corresponding data of the selected ~5,560 German WWTPs (~62% of total ~9,000 WWTPs serving ~96% of total residents) (DWA, 2019).

the i-th WWTP is regarded as a nested sub-basin outlet with a corresponding UDF ($UDF_{basin,i}$). Within the scope of this study, the complementary perspective enables a horizontally expanded understanding of the adverse impacts of WWTP-driven pressures on river water quality. Thus, the value of $UDF_{basin,i}$ is calculated as

$$UDF_{basin,i}(t) = \frac{Q_{basin,U,i}}{Q_{basin,U,i} + Q_{R,i}(t)} \tag{3}$$

$$Q_{basin,U,i} = \left(\sum_{j<i} Q_{basin,U,j}\right) + Q_{U,i} \tag{4}$$

where the index j denotes all WWTPs located upstream of the i-th WWTP, and thereby the effluents of individuals WWTPs j are converged to the receiving river of the i-th WWTP.

3.3. Local-Scale Mixing Concentrations of Pollutants in Receiving Waters

We focused on the two key pollutants discharged from WWTPs, total phosphorous (TP) and ammonium-nitrogen (NH4N), whose values are established proxies for ecological risks in receiving river bodies (Ansorge et al., 2020). However, the approach can be applied to any other pollutants of known per-capita loadings discharged from WWTPs. The local mixing concentration for the nutrients C_Z (Z = TP and NH4N) is estimated as

$$C_{Z,i}(t) = \frac{L_{Z,U,i} + L_{Z,R,i}(t)}{Q_{U,i} + Q_{R,i}(t)} \tag{5}$$

where $L_{Z,U,i}$ indicates the loads of nutrient Z discharged from the i-th WWTP, and $L_{Z,R,i}(t)$ means the in-stream loads of nutrient Z at a direct upstream point before $L_{Z,U,i}$ is mixed in a reach. For the local scale of our interest, $L_{Z,R,i}(t)$ is set as 0 to examine the sole impact of point-source nutrient loads discharged from WWTPs on water quality impairment (i.e., "zero" background concentrations for nutrients in receiving river).

Indeed, the value of $L_{Z,U,i}$ ultimately indicates the residual loads' magnitude after influent loads to the i-th WWTP is removed through treatment processes. Note that $L_{Z,U,i}$ for NH4N was estimated from the residual loads of total nitrogen (TN) because of the lack of direct reference on per-capita NH4N load production, by multiplying the fraction of NH4N to TN in the effluents of WWTPs by size-classes (γ_k). We estimated the value of $L_{Z,U,i}$ as follows: First, the total influent loads of TP and TN to the i-th WWTP ($L_{Z,U,i,in}$) were calculated as the product between PE and daily production of the nutrient loads by one person. We employed the national-scale estimates of 1.8 gP/day/person for TP and 11 gN/day/person for TN over Germany (ATV-DVWK, 2003; Nivala et al., 2018; Westphal et al., 2019). Then, the removed nutrient loads of TP and TN at the i-th WWTP were computed by multiplying the total influent loads and the treatment efficiency for each of TP and TN, which was distinct by WWTP size-classes ($\varepsilon_{TP,k}$ and $\varepsilon_{TN,k}$, respectively). Consequently, subtracting the removed loads from the total loads resulted in the residual loads at the end-of-pipe for TP and TN. Respective values of $\varepsilon_{TP,k}$, $\varepsilon_{TN,k}$, and γ_k used in this study are given in Table 2. The results and discussion presented in the main text are based on values of ε and γ equal to their 10th and 90th percentiles, respectively.

3.4. The *k*-*ω* Framework for Risk Assessment

We applied a binary classification of WWTP, "Causing Ecological Risk" or "not Causing Ecological Risk" (WWTP-CER or WWTP-not-CER), to classify each individual WWTP-effluent regarding the proxy indicators UDF, C_{TP}, and C_{NH4N} in the three studied catchments. In this study, the binary category was determined by comparing two probabilities: a probability related to the vulnerability of aquatic ecosystems and a probability that summarizes acceptable situations from a management viewpoint. The former was defined as the exceedance probability of each indicator value equaling the environmental threshold (Figure S2 in Supporting Information S1), required to achieve surface water bodies with at least *Good-Ecological-Status* as set by the EU

WFD (European Commission, 2000). The thresholds applied for the three indicators are UDF* = 3.1% (Büttner et al., 2020), C_{TP}* = 0.1 mgP/L (Heidecke et al., 2015), and C_{NH4N}* = 0.1 mgN/L (BMUB/UBA, 2014). The latter instead represents a threshold on the degree of acceptability of experiencing noncompliance with regulatory thresholds (i.e., conditions of vulnerability of ecosystems to WWTP-effluents), and it is thus named as a management threshold (WWTP*$_{threshold}$; range from 0% to 100% in principle).

When the vulnerable probability for a given indicator is above WWTP*$_{threshold}$, WWTP-CER is assigned for the indicator. Otherwise, WWTP-not-CER is allocated. Note that the WWTP-CER labeling varies with different values of WWTP*$_{threshold}$. Accordingly, we conducted a sensitivity analysis by changing WWTP*$_{threshold}$ from 10% to 100% with an interval of 5%. For a given WWTP*$_{threshold}$, the identical value was applied to all three catchments for both climate scenarios. Results and discussion in the main text of the manuscript refer to a WWTP*$_{threshold}$ of 50% for median case (WWTP*$_{50}$), by knowing WWTP*$_{threshold}$ represents indeed the amount of time (in percent of the total time period, i.e., 30 years in this study) for which we consider a break of the regulatory threshold acceptable. 50% indicates half of the time, which is a typical starting point. The threshold of 50% can therefore be justified as a neutral choice. For a comparative understanding, we additionally refer in Section 4.1 to results coming from a WWTP*$_{threshold}$ of 20% (WWTP*$_{20}$), representing a more stringent management threshold.

By catchments, the resultant binary risk-class for each indicator under a selected climate scenario is presented in a two-dimensional domain consisting of stream-order ω on the abscissa and WWTP size-class k on the ordinate, named as "the k-ω framework." Individual grids in the k-ω framework can contain any metric of interest, such as the number of WWTP-CER or the fraction of WWTP-CER out of their total number.

4. Results and Discussion

4.1. Local-Scale Assessment of WWTPs Causing Ecological Risk

4.1.1. Predicted Status in Far Future

For all three rivers, the number of "WWTPs causing ecological risk (WWTP-CER)" in the far future was the highest for the NH4N indicator and the lowest for the TP indicator (Figure 2). WWTP-CER for NH4N, UDF, and TP indicators accounted for 40% ~ 50%, 35% ~ 45%, and 25% ~ 33% of total WWTPs, respectively, in each river basin (Figures 2a–2c). Distinctive differences of the fraction of WWTP-CER for all three indicators were identified: the Weser exhibited the lowest fraction, whereas the Elbe and the Rhine had similarly higher fraction. The finding suggests that, in the Weser basin, partitioning residential communities served by their own WWTP and location of the WWTPs were better balanced to less compromise water quality of individual receiving streams affected by local-scale WWTP-effluents. Indeed, individuals WWTPs in the Weser served on average ~13%–28% smaller PE (~26K PE/WWTP) than those in the other two catchments. Furthermore, the Weser had ~6% lower fraction of WWTPs in smaller streams ($\omega < 3$) than the other two.

None of the three indicators (NH4N, UDF, and TP) showed uniform distributions of WWTP-CER over the entire ranges of WWTP size-class (k) and stream-order (ω) in the three rivers (Figures 2d–2l). Rather, WWTP-CER for each indicator were strongly clustered in lower-order streams ($\omega < 3$) and spanned across almost all size-classes. Averaged over the three catchments, the fraction of WWTP-CER in $\omega < 3$ was estimated as ~80%, ~64%, and ~56% for NH4N, UDF, and TP, which were greater than that in $\omega \geq 3$ by a factor of ~5, ~4, and ~10, respectively. To be more specific, the WWTP-CER accounted for the greater fraction out of the total WWTPs for a given k-ω combination, as k and ω became greater and lower, respectively. Please refer to Table S1 in Supporting Information S1 for the absolute numbers of WWTP-CER. The results indicated that smaller receiving streams were highly vulnerable to direct effluents of WWTPs, which inevitably related to impairment of river water quality and degradation of aquatic ecosystem integrity. This tendency has been noted previously: Yang, Büttner, Kumar, et al. (2019) show that smaller streams account for ~90% of total streams for which local-scale mixing concentrations for total P and total N loads discharged from WWTPs in the Weser river exceed the desirable concentration to achieve at least *Good-ecological-status* under steady-state median flow. Büttner et al. (2020) also show that across Germany, linearly decreasing ecological status for increasing the local-scale urban discharge fraction is significantly evident in low-order streams, whereas no dependence is found for higher-order streams. Moreover, Rice and Westerhoff (2017) demonstrate that lower-order streams across the continuous US are more likely to be at higher risk of up to four orders of magnitude for endocrine disrupting compounds discharged from WWTPs.

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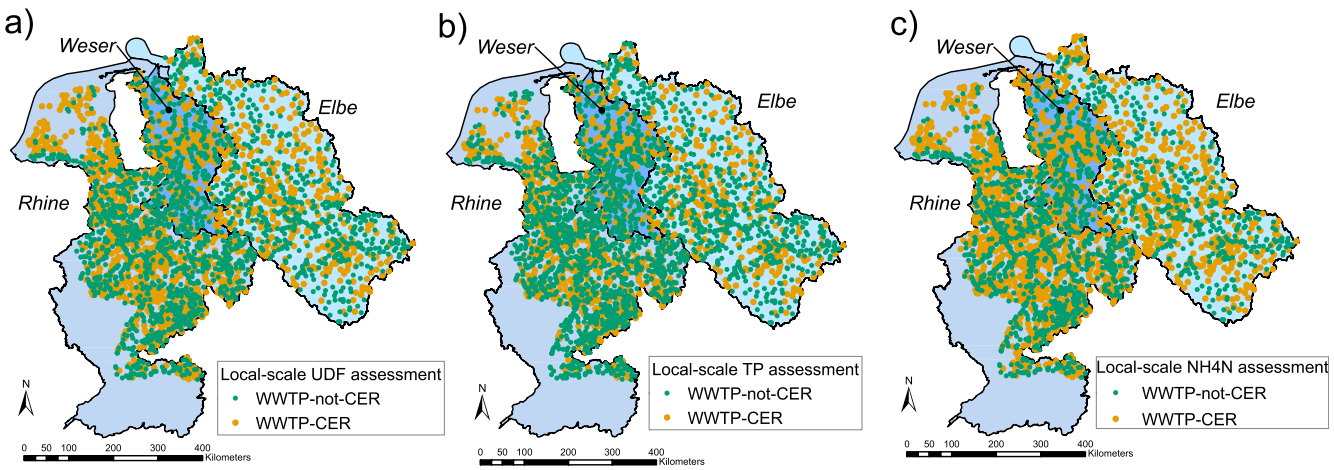

	Far future period	
Catchment	Not-CER	CER
WESER	65%	35%
ELBE	60%	40%
RHINE	56%	44%

	Far future period	
Catchment	Not-CER	CER
WESER	74%	26%
ELBE	67%	33%
RHINE	67%	33%

	Far future period	
Catchment	Not-CER	CER
WESER	58%	42%
ELBE	49%	51%
RHINE	49%	51%

Figure 2. Spatial map and the associated k-ω framework representation under the threshold of WWTP*$_{50}$ for all proxy indicators estimated from a given wastewater treatment plants (WWTP) effluent estimated with $\gamma = 90$th and $\varepsilon = 10$th percentiles at the local scale under the far future scenario. (a–c) For the three catchments, the risk assessments for urban discharge fraction, TP and NH4N concentrations at the local scale are presented as the WWTP-CER (Causing Ecological Risk) in orange and the opposite case (WWTP-not-CER) in green. The fraction of the two cases by catchments is given in the inset tables below the maps. (d–l) The results of the risk assessments shown from panel (a) to panel (c) are distributed by stream-order ω in the abscissa and by WWTP-size k in the ordinate using the same color-coding for the two statuses. The case WWTP-CER > 50% of total is highlighted with a red frame.

It was noteworthy that each indicator exhibited distinct distribution patterns of the WWTP-CER in the k-ω domain, despite the overall common character (i.e., clustering in higher k—lower ω). For a given catchment, WWTP-CER for the UDF indicator were dominantly found in a wider range of ω (from 1 to 5) for intermediate and large sizes ($k = 3 \sim 5$) (Figures 2d–2f). On the other hand, WWTP-CER for the NH4N indicator were more prominent in a relatively narrower range of ω (from 1 to 4) but extended to a smaller size ($k = 2 \sim 5$) (Figures 2j–2l). Remarkably, the dominance of the WWTP-CER for the TP indicator was manifested in the least extent of the k-ω domain (i.e., $k = 2 \sim 5$, $\omega = 1 \sim 2$). The existence of WWTP-CER with smaller k for the NH4N and TP indicators was in good agreement with the absence of regulatory guidelines for the nutrients concentration discharged from smaller WWTPs ($k = 1 \sim 3$) (EEC, 1991). The fraction of WWTP-CER for all indicators was augmented by decreasing WWTP*$_{threshold}$ (Figure S3 in Supporting Information S1). For instance, compared to WWTP*$_{50}$, the lower threshold WWTP*$_{20}$ increased WWTP-CER by ~15.5% (for UDF and TP) and ~12% (for NH4N) on average over the three catchments. The fraction of WWTP-CER remained the highest for NH4N, followed by UDF and TP. The additional WWTP-CER manifested themselves in the k-ω domain as a gradual expansion of the range of ω affected (Figure S4 and Table S2 in Supporting Information S1). While the expansion for water quality indicators (TP and NH4N) spanned across all size-classes, that for water quantity metric (UDF) occurred prominently in the largest size-class. The distinct patterns suggested different responses of water quantity and quality indicators to sensitivity of thresholds defining WWTP-CER. Accordingly, complementary examination of both aspects can assist a comprehensive assessment of WWTP-effluents' effects on receiving rivers.

4.1.2. Differences Induced by Climate Change

To understand the effects of climate-induced dynamics of streamflow on increasing WWTP-CER, we calculated the differences of the fraction for individual indicators between the two periods (Figure 3). Most of the differences were found in smaller streams receiving effluents of intermediate-size WWTPs (i.e., $\omega < 4$, $k = 2 \sim 4$) for all three catchments. The common pattern revealed that the largest WWTPs already impaired small and intermediate streams during the wetter historical period due to insufficient dilution capacity. For all three indicators, the fraction of WWTP-CER in the far future mostly increased for the Elbe and the Rhine, but decreased

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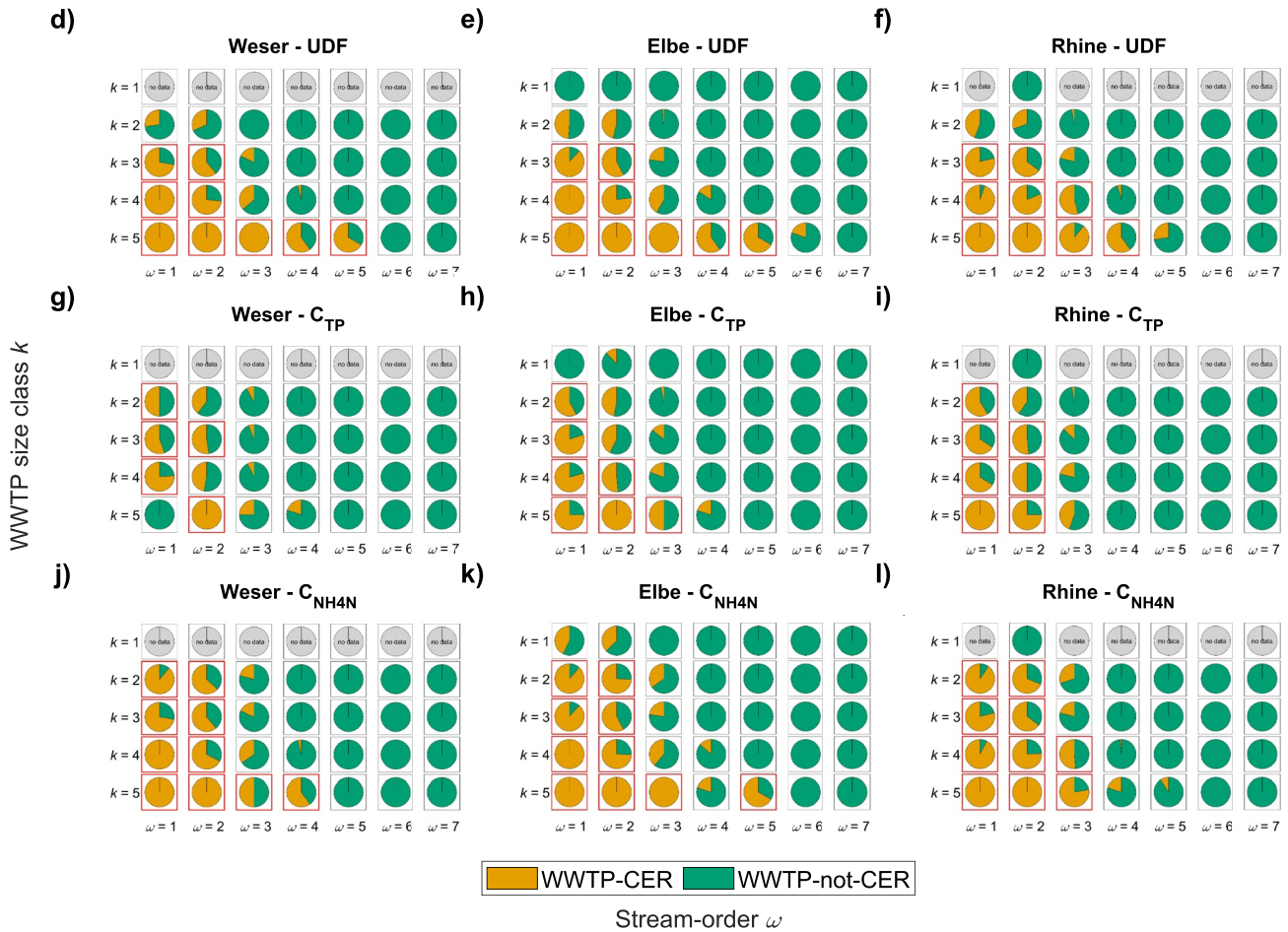

Figure 2. (Continued)

for the Weser. The trend was remarkably consistent in the lower management threshold $WWTP^*_{20}$ (Figure S5 in Supporting Information S1). Nonetheless, the decreasing magnitude was weakened for the Weser, while the increasing degree for the Elbe and the Rhine was enhanced with an expansion of WWTP-CER toward smaller k and greater ω. These results suggest that the management threshold plays a crucial role for assessing the ecological risks anticipated in the far future, and thus for establishing suitable action plans for mitigating the risks. Thus, this study pinpoints the importance of applying a reasonable management threshold into the catchments of interest, based on sufficient discussion among diverse stakeholders involved in managing river basin systems.

Moreover, it should be noted that the lower number of WWTP-CER in the far future should not be interpreted as a result of wetter hydrological regime in the Weser, because the shape of the entire flow duration curve affects the estimated exceedance probability (Text S3 and Figure S6 in Supporting Information S1). Thus, the finding may be attributable to the less frequent occurrence of extreme dry streamflow in the far future in the Weser River (Table 1). Basin-scale analyses of changes of hydro-meteorological properties and their linkage to the effects of WWTP-effluents on river water quality are worthy of future studies.

4.2. Multi-Criteria Risk Assessment at the Local Scale

Multi-criteria assessment of WWTP-CER was conducted for the same results presented in Figure 2, by simultaneously evaluating the risk-status for all three indicators and thus categorizing four risk types from none to all indicators (Figures 4a, 4c, and 4e). Around half of the total WWTPs for each catchment (47% ~ 56%) were estimated as not causing risks for all three indicators (i.e., WWTP-not-CER). In contrast, the WWTP-CER for all indicators accounted for ~23% for the Weser and ~30% for both the Elbe and the Rhine. The estimated fraction

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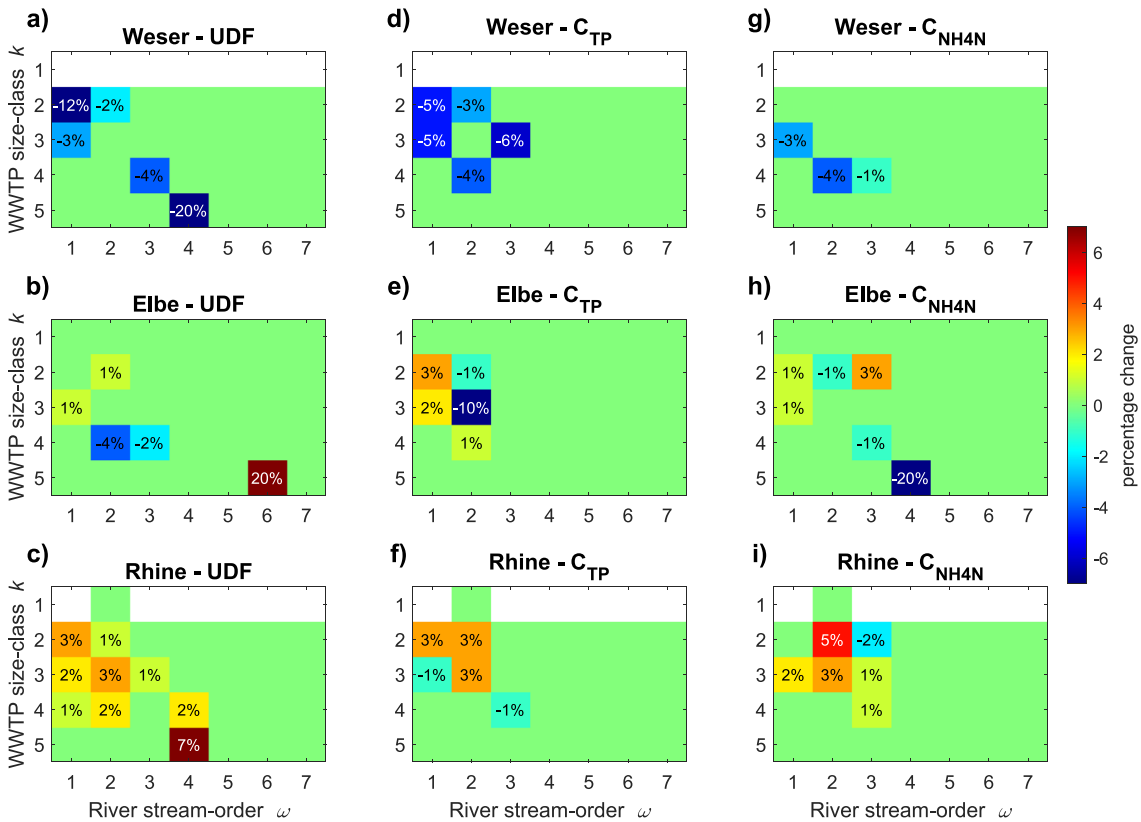

Figure 3. Changed % of wastewater treatment plants causing ecological risks (WWTP-CER) from historical to the far-future scenario at the local scale based on $WWTP^*_{50}$ threshold. Each % in difference was distributed by stream-order ω in the abscissa and by WWTP-size k in the ordinate for three proxy indicators (by columns) in each catchment (by rows). Presented results were based on the WWTP-effluents estimated with $\gamma = 90$th and $\varepsilon = 10$th percentiles.

for each of the three catchments was slightly lower than that of WWTP-CER solely for the TP indicator, indicating that the risk-status for the TP mostly coincided with that for the other two indicators. The finding suggests that prioritizing technological advancement for the WWTP-CER for the TP can yield remarkable reductions of the most severe risk from all three indicators as well as to resolve the water quality problem from the TP itself.

More than 90% of the WWTP-CER for all three indicators were clustered in the lower-order streams ($\omega < 3$) for the three catchments (Figures 4b, 4d, and 4f, and Table S3 in Supporting Information S1). Distributions of the WWTP-CER for one or two indicators were also strongly skewed toward the smaller streams. On the other hand, less than 20% of the WWTPs without any risks were located in the streams of lower-orders. The findings evidently showed that smaller streams are highly endangered by the WWTP-effluents for what concerns both quantity and quality indicators at the local scale. Our results agree with a recent work of Büttner et al. (2022), reporting the much more vulnerable status of smaller streams to WWTP-effluents at the entire European scale.

The projected vulnerability of lower-order streams to WWTPs-effluents under climate change obviously underpins the urgent needs for planning and implementing adaptation measures to protect smaller streams, which are biogeochemically more reactive and ecologically more diverse (Richardson, 2019; Wohl, 2017). A typical transport route of urban wastewater and therein contaminants from sources, modulator, and to receiver (i.e., multiple households, a WWTP, and a river reach, respectively) suggests the three main stages to which distinct adaptation measures shall be applied. Note that again, here we focus on discussing feasible adaptation pathways. Thereby, any specific cost-effective analysis for the application of sole or combined measure(s) is outside the scope of this study.

First, controlling domestic sources of nutrients can contribute to a decline in total input loads incoming to a WWTP (people-lifestyle adaptation). Main sources of the nutrients (N and P) in daily life are attributed to foods and cleansing products. Knowing the required essential intake of N and P ingredients for human nutrition balance,

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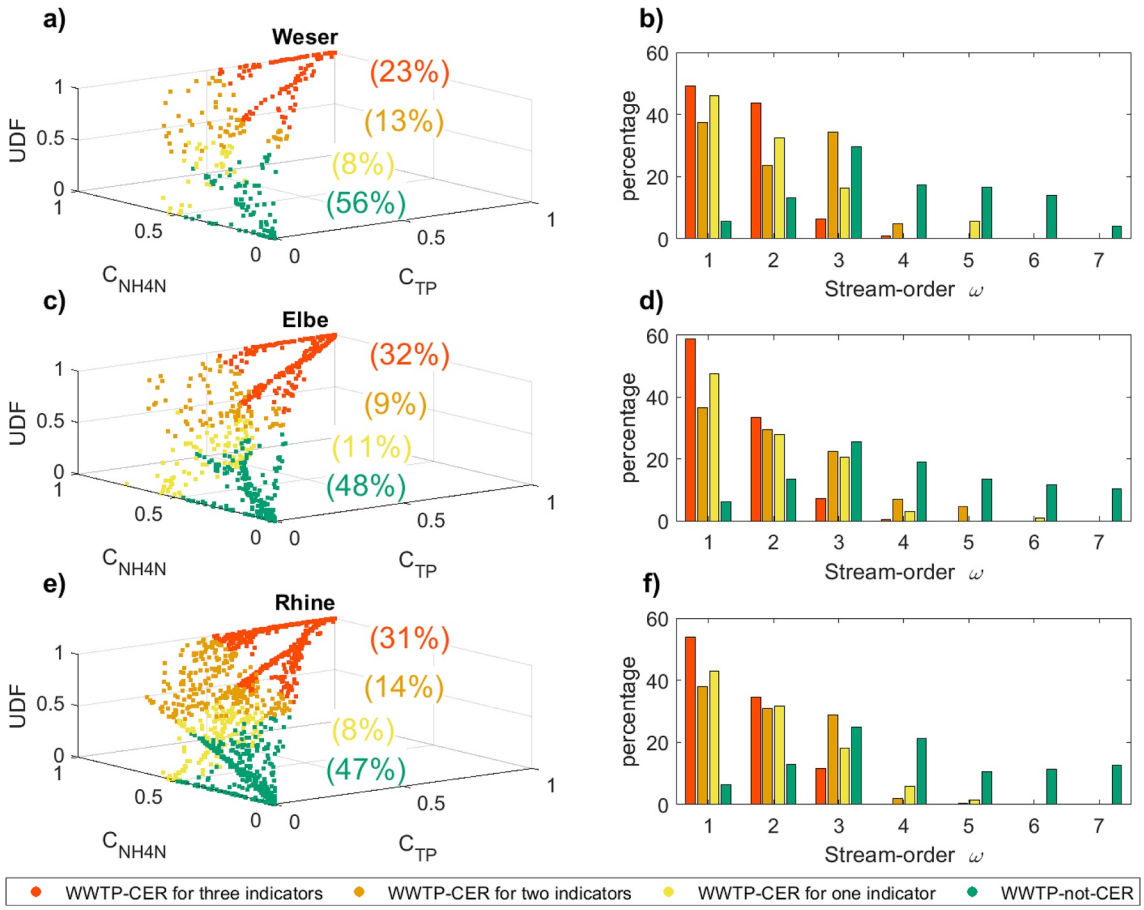

Figure 4. Multi-criteria assessment to categorize wastewater treatment plants (WWTPs) by four levels of risk types at the local scale under the far future scenario. Note that the assessment used the same results presented in Figure 2 for (a) the Weser, (c) the Elbe, and (e) the Rhine. The four levels of risk types are defined as the number of indicators causing ecological risks (zero, one, two, and three indicators at risk are displayed in green, yellow, orange, and red colors, respectively). Numbers in parentheses indicate the fraction of WWTPs by each category out of total WWTPs (~480 in the Weser, ~900 in the Elbe, and ~1,820 in the Rhine). Distribution for each level of risk type over stream-orders ω are given in % for (b) the Weser, (d) the Elbe, and (f) the Rhine.

the associated adaptation strategy needs to be targeted for advising environmentally sustainable food consumption patterns, such as lower frequency of taking high protein nutritional supplements (Wang et al., 2019). Innovative developments and replacement of cleansing and consumer products currently containing phosphate with functionally equivalent but environmentally less polluted chemical compounds are required (Chong et al., 2019).

Secondly, upgrading treatment technology to remove more nutrients in the WWTPs can reduce nutrient loads outgoing at the end-of-pipe (engineered-system adaptation). Smaller WWTP-CER can be improved by installing the advanced tertiary treatment technology (Mažeikienė, 2019; Rahman et al., 2016). On the other hand, remarkable breakthroughs in developing new treatment technology is highly vital for resolving the expected risks of larger WWTPs, since they already deploy the most advanced technology at present standards (Carvalho et al., 2019; Stoddard et al., 2003).

Last adaptation measures can be formulated by redirecting travel paths of WWTP discharges before reaching its receiving river stream (effluent-pathway adaptation). For instance, building constructed wetlands in combination with the engineered treatment systems can delay the arrival time and amount of nutrient loads discharged from WWTPs to receiving river bodies (Bolton et al., 2019; Wu et al., 2015). Particularly for larger WWTP-CER discharging to a smaller stream, dispersing a proportion of its effluents into near larger streams with greater dilution capacity can distribute the associated stresses induced by urban wastewater that are currently entirely borne by smaller stream reaches. Accordingly, smaller streams can be less compromised by larger WWTP-effluents at the local scale.

In addition to the aforementioned measures for adaptation in socio-economic and technologic manners, adaptive adjustments need to be conducted regarding environmental policies and regulations of WWTP-effluents. At present, EU-scale regulations of nutrient concentrations at the end-of-pipe are defined by technology-based approaches, and thus applied only for two largest size-classes of WWTPs (EEC, 1991). Over the three catchments, ~1,570 WWTPs (~49%) of the total ~3,200 were legitimate targets for applying the regulations. In the case of the TP indicator, ~91% of total WWTPs under the regulations satisfied the requirements for WWTP-discharged concentration to be lower than the defined thresholds (i.e., 2 and 1 mgP/L for size-classes 4 and 5, respectively). Nevertheless, the remarkably high compliance of larger WWTPs with the requirements did not completely ensure water quality protection of their receiving river reaches, as shown by the evaluated ~30% of the larger WWTPs at local-scale risk. The finding highlighted the need for incorporating hydrological conditions and water-quality-based effluent standards into the original regulations purely based on technology, as actively acknowledged since a few decades (Çelebi et al., 2021; Gursoy-Haksevenler et al., 2021; Ragas et al., 2005). The proposed k-ω framework stands as a crucial solution to conceptualize and systematize the combined technology- and water-quality-based discharge limits at the scale of entire river networks.

4.3. Basin-Scale Risk Assessment

We evaluated the cumulative effects of water flux in the WWTP-effluents by calculating the basin-scale UDF under the *far future* scenario (Equation 3) at the points of each WWTP location along converging flow paths. The basin-scale risk for UDF was assessed under the threshold of $WWTP^*_{50}$. The assessment criteria based on the explicit WWTP locations and the $WWTP^*_{50}$ threshold under the far future scenario aimed to be consistent with the perspectives used for the local-scale risk assessment, which is primarily described in the main text. Note that we newly introduced the perspective of identifying the WWTP location facing risk for UDF at the basin scale, termed as "WWTP-LFR (Location Facing Risk)," instead of WWTP-CER. This helps to differentiate the basin-scale results from the local-scale ones related to the direct effluent of a given WWTP.

The cumulative effects for UDF contributed to the increase of WWTP-LFR for all three catchments (Figures 5a–5c). The incremental fraction was estimated as ~34% for the Weser, and ~40% for the Elbe and the Rhine, out of all WWTP locations for each catchment. It was attributed to cases that the UDF risk status was shifted from *not-causing-risk under local-scale* to *causing-risk under basin-scale*, which is hereafter denoted as "$\Delta Risk_{BS}$." The finding suggests that the locally sufficient dilution capacity of river discharge at a given location can be jeopardized by the cumulated magnitude of WWTP-effluents along converging flow paths.

The WWTP-LFR with $\Delta Risk_{BS}$ were not evenly distributed across stream-orders ω and size classes k (Figures 5d–5f). Rather, they were strongly clustered in $\omega > 3$, accounting for 70% ~ 80% of total cases for each catchment. The finding reasonably mirrored the fact that the large fraction of WWTPs in $\omega < 4$ was already evaluated as WWTP-CER under the local scale. Within higher-order streams ($\omega > 3$), the greatest fraction of the WWTP-LFR with $\Delta Risk_{BS}$ was found in $k = 4$ for all three catchments, because of the largest fraction of WWTPs with $k = 4$ among all size classes. However, for a given k across $\omega = 4 \sim 7$, the fraction of the WWTP-LFR with $\Delta Risk_{BS}$ to the total number of WWTP-LFR was the largest in $k = 5$ (90% ~ 95%), followed by $k = 4$ (78% ~ 90%) for all three catchments. The findings can help in understanding how water quality pollution induced by less reactive pollutants in WWTP emissions would be propagated downstream as they flow along stream networks (Schmidt et al., 2020). Furthermore, the core principle of basin-scale assessment conducted here can be transferred to discern cascading degradation downstream for river water quality induced by WWTP-discharged pollutants, particularly those which are not affected by in-stream processes (e.g., retention or release). To the best of our knowledge, the accumulated effect of urban wastewater pressure from WWTPs has been explored in a spatially implicit approach as a simple sum of water flux over large catchments (Abily et al., 2021; Karakurt et al., 2019; Keller et al., 2014). Conversely, the application of the k-ω framework allows for a spatially explicit evaluation of cumulative effluents for multiple WWTPs within a catchment.

4.4. Limitations of This Study

We acknowledge inherent uncertainties in our assumptions of spatially homogeneous and temporally constant per-capita daily production of water and nutrients for people residing in the three study catchments. On the other hand, we explicitly accounted for the spatial and temporal dynamics of hydrological flow regimes in the

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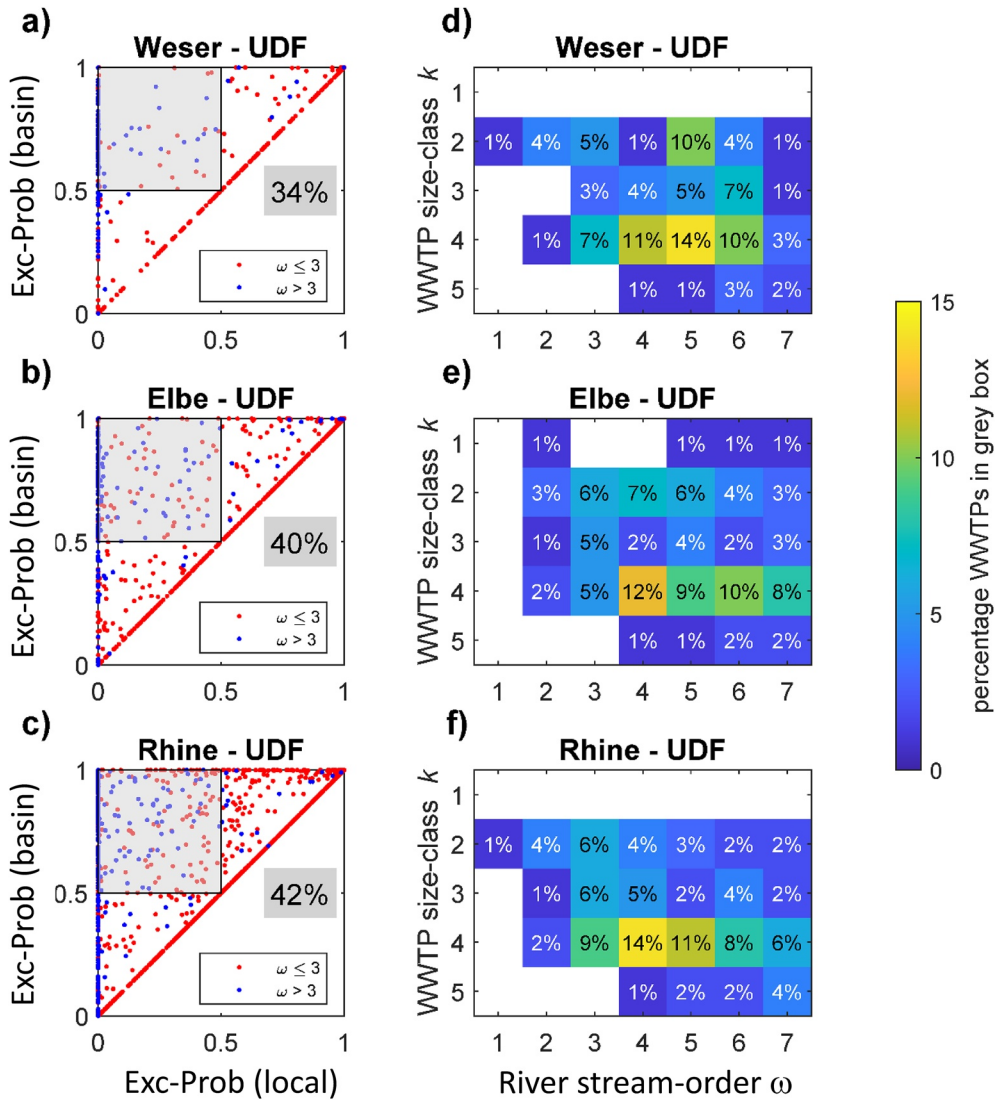

Figure 5. Assessment of the wastewater treatment plant location facing risk (WWTP-LFR) for urban discharge fraction (UDF) at the basin scale, with accumulated aspect under the far future scenario. (a–c) Comparison of the exceedance probability having UDF* for individual WWTPs, between the local scale and the basin scale (i.e., Exc-Prob (local) and Exc-Prob (basin), respectively) for the three river basins. Gray regions represent the case of not-causing-risk under the local scale but causing-risk under the basin scale (i.e., defined as $\Delta Risk_{BS}$) by applying the $WWTP^*_{50}$ threshold. The fraction of WWTPs within the gray regions is given in each inserted number. (d–f) Distribution of the WWTPs within each gray region by stream-order ω in the abscissa and by WWTP-size k in the ordinate, for the three river basins.

receiving rivers for both the historical and far future periods. This was intended in the scope of this study, which primarily aimed to develop an analytical, generic, and transferable framework for ecological risk assessments of WWTP-effluents in any river basin for any time period under expected global changes. Thus, it was inevitable to reduce the spatiotemporal complexity in the emission to receiving water perspective, in turn focusing on changing river hydrological conditions that influence multiple sectors, including drinking water availability, under global changes. Simultaneously, we noted that residents in the selected three catchments shared and will display strong similarities in lifestyles and behavioral characters, such as food and water consumption, with the situation in other European countries (EurEup, 2020; Wang et al., 2019). This suggests that the main results of this study (e.g., at the local scale, smaller streams are expected to be more compromised by effluents of greater WWTPs) will be confirmed, while the fraction of WWTP-CER only will change, as future-projected values for the resource consumption per person will be explicitly reflected in the estimates. Note that uncertainty analyses for the three catchments are outside the scope of this study, rather anticipated in following research.

Another source of uncertainty for our results might be simulation data obtained from only one hydrological model and referring to the climate condition simulated by a single GCM considering only the most extreme scenario regarding RCP and warming level. Uncertainty in each simulation result from different combinations between climate change and hydrological models exists. Hence, additional processes that quantify the uncertainty within a model or across different models (e.g., ensemble analyses) are importantly conducted in the fields of modeling future-projected climatic or hydrological properties. However, it should be noted that our scope is not to provide the most accurate assessment of the ecological risk of WWTP-effluents under climate change, but to underpin the proposed risk assessment framework based on a plausible and reliable scenario for projected climate change. For a given set of climate-hydrological models, additional intermediate RCP and lower warming level conditions will lead to an overall less fraction of WWTP-CER over a river network, but not necessarily proportional across stream-orders due to nonlinearity in the coupled meteorological-hydrological processes. Decisions on the types of climate and hydrological models and the conditions of RCP and warming level are not crucial in the scope of this study, which aims at presenting the main features and an application of the risk assessment framework.

4.5. Future Outlook

The analytical and generic risk assessment framework presented here can be flexibly transferred to larger spatial extents, more elaborated temporal dynamics, and other proxies regarding ecological or human health related impacts. Continental or global-scale extensions of the framework will allow for evaluating the environmental policy on discharges from WWTPs or for assessing progress to meet legally binding international commitments to water quality protection in a consistent way. For example, all EU states are required to achieve the EU WFD's goal to make all river water bodies be at least with good-ecological status by 2027. To mitigate adverse impacts of WWTP-effluents on this goal, the EU UWWTD has been implemented commonly across the EU to regulate WWTP-effluents categorized by the WWTP-sizes related to treatment levels. In fact, the European countries spanning the study catchments are categorized in the uppermost class regarding the fraction of people connected to the most advanced tertiary level treatment, which is 15% $\sim$ 30% higher than the EU-average as of 2017 (EEA, 2020b). The EU UWWTD stipulates the most stringent regulations for WWTPs deploying the tertiary level in terms of nutrient concentrations discharged from WWTPs. Nonetheless, as this study showed, even the most stringent regulations on WWTP-effluents can pose ecological risks especially to smaller streams. Thus, our findings underpin the necessity of additional systematic categories to elaborate the regulations, such as by following the river network hierarchy. Adverse effects highlighted here may be more prominent in other European catchments spanning countries with lower connection of people to tertiary level treatment. Further analyses at the EU scale are required to support or contradict this hypothesis; such efforts are currently underway.

In addition, global-scale extension is necessary and promising. Although the global south have released wastewaters to the environment without appropriate treatment, the majority of river streams in the region are still in good condition (UNEP, 2016). However, facing a worldwide growing population, the number of births in the global south will continue to increase until 2050, and thereby waste and wastwater production and economic activities will be highly accelerated (UN, 2019). If the current options of wastewater management remain the same over the upcoming decades, further pollution in rivers are a matter of time and already-polluted streams lose an opportunity for restoration. This prospect is strongly coupled with geopardizing the achievement of SDG 6 which ultimately aims for universal access to clean water. To prevent the undesirable outcome, upscaling the presented framework globally can play a pivotal role in assessing progress toward the accomplishment of SDG 6 by integrating open-source global-scale data for river networks, WWTPs (Jones et al., 2021; Lehner et al., 2008) as well as projected data of population, water and food consumption patterns, and hydrological modeling (Krysanova et al., 2020; UN, 2019; Wang et al., 2019). Indeed, the Horton-Strahler stream-ordering scheme is consistently applicable to any river network in any geographical region. On the other hand, a category of WWTP size classes is highly dependent on environmental regulations by countries and/or continents. As to our knowledge there is no internationally agreed size-class definition, the classification used in this paper can serve as a reasonable starting point. We note that what matters for the k-ω-framework is the application of a consistent categorization to all WWTPs-data.

This study laid the foundation toward a holistic decision support system by providing a framework to identify hotspots of impact by WWTP-effluents and the discharges of treated wastewater into streams and rivers. To achieve such a holistic assessment, the following steps are essential: (a) multiple scenario analyses to guide for

reducing WWTP-CER and (b) cost-effectiveness analysis for each scenario to inform decision options. Note that a wide range of measures exist in formulating alternative scenarios and are inevitably dependent on regional conditions regarding technology, economy, policy, and residents' lifestyles (UNEP, 2016). Exemplary technical measures are to upgrade treatment technology, change the number of people served, redirect sewage to already existing or newly built WWTPs with greater dilution capacity, and deploy nature-based solutions, such as constructed wetlands, to delay the arriving time of WWTP-effluents to river networks. A typical but complex, nontechnical societal measure can be to legislate household sources control policy by regulating per-capita consumption of water, harmful substances, and/or supplementary protein foods. Future studies on scenario-based analyses are expected to demonstrate the suitability and effectiveness of individual or coupled adaptation measures to reduce the adverse effects of WWTP-effluents on smaller streams under climate change. Future studies need to incorporate other types of urban sanitary treatment systems besides centralized WWTPs into the k-ω assessment framework for a comprehensive assessment of household-produced wastewater impacts on river ecological risks. Septic tanks could be prioritized since they are typical examples of decentralized wastewater treatment systems that are complementary to centralized WWTP systems. Such studies can contribute to secure water quality and the integrity of aquatic ecosystems under global changes.

5. Conclusions

This study aimed to develop a new, analytical, and transferable framework providing first-order diagnosis for ecological risks caused by individual WWTP-effluents on receiving streams and rivers at the catchment scale under climate change. The framework consisted of the combination of two fundamental characteristics for the number of people served by WWTPs (WWTP-size k) and for river network hierarchy (stream-order ω), named the k-ω framework. Three environmental proxy indicators of concern were analyzed: The local-scale concentrations of dominant wastewater components (total phosphorus and ammonium-nitrogen; P and NH4N) and the local fraction of treated wastewater pressures related to natural flows (UDF). Our novel approach for risk analyses synthesizes multiple data on river network, WWTP-properties, modeled streamflow under a projected climate scenario, and socioeconomic characteristics. Focusing on the scope of this study, we only referred to the most extreme warming scenario in the far future and its corresponding simulated streamflow data from a mesoscale hydrological model. The historical period was compared as a reference condition.

We selected three large central European catchments as study areas (Rhine, Elbe, and Weser). The Rhine and the Elbe possess noticeably contrasting characteristics in terms of climatic, hydrological, and river ecosystem responses, whereas surprisingly common behaviors exist in the coupled human-WWTP systems and daily lifestyles. For all indicators, lower-order streams in the three catchments were more threatened from WWTP-effluents, especially by greater sizes. This finding suggests the urgent needs for organizing next actions to protect smaller streams that play crucial roles in providing sources of water flows, and in keeping aquatic ecosystems integrity. The lowest fraction of WWTP-CER for all indicators in the Weser underpinned that less centralized distribution of people and wastewater infrastructures might yield less ecological risks in receiving rivers. Among the three considered indicators, the risk for NH4N was the highest due to higher contributions of smaller-sized WWTPs in all catchments. This result urges to develop legislative regulations for effluents from smaller WWTPs, which are not currently defined. Changing hydrological conditions from historical to future climates resulted in catchment- and indicator-specific differences in the fraction of WWTP-CER. Nonetheless, most of the differences were identified in smaller streams receiving effluents of intermediate-sized WWTPs.

In addition to the perspectives for individual indicators, multi-criteria risk assessment approaches, such as the one presented here, can be used to design, plan, and implement tailored solutions to reduce hotspots of environmental pressures induced by WWTP-effluents. The holistic decision support system can be applicable to any geophysical regions spanning any climate zones. A strategic prioritization of technical adaptations for WWTP-CER can be facilitated by categorizing the type of ecological risks. In this context, the proposed k-ω framework based on a data-model synthesis has significant potential by serving as a tool to screen the effects of the new policy and/or evaluate currently implemented policies. The k-ω framework can also be applied for newly emerging contaminants in WWTP-effluents, and thus help to advance the science base for setting environmental standards. Furthermore, the k-ω framework allows for scenario-based analyses by explicitly taking into consideration the growing population and changing lifestyles over the upcoming decades, and thereby facilitates the assessment of progress

toward continental- and global-scale goals for securing water resources and aquatic ecosystems including their biodiversity.

Data Availability Statement

The daily streamflow data simulated using the mHM is available through Marx et al. (2018), Thober et al. (2018), and Samaniego et al. (2019). Data set of the EU Hydro and the EEA WWTPs are publicly available.

Acknowledgments
This study was funded by the Helmholtz Climate Initiative Project (www.helmholtz-klima.de) being conducted across the Helmholtz Association. We thank Gert Schwentner from the German Association for Water, Wastewater and Waste (DWA) for providing data on treatment efficiency of German WWTPs. We also acknowledge data sets of the mHM simulations from EDgE project (http://edge.climate.copernicus.eu/; Projection data at https://doi.org/10.24381/cds.ccf781a2) of the Copernicus Climate Change Service. We thank the Editor (Carole Dalin), the Associate Editor (anonymous), and two reviewers (Serena Ceola and anonymous) for constructive and valuable comments, which have greatly helped to improve the manuscript. Open Access funding enabled and organized by Projekt DEAL.

References

Abily, M., Acuña, V., Gernjak, W., Rodríguez-Roda, I., Poch, M., & Corominas, L. (2021). Climate change impact on EU rivers' dilution capacity and ecological status. *Water Research*, *199*, 117166. https://doi.org/10.1016/j.watres.2021.117166

Ansorge, L., Stejskalová, L., & Dlabal, J. (2020). Effect of WWTP size on grey water footprint—Czech Republic case study. *Environmental Research Letters*, *15*(10), 104020. https://doi.org/10.1088/1748-9326/aba6ae

Arden, S., & Jawitz, J. W. (2019). The evolution of urban water systems: Societal needs, institutional complexities, and resource costs. *Urban Water Journal*, *16*(2), 92–102. https://doi.org/10.1080/1573062X.2019.1634109

ATV-DVWK. (2003). Arbeitsblatt A-198. Vereinheitlichung und Herlegung von Bemessungswerten für Abwasseranlagen. (Translation: Standardization and Derivation of Design for Sewer Systems).

BBC. (2018). Heatwave leads to increased demand for water. UK Heatwaves. Retrieved from https://www.bbc.com/news/uk-northern-ireland-44676684

BMUB/UBA. (2014). *Water resource management in Germany - Part 2: Water quality*. The environment, nature conservation, building and nuclear safety (BMUB) and the German federal environmental agency (UBA). Retrieved from https://www.umweltbundesamt.de/publikationen/water-resource-management-in-germany-part-2

Bolton, L., Joseph, S., Greenway, M., Donne, S., Munroe, P., & Marjo, C. E. (2019). Phosphorus adsorption onto an enriched biochar substrate in constructed wetlands treating wastewater. *Ecological Engineering*, *142*, 100005. https://doi.org/10.1016/j.ecoena.2019.100005

Büttner, O., Jawitz, J. W., Birk, S., & Borchardt, D. (2022). Why wastewater treatment fails to protect stream ecosystems in Europe. *Water Research*, *217*, 118382. https://doi.org/10.1016/j.watres.2022.118382

Büttner, O., Jawitz, J. W., & Borchardt, D. (2020). Ecological status of river networks: Stream order-dependent impacts of agricultural and urban pressures across ecoregions. *Environmental Research Letters*, *15*(10), 1040b1043. https://doi.org/10.1088/1748-9326/abb62e

Carvalho, L., Mackay, E. B., Cardoso, A. C., Baattrup-Pedersen, A., Birk, S., Blackstock, K. L., et al. (2019). Protecting and restoring Europe's waters: An analysis of the future development needs of the Water Framework Directive. *Science of the Total Environment*, *658*, 1228–1238. https://doi.org/10.1016/j.scitotenv.2018.12.255

Çelebi, S., Yetis, U., & Ünlü, K. (2021). A practical approach for the determination of environmental quality standards-based discharge limits: The case of Tersakan sub-basin of Yeşilırmak River in Turkey. *Environmental Science & Pollution Research*, *28*(29), 38730–38748. https://doi.org/10.1007/s11356-021-13350-6

Ceola, S., Laio, F., & Montanari, A. (2019). Global-scale human pressure evolution imprints on sustainability of river systems. *Hydrology and Earth System Sciences*, *23*(9), 3933–3944. https://doi.org/10.5194/hess-23-3933-2019

Chong, J., Dominish, E., Tjandraatmadja, G., Prentice, E., Retamal, M., & Mitrovic, S. (2019). *Ecosystem impacts of phosphorus and surfactants in consumer products*. Institute for Sustainable Futures, University of Technology.

DWA. (2019). *Leistungsnachweis kommunaler Kläranlagen - Klärschlammanfall. [32. Proof of achievement: Municipal sewage treatment plants]* (p. 32). German Association for Water Management, Wastewater and Waste.

EC& EEA. (2020). *Water uses to cope with heat waves in cities*. Climate-ADAPT. Retrieved from https://climate-adapt.eea.europa.eu/metadata/adaptation-options/water-uses-to-cope-with-heat-waves-in-cities

EEA. (2015). *Freshwater quality*. European Environment Agency (EEA) Report.

EEA. (2019). UWWTD database version 5. Retrieved from https://www.eea.europa.eu/data-and-maps/data/waterbase-uwwtd-urban-waste-water-treatment-directive-6

EEA. (2020a). EU-hydro – river Network database, version 1.0. Retrieved from https://www.eea.europa.eu/data-and-maps/data/external/eu-hydro-2013-river-network.0

EEA. (2020b). *Urban waste water treatment in Europe*. European Environment Agency. Retrieved from https://www.eea.europa.eu/data-and-maps/indicators/urban-waste-water-treatment/urban-waste-water-treatment-assessment-5

EEC. (1991). *Council directive of 21 May 1991 concerning urban waste water treatment (91/271/EEC)*. Official Journal of the European Communities. Retrieved from http://ec.europa.eu/environment/water/water-urbanwaste/index_en.html

EurEup. (2020). The governance of water services in Europe.

European Commission. (2000). *Directive 2000/60/EC of the European parliament and of the council of 23 October 2000 Establishing a framework for community action in the field of water policy*. Official Journal of the European Communities. Retrieved from http://ec.europa.eu/environment/water/water-framework/index_en.html

Freni, G., Mannina, G., & Viviani, G. (2010). Emission standards versus immission standards for assessing the impact of urban drainage on ephemeral receiving water bodies. *Water Science and Technology*, *61*(6), 1617–1629. https://doi.org/10.2166/wst.2010.887

Glassman, M. (2018). *Residents asked to space out water usage during heat wave*. Allen Media Broadcasting. Retrieved from https://www.kezi.com/content/news/SUB-is-asking-people-to-space-out-water-usuage-ahead-of-heatwave--487955431

Gücker, B., Brauns, M., & Pusch, M. T. (2006). Effects of wastewater treatment plant discharge on ecosystem structure and function of lowland streams. *Journal of the North American Benthological Society*, *25*(2), 313–329. https://doi.org/10.1899/0887-3593(2006)25[313:eowtpd]2.0.co;2

Gursoy-Haksevenler, B. H., Atasoy-Aytis, E., Dilaver, M., Yalcinkaya, S., Findik-Cinar, N., Kucuk, E., et al. (2021). A strategy for the implementation of water-quality-based discharge limits for the regulation of hazardous substances. *Environmental Science & Pollution Research*, *28*(19), 24706–24720. https://doi.org/10.1007/s11356-020-10220-5

Hardenbicker, P., Rolinski, S., Weitere, M., & Fischer, H. B. (2014). Contrasting long-term trends and shifts in phytoplankton dynamics in two large rivers. *International Review of Hydrobiology*, *99*(4), 287–299. https://doi.org/10.1002/iroh.201301680

Hardenbicker, P., Weitere, M., Ritz, S., Schöll, F., & Fischer, H. (2016). Longitudinal plankton dynamics in the Rivers Rhine and Elbe. *River Research and Applications*, *32*(6), 1264–1278. https://doi.org/10.1002/rra.2977

Hari, V., Rakovec, O., Markonis, Y., Hanel, M., & Kumar, R. (2020). Increased future occurrences of the exceptional 2018–2019 Central European drought under global warming. *Scientific Reports*, *10*(1), 12207. https://doi.org/10.1038/s41598-020-68872-9

HDR. (2002). *Handbook of public water systems* (2nd edn.). John Wiley & Sons.

Heidecke, C., Hirt, U., Kreins, P., Kuhr, P., Kunkel, R., Mahnkopf, J., et al. (2015). Endbericht zum Forschungsprojekt "Entwicklung eines Instrumentes für ein flussgebietsweites Nährstoffmanagement in der Flussgebietseinheit Weser". Thünen Report. (Vol. *21*).

Hempel, S., Frieler, K., Warszawski, L., Schewe, J., & Piontek, F. (2013). A trend-preserving bias correction - The ISI-MIP approach. *Earth System Dynamics*, *4*(2), 219–236. https://doi.org/10.5194/esd-4-219-2013

Hoffmann, T. O., Baulig, Y., Fischer, H., & Blöthe, J. (2020). Scale breaks of suspended sediment rating in large rivers in Germany induced by organic matter. *Earth Surface Dynamics*, *8*(3), 661–678. https://doi.org/10.5194/esurf-8-661-2020

Horton, R. E. (1945). Erosional development of streams and their drainage basins; hydrophysical approach to quantitative morphology. *The Geological Society of America Bulletin*, *56*(3), 275–370. https://doi.org/10.1130/0016-7606(1945)56[275:EDOSAT]2.0.CO;2

Hulley, G. C., Dousset, B., & Kahn, B. H. (2020). Rising trends in heatwave metrics across southern California. *Earth's Future*, *8*(7), e2020EF001480. https://doi.org/10.1029/2020EF001480

Ionita, M., & Nagavciuc, V. (2020). Forecasting low flow conditions months in advance through teleconnection patterns, with a special focus on summer 2018. *Scientific Reports*, *10*(1), 13258. https://doi.org/10.1038/s41598-020-70060-8

Jones, E. R., van Vliet, M. T. H., Qadir, M., & Bierkens, M. F. P. (2021). Country-level and gridded estimates of wastewater production, collection, treatment and reuse. *Earth System Science Data*, *13*(2), 237–254. https://doi.org/10.5194/essd-13-237-2021

Kamjunke, N., Rode, M., Baborowski, M., Kunz, J. V., Zehner, J., Borchardt, D., & Weitere, M. (2021). High irradiation and low discharge promote the dominant role of phytoplankton in riverine nutrient dynamics. *Limnology & Oceanography*, *66*(7), 2648–2660. https://doi.org/10.1002/lno.11778

Karakurt, S., Schmid, L., Hübner, U., & Drewes, J. E. (2019). Dynamics of wastewater effluent contributions in streams and impacts on drinking water supply via riverbank filtration in Germany—A national reconnaissance. *Environmental Science & Technology*, *53*(11), 6154–6161. https://doi.org/10.1021/acs.est.8b07216

Keller, V. D. J., Williams, R. J., Lofthouse, C., & Johnson, A. C. (2014). Worldwide estimation of river concentrations of any chemical originating from sewage-treatment plants using dilution factors. *Environmental Toxicology and Chemistry*, *33*(2), 447–452. https://doi.org/10.1002/etc.2441

Krysanova, V., Vetter, T., Eisner, S., Huang, S., Pechlivanidis, I., Strauch, M., et al. (2017). Intercomparison of regional-scale hydrological models and climate change impacts projected for 12 large river basins worldwide—A synthesis. *Environmental Research Letters*, *12*(10), 105002. https://doi.org/10.1088/1748-9326/aa8359

Krysanova, V., Zaherpour, J., Didovets, I., Gosling, S. N., Gerten, D., Hanasaki, N., et al. (2020). How evaluation of global hydrological models can help to improve credibility of river discharge projections under climate change. *Climatic Change*, *163*(3), 1353–1377. https://doi.org/10.1007/s10584-020-02840-0

Kumar, R., Samaniego, L., & Attinger, S. (2013). Implications of distributed hydrologic model parameterization on water fluxes at multiple scales and locations. *Water Resources Research*, *49*(1), 360–379. https://doi.org/10.1029/2012WR012195

Labbe, S. (2021). *Heat wave pushes Metro Vancouver to "unprecedented" early season water use*. Glacier Media Group. Retrieved from https://www.vancouverisawesome.com/local-news/heat-wave-pushes-metro-vancouver-to-unprecedented-early-season-water-use-3921388

Lehner, B., Verdin, K., & Jarvis, A. (2008). New global hydrography derived from spaceborne elevation data. *Eos, Transactions American Geophysical Union*, *89*(10), 93–94. https://doi.org/10.1029/2008EO100001

Li, R., Hua, P., Zhang, J., & Krebs, P. (2020). Effect of anthropogenic activities on the occurrence of polycyclic aromatic hydrocarbons in aquatic suspended particulate matter: Evidence from Rhine and Elbe Rivers. *Water Research*, *179*, 115901. https://doi.org/10.1016/j.watres.2020.115901

Lomax, C. (2021). Water consumption soars during heatwave. Retrieved from https://www.ilkleygazette.co.uk/news/19457870.water-consumption-soars-heatwave

Marx, A., Kumar, R., Thober, S., Rakovec, O., Wanders, N., Zink, M., et al. (2018). Climate change alters low flows in Europe under global warming of 1.5, 2, and 3°C. *Hydrology and Earth System Sciences*, *22*(2), 1017–1032. https://doi.org/10.5194/hess-22-1017-2018

Mažeikienė, A. (2019). Improving small-scale wastewater treatment plant performance by using a filtering tertiary treatment unit. *Journal of Environmental Management*, *232*, 336–341. https://doi.org/10.1016/j.jenvman.2018.11.076

Naik, K. S., & Stenstrom, M. K. (2012). Evidence of the influence of wastewater treatment on improved public health. *Water Science and Technology*, *66*(3), 644–652. https://doi.org/10.2166/wst.2012.144

Nivala, J., van Afferden, M., Hasselbach, R., Langergraber, G., Molle, P., Rustige, H., & Nowak, J. (2018). The new German standard on constructed wetland systems for treatment of domestic and municipal wastewater. *Water Science and Technology*, *78*(11), 2414–2426. https://doi.org/10.2166/wst.2018.530

Pearlman, J. (2018). *Sydney swelters in heatwave, water usage at highest level in 15 years*. Straits Times. Retrieved from https://www.straitstimes.com/asia/australianz/weekend-heatwave-makes-sydney-the-hottest-city-on-earth

Pfeiffer, M., & Ionita, M. (2017). Assessment of hydrologic alterations in Elbe and Rhine Rivers, Germany. *Water*, *9*(9), 684. https://doi.org/10.3390/w9090684

Pusch, M. T., Behrendt, H., Gancarczyk, A., Kronvang, B., Sandin, L., Stendera, S., et al. (2009). Chapter 14 - Rivers of the Central European highlands and plains. In K. Tockner, U. Uehlinger, & C. T. Robinson (Eds.), *Rivers of Europe* (pp. 525–576). Academic Press.

Ragas, A. M. J., Scheren, P. A. G. M., Konterman, H. I., Leuven, R. S. E. W., Vugteveen, P., Lubberding, H. J., et al. (2005). Effluent standards for developing countries: Combining the technology- and water quality-based approach. *Water Science and Technology*, *52*(9), 133–144. https://doi.org/10.2166/wst.2005.0304

Rahman, S. M., Eckelman, M. J., Onnis-Hayden, A., & Gu, A. Z. (2016). Life-cycle assessment of advanced nutrient removal technologies for wastewater treatment. *Environmental Science & Technology*, *50*(6), 3020–3030. https://doi.org/10.1021/acs.est.5b05070

Rakovec, O., Samaniego, L., Hari, V., Markonis, Y., Moravec, V., Thober, S., et al. (2022). The 2018–2020 multi-year drought sets a new benchmark in Europe. *Earth's Future*, *10*(3), e2021EF002394. https://doi.org/10.1029/2021EF002394

Rice, J., & Westerhoff, P. (2017). High levels of endocrine pollutants in US streams during low flow due to insufficient wastewater dilution. *Nature Geoscience*, *10*(8), 587–591. https://doi.org/10.1038/ngeo2984

Richardson, J. S. (2019). Biological diversity in headwater streams. *Water*, *11*(2), 366. https://doi.org/10.3390/w11020366

Ritz, S., & Fischer, H. (2019). A mass balance of nitrogen in a large lowland river (Elbe, Germany). *Water*, *11*(11), 2383. https://doi.org/10.3390/w11112383

Samaniego, L., Kumar, R., & Attinger, S. (2010). Multiscale parameter regionalization of a grid-based hydrologic model at the mesoscale. *Water Resources Research*, *46*(5), W05523. https://doi.org/10.1029/2008wr007327

Samaniego, L., Thober, S., Wanders, N., Pan, M., Rakovec, O., Sheffield, J., et al. (2019). Hydrological forecasts and projections for improved decision-making in the water sector in Europe. *Bulletin of the American Meteorological Society*, *100*(12), 2451–2472. https://doi.org/10.1175/BAMS-D-17-0274.1

Schmidt, C., Kumar, R., Yang, S., & Büttner, O. (2020). Microplastic particle emission from wastewater treatment plant effluents into river networks in Germany: Loads, spatial patterns of concentrations and potential toxicity. *Science of the Total Environment*, *737*, 139544. https://doi.org/10.1016/j.scitotenv.2020.139544

Sheldon, F., Barma, D., Baumgartner, L. J., Bond, N., Mitrovic, S. M., & Vertessy, R. (2021). Assessment of the causes and solutions to the significant 2018–19 fish deaths in the Lower Darling River, New South Wales, Australia. *Marine and Freshwater Research*, *73*(2), 147. https://doi.org/10.1071/MF21038

Siddiqui, S., Conkle, J. L., Scarpa, J., & Sadovski, A. (2020). An analysis of U.S. wastewater treatment plant effluent dilution ratio: Implications for water quality and aquaculture. *Science of the Total Environment*, *721*, 137819. https://doi.org/10.1016/j.scitotenv.2020.137819

Stoddard, A., Harcum, J. B., Simpson, J. T., Pagenkopf, J. R., & Bastian, R. K. (2003). *Municipal wastewater treatment: Evaluating improvements in national water quality*. John Wiley & Sons.

Stott, P. (2016). How climate change affects extreme weather events. *Science*, *352*(6293), 1517–1518. https://doi.org/10.1126/science.aaf7271

Strahler, A. N. (1957). Quantitative analysis of watershed geomorphology. *Eos, Transactions American Geophysical Union*, *38*(6), 913–920. https://doi.org/10.1029/TR038i006p00913

Thober, S., Kumar, R., Wanders, N., Marx, A., Pan, M., Rakovec, O., et al. (2018). Multi-model ensemble projections of European river floods and high flows at 1.5, 2, and 3 degrees global warming. *Environmental Research Letters*, *13*(1), 014003. https://doi.org/10.1088/1748-9326/aa9e35

Uehlinger, U., Arndt, H., Wantzen, K. M., & Leuven, R. S. E. W. (2009). Chapter 6 - The Rhine river basin. In K. Tockner, U. Uehlinger, & C. T. Robinson (Eds.), *Rivers of Europe* (pp. 199–245). Academic Press.

UN. (2015). Transforming our world: The 2030 agenda for sustainable development. Retrieved from https://sustainabledevelopment.un.org/post2015/transformingourworld/publication

UN. (2016). *World cities report 2016: Urbanization and development - emerging futures*. United Nations Human Settlements Programme (UN-Habitat).

UN. (2019). *World population Prospects 2019*. United Nations Department of Economic and Social Affairs, Population Division.

UNEP. (2016). *A Snapshot of the world's water quality: Towards a global assessment*. United Nations Environment Programme.

USEPA. (1972). Federal water pollution control act (clean water act). 33 U.S.C §§ 1251 et seq.

USEPA. (2015). *Incident action checklist – extreme heat*. Office of Water. Retrieved from https://www.epa.gov/system/files/documents/2021-10/incident-action-checklist-extremeheat_508c-final.pdf

Viessman, W., Hammer, M. J., Perez, E. M., & Chadik, P. A. (2009). *Water supply and pollution control* (8th edn.). Pearson.

Wang, X., Daigger, G., de Vries, W., Kroeze, C., Yang, M., Ren, N.-Q., et al. (2019). Impact hotspots of reduced nutrient discharge shift across the globe with population and dietary changes. *Nature Communications*, *10*(1), 2627. https://doi.org/10.1038/s41467-019-10445-0

Warszawski, L., Frieler, K., Huber, V., Piontek, F., Serdeczny, O., & Schewe, J. (2014). The inter-sectoral impact model intercomparison project (ISI–MIP): Project framework. *Proceedings of the National Academy of Sciences of the United States of America*, *111*(9), 3228–3232. https://doi.org/10.1073/pnas.1312330110

Weitere, M., Altenburger, R., Anlanger, C., Baborowski, M., Bärlund, I., Beckers, L.-M., et al. (2021). Disentangling multiple chemical and non-chemical stressors in a lotic ecosystem using a longitudinal approach. *Science of the Total Environment*, *769*, 144324. https://doi.org/10.1016/j.scitotenv.2020.144324

Wells, R. (2014). *Heatwave sparks record-breaking water use in Melbourne households*. Sydney Morning Herald. Retrieved from https://www.smh.com.au/environment/weather/heatwave-sparks-recordbreaking-water-use-in-melbourne-households-20140128-3l1l1p

Westphal, K., Graeber, D., Musolff, A., Fang, Y., Jawitz, J. W., & Borchardt, D. (2019). Multi-decadal trajectories of phosphorus loading, export, and instream retention along a catchment gradient. *Science of the Total Environment*, *667*, 769–779. https://doi.org/10.1016/j.scitotenv.2019.02.428

Wohl, E. (2017). The significance of small streams. *Frontiers of Earth Science*, *11*(3), 447–456. https://doi.org/10.1007/s11707-017-0647-y

Wolfe, M., Kaur, M., Yates, T., Woodin, M., & Lantagne, D. (2018). A systematic review and meta-analysis of the association between water, sanitation, and hygiene exposures and cholera in case-control studies. *The American Journal of Tropical Medicine and Hygiene*, *99*(2), 534–545. https://doi.org/10.4269/ajtmh.17-0897

Wu, H., Zhang, J., Ngo, H. H., Guo, W., Hu, Z., Liang, S., et al. (2015). A review on the sustainability of constructed wetlands for wastewater treatment: Design and operation. *Bioresource Technology*, *175*, 594–601. https://doi.org/10.1016/j.biortech.2014.10.068

WWAP. (2017). *The United Nations world water development report 2017: Wastewater, the untapped resource*. United Nations World Water Assessment Programme, UNESCO.

Yang, S., Bertuzzo, E., Borchardt, D., & Rao, P. S. C. (2021). Hortonian scaling of coupled hydrological and biogeochemical responses across an intensively managed river basin. *Frontiers in Water*, *3*, 97. https://doi.org/10.3389/frwa.2021.693056

Yang, S., Büttner, O., Jawitz, W. J., Kumar, R., Rao, P. S. C., & Borchardt, D. (2019). Spatial organization of human population and wastewater treatment plants in urbanized river basins. *Water Resources Research*, *55*(7), 6138–6152. https://doi.org/10.1029/2018WR024614

Yang, S., Büttner, O., Kumar, R., Jäger, C. G., Jawitz, J. W., Rao, P. S. C., & Borchardt, D. (2019). Spatial patterns of water quality impairments from point source nutrient loads in Germany's largest national River Basin (Weser River). *Science of the Total Environment*, *697*, 134145. https://doi.org/10.1016/j.scitotenv.2019.134145

Zink, M., Kumar, R., Cuntz, M., & Samaniego, L. (2017). A high-resolution dataset of water fluxes and states for Germany accounting for parametric uncertainty. *Hydrology and Earth System Sciences*, *21*(3), 1769–1790. https://doi.org/10.5194/hess-21-1769-2017