Global Change Can Make Coastal Eutrophication Control in China More Difficult

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Abstract Fast socio-economic development in agriculture and urbanization resulted in increasing nutrient export by rivers, causing coastal eutrophication in China. In addition, climate change may affect hydrology, and as a result, nutrient flows from land to sea. This study aims at a better understanding of how future socio-economic and climatic changes may affect coastal eutrophication in China. We modeled river export of total dissolved nitrogen (TDN) and phosphorus (TDP) in 2050 for six scenarios combining socio-economic pathways (SSPs) and Representative Concentration Pathways (RCPs). We used the newly developed MARINA 2.0 (Model to Assess River Inputs of Nutrients to seAs) model. We found that global change can make coastal eutrophication control in China more difficult. In 2050 coastal waters may be considerably more polluted or considerably cleaner than today depending on the SSP-RCP scenarios. By 2050, river export of TDN and TDP is 52% and 56% higher than in 2012, respectively, in SSP3-RCP8.5 (assuming large challenges for sustainable socio-economic development, and severe climate change). In contrast, river export of nutrients could be 56% (TDN) and 85% (TDP) lower in 2050 than in 2012 in SSP1-RCP2.6 (assuming sustainable socio-economic development, and low climate change). Climate change alone may increase river export of nutrients considerably through hydrology: We calculate 24% higher river export of TDN and 16% higher TDP for the SSP2 scenario assuming severe climate change compared to the same scenario with low climate change (SSP2-RCP8.5 vs. SSP2-RCP2.6). Policies and relevant technologies combining improved nutrient management and climate mitigation may help to improve water quality in rivers and coastal waters of China.

Plain Language Summary Fast socio-economic development has resulted in increasing nutrient export by rivers, causing coastal eutrophication in China. In addition, climate change may affect river discharge, and as a result, nutrient flows from land to sea. We explored how future socio-economic and climatic changes may affect coastal eutrophication in China. We found that in 2050 coastal waters may be considerably more polluted or considerably cleaner than today depending on the socio-economic development and climate change. By 2050, river export of total dissolved nitrogen (TDN) and phosphorus (TDP) is 52% and 56% higher than in 2012, respectively, in a scenario assuming large challenges for sustainable socio-economic development and severe climate change. In contrast, river export of nutrients could be 56% (TDN) and 85% (TDP) lower in 2050 than in 2012 in the scenario assuming sustainable socio-economic development and low climate change. Climate change alone may increase river export of nutrients considerably: we calculate 24% higher river export of TDN and 16% higher TDP for a scenario assuming severe climate change compared to the same scenario with low climate change. Policies combining improved nutrient management and climate mitigation may help to improve water quality in rivers and coastal waters of China.
China experienced rapid socio-economic development in the last decades (Bai et al., 2014). Urbanization and food production have caused increasing nutrient inputs to water systems (Ma et al., 2014; Yu et al., 2019). The use of synthetic fertilizers in China increased by almost fivefold from 13 Mton in 1980 to 58 Mton in 2012 (National Bureau of Statistics of China, 2013). However, typically less than half of these nutrients are used by crops (Ma et al., 2012; Wang, Ma, Strokal, Chu, et al., 2018). The remainder is lost to air or waters. In 2012, China produced in total 68,000 Mton of wastewater (Ministry of Environmental Protection, 2012). Only 60%–82% of this was treated before being discharged to rivers (Ministry of Housing and Urban-Rural Development of the People’s Republic of China, 2012). As a result, rivers transport increasing amounts of nutrients to coastal waters, causing coastal eutrophication and harmful algal blooms (Dai et al., 2010; Huan et al., 2016; Zhou et al., 2018). Such blooms occurred approximately 500 times in coastal areas of China between 2006 and 2012 (China’s State Oceanic Administration, 2012). The area affected by coastal eutrophication in China has been increasing to be 98,000 km² in 2012 (SOA, 2012).

There may be more coastal eutrophication in the future as a result of socio-economic development (Crespo, 2017). In addition, global climate change may affect hydrology and, as a result, river discharge. This may, in turn, affect nutrient levels in rivers (Han et al., 2009; Howarth et al., 2006; Rabalais et al., 2009; Sinha et al., 2017). For example, Sinha et al. (2017) conclude that climate change-induced changes in precipitation alone may substantially increase river export of nitrogen in Eastern China by the end of the century. Considerable increases in temperature and precipitation in the future are projected in China as a result of climate change (Lee et al., 2014; Sun et al., 2015; Wang & Chen, 2014; Zhou et al., 2014). For example, in Sun et al. (2015), precipitation in China is projected to increase during the 21st century by 13.4% ± 12.6% for RCP8.5. To what extent climate change may increase water pollution by nutrients, however, is unclear. A better understanding of the combined impacts of socio-economic and climate change on future river export of nutrients will contribute to the formulation of effective policies for improving water quality in China and other rapidly developing world regions.

Modeling studies exist to quantify past trends in river export of nutrients to seas in China (Liu et al., 2018; Liu et al., 2019; Qu & Kroese, 2010; Strokal et al., 2016; UNEP, 2016). A few of these studies (Qu & Kroese, 2010; Strokal et al., 2016; Strokal et al., 2017) explored future trends and focused on the impacts of socio-economic change based on the old Millennium Ecosystem Assessment scenarios (Alcamo et al., 2005). So far, the shared socio-economic pathways (SSPs) and Representative Concentration Pathways (RCPs) have not yet been used to assess future nutrient export by rivers in China. SSPs are global storylines describing future socio-economic development such as changes in population, urbanization, and gross domestic product (O’Neill et al., 2014). RCPs are four pathways covering a range of radiative forcing values of the year 2100 from 2.6 to 8.5 W/m² (van Vuuren et al., 2011). Such an assessment is important, since socio-economic development and climate change together may strongly impact river export of nutrients to coastal waters (Sinha et al., 2019).

This study, therefore, aims at a better understanding of how future socio-economic and climatic changes may affect coastal eutrophication through future river export of total dissolved nitrogen (TDN) and phosphorus (TDP) by Chinese rivers at the subbasin scale and by source. To this end, we applied the newly developed MARINA 2.0 (Model to Assess River Inputs of Nutrients to seAs) model for 12 Chinese rivers and their subbasins. For 2050, six scenarios combining the SSPs and RCPs were explored. We analyzed hotspots of N and P losses to rivers in 2012 and 2050 to better understand the impacts of socio-economic change on river export of nutrients. We also analyzed the potential impacts of climate change on river export of nutrients. Finally, we discussed opportunities for improving water quality in Chinese rivers and coastal waters. This study will contribute to a better understanding of future coastal eutrophication as affected by global change in China. The MARINA 2.0 model and the scenario approach have potential applications to other regions of the world.

2. Method

2.1. The Study Area

We quantified river export of TDN and TDP for 12 Chinese rivers draining into four seas: Bohai Gulf, Yellow Sea, East China Sea, and South China Sea (Figure 1). The drainage area of these rivers covers 40% of China’s land (around 4 million km⁻²). The drainage basins of Yellow (1.92 million km⁻²), Yangtze (0.76 million km⁻²), and Pearl (0.43 million km⁻²) were divided into upstream, middle steam, and downstream.
subbasins largely following Strokal et al. (2016) (see Figures 2 and SA3–SA5 in the supporting information (SI) for names and locations of subbasins).

2.2. MARINA 2.0

We developed the MARINA 2.0 model to quantify river export of total dissolved N (TDN) and P (TDP) by 12 Chinese rivers, by source, and at the subbasins scale for 2012 and 2050. TDN is the sum of dissolved inorganic (DIN) and dissolved organic (DON) N. TDP is the sum of dissolved inorganic (DIP) and dissolved organic (DOP) P. MARINA 2.0 quantifies dissolved N and P export by rivers as a function of N and P inputs to surface waters (rivers) from diffuse and point sources and retentions of N and P in rivers based on the overall equation below:

\[ M_{F,Y,j} = \left( R_{Sdif,F,Y,j} + R_{Spnt,F,Y,j} \right) \cdot F_{E_{riv,F,outlet,j}} \cdot F_{E_{riv,F,mouth,j}} \]  

(1)

where \( M_{F,Y,j} \) is river export of N and P in form F (DIN, DON, DIP, and DOP) by source y from subbasin j (kg year\(^{-1}\)). \( R_{Sdif,F,Y,j} \) is N and P inputs in form F to rivers (surface waters) from diffuse sources y in subbasin j (kg year\(^{-1}\)). \( R_{Spnt,F,Y,j} \) is N and P inputs in form F to rivers from point sources y in subbasin j (kg year\(^{-1}\)).
FEriv,F.outlet,j is the fraction of N and P in form F exported to the outlet of subbasin j (0–1). FEriv,F.mouth,j is the fraction of N and P in form F exported from the outlet of subbasin j to the river mouth (0–1). We summarized the equations to quantify RSdifF,y.j, RSpntF,y,j, FEriv,F.outlet.j, and FEriv,F.mouth.j in Boxes SA1 and SA2 in SI.

The MARINA 2.0 model is based on the MARINA 1.0 model, developed by Strokal et al. (2016) for six large rivers: Yellow, Yangtze, Pearl, Liao, Huai, and Hai rivers in China. MARINA 1.0 quantifies river export of four nutrient forms (DIN, DON, DIP, and DOP) to the river mouth by subbasins from different sources. In this study, the MARINA 1.0 model was modified and updated to MARINA 2.0 for 12 rivers in China. We focused on rivers that contain at least five grids of 0.5°×0.5° spatial resolution.

MARINA 2.0 differs from MARINA 1.0 in the following aspects. First, we included in MARINA 2.0 human waste from rural population that is connected to sewage systems. This source was not considered in MARINA 1.0 because the rural population in China had very limited connections to sewage systems in 2000 (MOHURD, 2001). Second, we created the basin delineation for the Chinese rivers using the 30‐arcmin (0.5°×0.5°) spatial resolution drainage direction map 30 (Döll & Lehner, 2002). The MARINA 1.0 model (Strokal et al., 2016) used the 30‐arcmin Simulated Topological Networks (STN‐30) (Vörösmarty et al., 2000). Third, we updated the approach to quantify human waste based on the MARINA‐Global model (Strokal et al., 2019). This was done by adjusting the method to calculate protein N intake using units of 2005 US$ instead of 1995 US$ for GDP (gross domestic product) (see equations in Box SA1 in SI).

To apply the MARINA 2.0 model to Chinese rivers for 2012, we updated the model inputs for (1) point sources (e.g., GDP, population, sewage connection, and sewage treatment) mainly based on Chinese statistics, (2) diffuse sources (e.g., use of synthetic fertilizers and animal manure) based on the NUFER (NUtrient flows in Food chains, Environment, and Resources use) model at the county scale, (3) hydrology (e.g., river discharge) with simulations based on the VIC (variable infiltration capacity) model from the study of (van Vliet, van Beek, et al., 2016; van Vliet, Wiberg, et al., 2016). The overview of the updated model inputs, their sources, and the methods that we used to aggregate these model inputs to the subbasin scale are in Figure SA1 and Tables SA2 and SA3 in SI.
We evaluated the performance of MARINA 2.0 by comparing the modeled results with measurements at the river mouths (Appendix D in SI), modeled results from other studies, and by performing sensitivity analysis. The results show that MARINA 2.0 has an acceptable performance in quantifying river export of nutrients for China. Detailed discussions on our model evaluation are available in section 4.1.

**Indicator for coastal eutrophication.** Based on the modeled results by MARINA 2.0, we assessed the potential impacts of river export of TDN and TDP on coastal eutrophication in 2012 and 2050 for the SSP-RCP scenarios. To do this, the Indicator for coastal eutrophication (ICEP) developed by Billen and Garnier (2007) was used. The ICEP indicator expresses the potential for new production of non-siliceous (possibly harmful) algae in coastal waters when receiving excess N or P over silica from rivers. ICEP is calculated by comparing the N, P, and Si loads to the Redfield molar ratios (C:N:P:Si ratios 106:16:1:20) (see the equations below to quantify ICEP). Either N- or P-ICEPs were calculated depending on if N or P is limiting (i.e., N is limiting if N:P < 16; otherwise P is limiting). Positive ICEP value indicates an excess of N or P over Si, which leads to high potentials for blooms of harmful algae. Negative ICEP value indicates a low potential for blooms of harmful algae. Following the approach of Garnier et al. (2010), we calculated ICEPs for the 12 Chinese rivers following the equations:

\[
N\text{–ICEP} = \frac{|TDNflx/(14 \cdot 16) – DSiflx/(28 \cdot 20)|}{106 \cdot 12} \text{ if } N : P < 16 \text{ (N is limiting) (2)}
\]

\[
P\text{–ICEP} = \frac{|TDPflx/(31 – DSiflx/(28 \cdot 20)|}{106 \cdot 12} \text{ if } N : P > 16 \text{ (P is limiting) (3)}
\]

where TDNflx and TDPflx are the fluxes of total dissolved nitrogen and phosphorus, respectively (kg km\(^{-2}\) year\(^{-1}\)). DSiflx is the flux of total dissolved silica (kg km\(^{-2}\) year\(^{-1}\)). TDNflx and TDPflx were based on our quantified river export of TDN and TDP by the MARINA 2.0 model in this study. DSiflx was from the Global NEWS-2 (Global Nutrient Export from WaterSheds) model (Seitzinger et al., 2010) (see Table SA5 for values of DSi).

**2.3. Scenarios**

We quantified river export of TDN and TDP by rivers for 2050. Five SSPs were selected for strong, rapid (SSP1—Sustainability and SSP5—Fossil fueled), moderate (SSP2—Middle of the Road), and slow (SSP3—Regional Rivalry and SSP4—Inequality) socio-economic developments (O’Neill et al., 2014). SSP1 assumes a gradual shift toward sustainability with low population growth, economic growth, efficient use of resources, high technical solutions, and improved environmental policies for environmental pollution. SSP2 assumes moderate progress toward sustainability and moderate population growth, and some improvements in resource use efficiencies and environment policies only for local pollution. SSP3 assumes a fragmented world to achieve energy and food security goals with high challenges including strong environmental degradation, high population growth, limited improvements in techniques, and environmental policies for pollution. SSP4 assumes inequalities in population growth, urbanization, economy, and environmental policies among countries and regions in the countries. SSP5 assumes integrated global societies with low population growth, high consumption of fossil fuels, and highly managed environmental conditions at local scales (O’Neill et al., 2017). Two RCPs were selected for the lowest (RCP2.6) and highest (RCP8.5) greenhouse gas concentrations for climate change (van Vuuren et al., 2011).

The six scenarios combining SSPs and RCPs are as follows: SSP1-RCP2.6, SSP2-RCP2.6, SSP2-RCP8.5, SSP3-RCP8.5, SSP4-RCP2.6, and SSP5-RCP8.5 (Figure 3). These combinations were selected based on the SSP-RCP matrix from Kok (2016) and added SSP2-RCP8.5 to analyze the impact of climate change on river export of nutrients.

Model inputs for MARINA 2.0 for hydrology (e.g., river discharge) for RCP 2.6 and RCP8.5 were from van Vliet, van Beek, et al. (2016). van Vliet, van Beek, et al. (2016) implemented bias-corrected model output of five global climate models from Intersectoral Impact Model Intercomparison Project (Hemel et al., 2013) for both RCP2.6 and RCP8.5 in the VIC hydrological model (Figure SA1 and Table SA3). Most model inputs for agricultural activities were derived by implementing the SSP storylines for China’s food system from Wang et al. (2017) to the NUFER model at the county scale (Wang, Ma, Strokal, Ma, et al., 2018) (Figure SA1 and Table SA3). Model inputs for calculating nutrient export from human waste for the selected scenarios were estimated based on our assumptions on (1) the fraction of population connected to sewage
systems and (2) N and P removal efficiencies during sewage treatment based on the SSP-RCP storylines and existing studies (O’Neill et al., 2017; van Drecht et al., 2009) (Table SA4). Important model inputs are presented in Figures SB1–SB8 in SI.

### 2.4. Hotspots for N and P Losses

In order to better understand how socio-economic change influences future river export of nutrients, we analyzed hotspots for N and P losses to rivers from diffuse (agriculture) and point (agriculture and human waste) sources for the SSPs. "Hotspots" are the regions where N and P losses to rivers are higher than in other regions, as a result of nutrient management in socio-economic activities. We followed the approach of Wang, Ma, Strokal, Ma, et al. (2018) to identify hotspots for N and P losses from agriculture. Our hotspots are presented at the gridded (0.5° × 0.5°) scale. Below we describe our

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**Figure 3.** (a) The six scenarios combining shared socio-economic pathways (SSPs) and Representative Concentration Pathways (RCPs), presented in a SSP-RCP matrix; (b) the qualitative storylines for agriculture, sewage systems, and climate mitigation in China in the SSP-RCP scenarios. SSP2-RCP8.5 is not the most likely combination but was added to analyze the impact of climate change on river export of nutrients. The storylines for agriculture are based on Wang et al. (2017). The quantitative scenarios for nutrient flows in agriculture and sewage systems and, for hydrology, (e.g., river discharge) are available in Figures SB1–SB8 in SI.

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Note: +++ strong increase, ++ moderate increase, + slight increase, --- strong decrease, -- moderate decrease, - slight decrease, * no changes
approach to identify the hotspots for diffuse and point sources based on the N and P losses to rivers at the gridded (0.5° × 0.5°) scale.

Diffuse sources include leaching, runoff, and erosion of nutrients (e.g., synthetic fertilizers and animal manure) in agriculture. To identify the hotspots, we first took the NUFER results for N and P losses to waters (rivers and groundwater) from leaching, runoff, and erosion in agriculture at the county scale. These results were aggregated from counties to grids (0.5° × 0.5°) as described in Table SA3 and Figure SA2 in SI. Next, the N and P losses were averaged by the area of the grids and were grouped into four groups (Figures 9 and SC4). The ranges of N and P losses for the four groups were the same as defined by Wang, Ma, Strokal, Ma, et al. (2018), who identified hotspots for N and P losses from food production to the environment. The top groups (ranking by N and P losses) were considered as hotspots. Based on Wang, Ma, Strokal, Ma, et al. (2018), for N, grids with losses to waters from diffuse sources exceeding 2452 kg N km⁻² year⁻¹ were qualified as hotspots. For P, grids with losses from diffuse sources exceeding 147 kg P km⁻² year⁻¹ were considered as hotspots.

Point sources include direct discharge of manure and discharge of collected (through sewage systems) and uncollected (not through sewage systems) human waste. The same approach to identify hotspots as for diffuse sources was applied. The N and P losses to rivers from discharge of collected and uncollected human waste were not available from NUFER. Thus, we quantified these on the gridded scale using the MARINA approach (see Box SA1) based on the gridded population and their connection to sewage systems (Figure SA1). Wang, Ma, Strokal, Ma, et al. (2018) did not identify the hotspots for N and P losses to rivers from discharge of collected and uncollected human waste. We, therefore, identified the hotspots for these sources based on the number of hotspots (grids) for the diffuse source. For diffuse sources, we identified around 575 grids as hotspots (i.e., 14% grid cells with highest N and P losses). The range of N and P losses for these top 575 grids (ranking by N or P losses) was thus used as the range for hotspots for discharge of collected and uncollected human waste (Figures 9 and SC4).

3. Results

3.1. River Export of Nutrients in 2012

In 2012, the 12 Chinese rivers exported in total 3,287 kton of DIN, 1,567 kton of DON, 295 kton of DIP, and 411 kton of DOP to seas (Figure 4). About 68%–80% of these nutrients originated from human activities (e.g., use of synthetic fertilizers, direct discharge of manure, and uncollected human waste) in the Hai and Liao rivers (Figures 5 and 6 and SC1 and SC2 in SI). Direct discharge of animal manure to rivers was responsible for 41% of DIN and for 68%–83% of DON, DIP, and DOP exports to the Bohai Gulf. The Yellow River has a large drainage area of 758,715 km² but only contributed by less than 17% to the TDN and TDP export to the Bohai Gulf in 2012. This is largely caused by the relatively high retentions of nutrients in the Yellow River basin (Figure SC6 in SI).

The Huai River is the only river flowing into the Yellow Sea in our model. In 2012, the Huai River exported 174 kton of DIN, 77 kton of DON, 37 kton of DIP, and 26 kton of DOP to the Yellow Sea (Figure 4). Direct discharge of animal manure was the most important source for DON, DIP, and DOP and accounted for 63%–86% of these forms entering the Yellow Sea (Figures 5 and 6 and SC1 and SC2 in SI). Use of synthetic fertilizers was another important source and accounted for 42% of the DIN inputs to the Yellow Sea.

The East China Sea received 1,699 kton of DIN, 904 kton of DON, 140 kton of DIP, and 233 kton of DOP in 2012 (Figure 4). More than 90% of these nutrients were exported by the Yangtze River. Close to 85% of DIN and DIP and 66%–69% of DON and DOP exported by the Yangtze River to coastal waters were from activities in middle stream and downstream subbasins (Figures 5 and 6 and SC1, SC2, and SC5 in SI). The main sources of nutrient export to the East China Sea were direct discharge of animal manure, use of synthetic
fertilizers, and discharge of uncollected human waste (Figures 5 and 6 and SC1 and SC2 in SI). In 2012, these sources accounted for 53%–94% of the nutrient inputs to seas depending on the nutrient forms. Direct discharge of animal manure was the most important source for DON, DIP, and DOP. Other sources such as use of synthetic fertilizers are important for DIN inputs to the East China Sea.

The South China Sea received 1,044 kton of DIN, 275 kton of DON, 55 kton of DIP, and 59 kton of DOP in 2012 (Figure 4). About 72%–84% of these nutrients were exported by the Pearl River. Most of the nutrients (79%–86%) exported by the Pearl River to coastal waters originated from the activities in middle-stream and downstream subbasins (Figures 5 and 6 and SC1, SC2, and SC5 in SI). Direct discharge of animal manure and direct discharge of uncollected human waste were responsible for 57% of DON, 64% of DIP, and 88% of DOP entering the South China Sea in 2012 (Figures 5 and 6 and SC1 and SC2 in SI). Atmospheric N deposition, biological N2 fixation, and use of synthetic fertilizers were the main sources of DIN export to the South China Sea.

Based on the river export of TDN and TDP, we assessed the potential impacts on coastal eutrophication for the 12 rivers. In 2012, the ICEP values range from −39 to +51 C·eq. km⁻²·year⁻¹ among rivers (Figure 7). Almost all 12 rivers have positive ICEP values (>0), indicating high potentials for harmful algae to develop in coastal waters. The Oujiang and Fuchun rivers are exceptions with negative ICEP values (<0), indicating low potentials to cause coastal eutrophication.

### 3.2. Future River Export of Nutrients

In 2050, rivers and coastal waters may be much more polluted or much cleaner than today, depending on the socio-economic and climatic changes in the SSP-RCP scenarios. For example, we project river export of TDN and TDP 52% and 56% higher in 2050 than in 2012, respectively, in SSP3-RCP8.5. In contrast, river export of...
Figure 5. River export of dissolved inorganic nitrogen (DIN, kton year$^{-1}$) from subbasins of 12 Chinese rivers by source in 2012 and 2050 for six SSP-RCP scenarios. SSPs are the shared socio-economic pathways. RCPs are the Representative Concentration Pathways. Details on the SSP-RCP scenarios are in section 2.3. The names and locations of the rivers are in Figures 1 and SA3–SA5.
Figure 6. River export of dissolved inorganic phosphorus (DIP, kton year$^{-1}$) from subbasins of 12 Chinese rivers by source in 2012 and 2050 for six SSP-RCP scenarios. SSPs are the shared socio-economic pathways. RCPs are the Representative Concentration Pathways. Details on the SSP-RCP scenarios are in section 2.3. The names and locations of the rivers are in Figures 1 and SA3–SA5.
nutrients is 56% (TDN) and 85% (TDP) lower in 2050 than in 2012 in SSP1-RCP2.6. Below, we describe the changes in river export of nutrients for each of the six scenarios in detail.

In SSP1-RCP2.6, total river export of dissolved N and P by Chinese rivers in 2050 is projected to be 50%–90% lower than in 2012 depending on the nutrient forms (Figure 4). The decreases also vary among subbasins. In most subbasins, the river export of N or P is reduced by up to half of the amount in 2012 (Figures 8 and SC3). The large decreases in nutrients export are explained by the assumed restricted discharge of animal manure, reduced use of synthetic fertilizers, and improved sewage connections and wastewater treatment in this scenario. As a result, atmosphere N deposition and biological N2 fixation become the dominant sources of DIN in river export and are responsible for up to 87% of DIN exported from subbasins. Discharge of collected (through sewage systems) and uncollected human waste is the most important sources of DON, DIP, and DOP in this scenario (Figures 5 and 6 and SC1 and SC2 in SI).

SSP2-RCP2.6 and SSP2-RCP8.5 assume the same moderate socio-economic developments but with different levels (low for RCP2.6, high for RCP8.5) of greenhouse gas concentrations and related changes in climate in 2050. River export to all four seas are projected to decrease between 2012 and 2050 for both scenarios. The four N and P forms entering Chinese Seas are 30%–58% lower in SSP2-RCP2.6 than in 2012 and are 11%–54% lower in SSP2-RCP8.5 (Figure 4). Direct discharge of animal manure is projected to reduce but may still remain the main source of DON, DIP, and DOP. For DIN, other sources such as atmospheric N deposition and biological N2 fixation will become the dominant sources for most subbasins (Figures 5 and 6 and SC1 and SC2 in SI).

River export of dissolved N and P is projected to increase considerably between 2012 and 2050 in SSP3-RCP8.5. We calculate increases in river export of DIN by 41%, of DIP by 67%, and of DOP by 54% (Figure 4). The amount of DON may almost double between 2012 and 2050. The large increase in nutrients transported to coastal waters is related to the intensive human activities in agriculture and urbanization and the poor nutrient management (e.g., overuse of synthetic fertilizers and insufficient wastewater treatment) in SSP3-RCP8.5. Direct discharge of manure is expected to remain the dominant source of DON, DIP, and DOP. Use of synthetic fertilizers is expected to be an important source and contributes by up to 50% to river export of DIN among the subbasins (Figures 5 and 6 and SC1 and SC2 in SI).

In SSP4-RCP2.6 and SSP5-RCP8.5 scenarios, river export of dissolved N and P in 2050 is slightly lower or slightly higher than in 2012 depending on the nutrient forms (Figure 4). Changes in nutrients export vary largely among subbasins and nutrient forms as a net effect of future changes in human activities, nutrient management, and river discharges. For example, most subbasins of the Pearl River are expected to deliver increased amounts of DON but decreased amounts of DIN to seas in SSP4-RCP2.6 in 2050. This is due to the reduced use of fertilizers for crops and the reduced direct discharge of manure in this scenario. In these two scenarios, direct discharge of manure is expected to remain the dominant source of DON, DIP, and DOP. Use of synthetic fertilizers remains an important source for river export of DIN (Figures 5 and 6 and SC1 and SC2 in SI).

As a result of the abovementioned trends, the potential of the river export of nutrients to cause coastal eutrophication may be higher or lower in the future depending on the SSP-RCP scenarios (Figure SB8 in SI). In 2012, the ICEP values range from −39 to +51 C-eq. km⁻² year⁻¹ among rivers. For the scenario focusing on sustainability (SSP1-RCP2.6), the ICEP values are with a range of −60 to +8 C-eq. km⁻² year⁻¹ among rivers, which are much lower than that in 2012. Several rivers (e.g., rivers draining into the South China Sea and East China Sea and the Luan River) have negative ICEP values, indicating low potentials for harmful algae to develop. In contrast, the potentials for coastal eutrophication are much higher for all rivers in SSP3-RCP8.5 than in 2012. The ICEP values of rivers range from −30 to +94 C-eq. km⁻² year⁻¹. Positive ICEP values are calculated for all rivers except for the Oujiang River. The ICEP values are lower in 2050 than in 2012 in both SSP2-RCP2.6 and SSP2-RCP8.5. They range from −49 to +32 C-eq. km⁻² year⁻¹ in SSP2-RCP2.6 and from −47 to +37 C-eq. km⁻² year⁻¹ in SSP2-RCP8.5 among rivers. However, the potentials of most rivers except for Oujiang and Fuchun to cause coastal eutrophication remain high in these two scenarios indicated by the positive ICEP values. ICEP values are slightly lower in SSP5-RCP8.5 than in 2012, ranging from −45 to +45 C-eq. km⁻² year⁻¹. In SSP4-RCP2.6, the potentials for coastal eutrophication are higher or lower than in 2012 depending on the rivers. For example, ICEP is calculated to increase at the river mouth of Hai. We quantified increased river export of TDN and decreased river export of TDP by Hai in SSP4-RCP2.6.
(Figure 8). According to the N:P ratio, N is limiting for this river (Figure SC1). Thus, a higher ICEP is calculated for Hai taking into account changes in DSi export (see equations (2) and (3) in section 2.2). Positive ICEP values are calculated for all rivers except for the Oujiang and Fuchun rivers in these SSP4-RCP2.6 and SSP5-RCP8.5.

3.3. Role of Socio-Economic Change: Hotspots for Pollution

We identified hotspots where N and P losses to rivers are higher than in other regions at the gridded (0.5° × 0.5°) scale for China. We compared changes in hotspots for N and P losses to rivers between 2012 and 2050 for the SSP scenarios (Figures 9 and SC4). We found that socio-economic change has a large impact on hotspots of N and P losses to rivers and, thus, has a large impact on river export of TDN and TDP. In SSP3 (with high challenges toward sustainability), the number of hotspots for both diffuse (agriculture) and point (agriculture and human waste) sources increase considerably. In 2012, up to 0.9 million km² of area (367 grids, 24% of the total river drainage area) in the river basins were identified as hotspots for N or P losses to rivers from the selected sources. The area of hotspots increases by 10%–76% (36–280 grids) for N losses and by 30%–96% (109–354 grids) for P losses from different sources in SSP3 compared to 2012 (Figures 9 and SC4). This results in increasing river export of TDN and TDP in SSP3 (Figure 4; section 3.2).

In the SSP1 and SSP2 scenarios (focusing on more sustainability), the number of hotspots decreases for N and P losses from diffuse (agriculture) and point (agriculture and uncollected human waste) sources between 2012 and 2050. In these two scenarios, 13%–68% (46–249 grids) of the hotspots for N and 20%–68% (75–245 grids) of the hotspots for P in 2012 are identified as nonhotspot area in 2050 (Figures 9 and SC4). This explains the relatively low river export of nutrients to seas in SSP1 and SSP2 (Figure 4; section 3.2). However, the number of hotspots for N and P losses to rivers from sewage systems (collected human waste) increases in SSP1 and SSP2 as a result of a higher fraction of the population with sewage connections (Figures 9 and SC4).

The number of hotspots for N and P losses from agriculture and human waste in SSP4 and SSP5 either increase or decrease depending on the SSP storylines (Figures 9 and SC4). This explains the trends in river export of nutrients (Figure 4; section 3.2). For example, the increases in hotspots for N and P losses to rivers from direct discharge of manure explain the relatively high river export of DON and DOP in SSP4 in 2050.

3.4. Role of Climate Change

Climate change will impact surface and subsurface runoff and river discharge patterns at different orders of magnitude (depending on the RCP, global climate model and hydrological model used). These hydrological
Figure 8. Changes (%) in river export of dissolved N (DIN) and P (DIP) by subbasins of the Chinese rivers between 2012 and 2050 for six SSP-RCP scenarios. SSPs are the shared socio-economic pathways. RCPs are the Representative Concentration Pathways. Details on the SSP-RCP scenarios are in section 2.3. The names and locations of the rivers are in Figures 1 and SA3-SA5.
changes can impact retention of nutrients on land and in rivers, as well as nutrient runoff, and thus affect river export of nutrients to seas. For example, in China, natural river discharge by most of the subbasins may increase (up to 21%) in RCP8.5 (assuming severe climate change) and decrease (up to 68%) in RCP2.6 (assuming lower climate change) between 2012 and 2050 (Figure SB8). Impacts of these hydrological changes on river export of nutrients are discussed below.

Figure 9. (a) Hotspots for N losses to waters (including rivers and groundwater) from leaching, runoff, and erosion and from direct discharge of animal manure and hotspots for N losses to rivers from sewage systems and from unconnected population (discharge of uncollected human waste that is not connected to sewage systems) in 2012; (b) changes in the number of hotspots in the rivers that drain into the Bohai Gulf, East China Sea, Yellow Sea, and South China Sea between 2012 and 2050 for the five SSPs. The hotspots were identified on the gridded scale following the approach described in section 2.4. SSPs are the shared socio-economic pathways. Details on the SSPs scenarios are in section 2.3.
Results show that climate change-induced changes in river discharge may increase nutrient inputs to Chinese seas in the future. We found that river export of TDN and TDP in SSP3-RCP8.5 is 3–16 times that in SSP1-RCP2.6 (Figure 4). This difference, however, cannot be completely attributed to climate change, since SSP1 and SSP3 differ in the assumed socio-economic change. To better understand the impacts of climate change alone on future river export of nutrients, we compared scenarios SSP2-RCP2.6 (low climate change) and SSP2-RCP8.5 (severe climate change). Our analysis indicates that climate change may increase nutrient inputs to seas considerably: Nutrient export by the 12 Chinese rivers is higher in SSP-RCP8.5 than in SSP-RCP2.6. The difference is 24% for TDN and 16% for TDP (Figure 4). This is explained by the lower reten-
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tions of nutrients on land and in rivers resulted from the higher runoff and river discharge in RCP8.5 com-
pared to RCP2.6 (Figures SB7, SB8, and SC6).

The impacts of climate change on river export of nutrients differ largely among subbasins and nutrient forms. For most subbasins except for Toudaoguai (subbasin of Yellow River), Liao, and Luan rivers, river export of DIN in 2050 in SSP2-RCP8.5 is 5%–525% higher than that in SSP2-RCP2.6. This range is 3%–192% for DON, 5%–275% for DIP, and 3%–174% for DOP. River export of nutrient forms in SSP2-RCP8.5 is slightly lower (up to 9%) than in SSP2-RCP2.6 for Toudaoguai, Liao and Luan subbasin (rivers). This is explained by the lower river discharge in SSP3-RCP8.5 than in SSP3-RCP2.6 in these subbasins (Figures SB7 and SB8).

4. Discussion

4.1. Comparison With Measurements and Other Studies

We compared the MARINA 2.0 modeled loads of DIN, DIP, TDN, and TDP with measurements (Figure SD1 in SI). Based on the comparison, we assessed the model performance using three indicators: Pearson's coefficient of determination \(R^2\), the Nash-Sutcliffe efficiency \(NSE\), and the root mean square error to the standard deviation of measured data \(RSR\) (see section D in SI for details). According to the performance ratings from Moriasi et al. (2007), our model shows a good performance \(R^2 > 0.5, NSE > 0.65, RSR < 0.6\) with the indicators: \(R^2 = 0.85\), \(NSE = 0.72\), and \(RSR = 0.53\).

We also compared our modeled results with other modeling studies for the Yangtze River (Li et al., 2011; Liu et al., 2018; Tong et al., 2017; Yan et al., 2010). Yan et al. (2010) and Li et al. (2011) modeled DIN and DIP exports to coastal waters from the Yangtze River in 2003. Liu et al. (2018) estimated river export of TN and TP by Yangtze in 2010, while Tong et al. (2017) quantified the source attribution of riverine TN and TP export by Yangtze from 2006 to 2012. The comparison shows that our modeled DIN (1,544 kton) export by the Yangtze River is comparable with the study of Yan et al. (2010) (1,611 kton of DIN). We modeled higher DIP (128 kton) than Li et al. (2011) (22–25 kton of DIP) knowing that Li et al. (2011) did not include direct discharge of manure to rivers in their calculations. Our modeled TDN (2,374 kton) and TDP (344 kton) is lower than the estimates of TN (5,708) and TP (371) by Liu et al. (2018). One reason for this difference is that we did not include the particulate forms in this study. Another reason is that Liu et al. (2018) considers other sources that are not included in MARINA 2.0 such as aquaculture, vegetation in floodplains and industrial wastewater. Comparing the TDN and TDP loads to available measurements (Table SD1), our modeled result is closer to the measurements. Results of this study and Tong et al. (2017) both show that the contribution to river export of nutrients increases from upstream to downstream subbasins of Yangtze River. Tong et al. (2017) concludes that nonpoint sources are important for N (36% of TN) and P (63% of TP) export by the river. This is different from our results showing that point sources are dominant for DON, DIP and DOP (see section 3.1) because Tong et al. (2017) did not consider direct discharge of manure to rivers as a point source. Based on the above, we consider the MARINA 2.0 model acceptable for quantifying river export of nutrients (DIN, DON, DIP, and DOP) by sources, at the subbasin scale for China.

Although our modeled results compare reasonably well to the measurements and other modeling studies, we need to realize our model has also uncertainties. Uncertainties in our model are related to the model structure, model inputs and parameters, and scenarios for the future. Uncertainties in the model structure are related to simplifications of nutrient flows from land to seas. The MARINA approach has been validated for large rivers (the Yellow, Yangtze, Pearl, Huai, Hai, and Liao) (Strokal et al., 2016) and several lakes (Lake Taihu, Dianchi, Baiyangdian, and Guanting) in China (Li et al., 2019; Wang, Strokal, Burek, Kroese, et al.,...
2019; Yang et al., 2019). These studies indicate an acceptable performance for MARINA 1.0 for quantifying river export of nutrients.

Uncertainties also exist in model inputs and parameters. Most of the inputs were taken from country statistics (e.g., GDP and sewage treatment plants), peer-reviewed papers, as well as published models (e.g., NUFER and VIC), which we believe, are the most reliable data sets for China (see Figure SA1 and Table A2). The MARINA approach (Strokal et al., 2016) was developed based on the Global NEWS-2 model (Mayorga et al., 2010). The Global NEWS-2 model has been calibrated and validated for rivers worldwide (Amin et al., 2017; Mayorga et al., 2010; Pedde et al., 2017; Strokal & Kroese, 2013) and for large Chinese rivers (the Yellow, Yangtze and Pearl) (Strokal et al., 2016). We tested the sensitivity of model outputs to changes in selected model inputs and parameters (Figures SE1–SE4). Our modeled river exports of DIN, DON, DIP, and DOP for 2012 are relatively sensitive to changes in river discharge, direct discharge of animal manure, and manure excretion. Most of these model inputs were derived from modeling studies published in peer-reviewed journals and published statistics, which are the most reliable data sets for China to our knowledge. Model inputs for river discharge were derived from the simulations by the VIC hydrological model (van Vliet, van Beek, et al., 2016) and Chinese Water Resources Bulletins (Ministry of Water Resources of the People’s Republic of China, 2013) (see Table SA3 for detailed approach). Our river discharge at the river mouth of the Yangtze (859 km$^3$), Yellow (78 km$^3$), and Pearl (302 km$^3$) rivers are within the range of measurements (Table SD1). The model inputs for direct discharge of animal manure and manure excretion were calculated by the NUFER model based on Chinese statistics, peer-reviewed papers and surveys to farmers (Ma et al., 2012; Wang, Ma, Strokal, Ma, et al., 2018).

There are also uncertainties related to the assumptions for future socio-economic developments and climate change in the SSPs-RCPs scenarios. For example, in the SSP1-RCP2.6 scenario, we assumed some optimistic environmental management strategies (e.g., no direct discharge of manure), which may be very challenging to reach in the future. Despite these uncertainties, our scenarios provide a better understanding of future coastal water pollution for a wide range of possible changes in society and climate. This will help to identify potential improved water pollution management options for China. The optimistic environmental management strategies can be taken by the policymakers as a basis to identify feasible or optimal solutions.

### 4.2. Implications for Future Policies

This is the first assessment of river export of nutrients by Chinese rivers combining the impacts of SSPs (socio-economic change) and RCPs (climate change). Results show that future rivers and coastal waters in China may be considerably more polluted or considerably cleaner depending on the socio-economic and climatic changes in the SSPs-RCPs scenarios. For example, by 2050, river export of TDN and TDP is 52% and 56% higher than in 2012, respectively, in SSP3-RCP8.5. In contrast, river export of nutrients could be 56% (TDN) and 85% (TDP) lower in 2050 than in 2012 in SSP1-RCP2.6. In particular, manure management will have an important impact on future river export of nutrients. In addition to this, global climate change was found to be an important driver of coastal eutrophication in Chinese seas. We calculate considerable increases in nutrient export by rivers as a result of climate change. This indicates that effective policies aiming at reducing river and coastal water pollution in China do not only focus on improved nutrient management, but also on climate mitigation.

Improved nutrient management implies (1) reducing direct discharge of manure without treatment, (2) improving nutrient use efficiencies in agriculture, and (3) improving sewage treatment in the middle-stream and downstream subbasins of rivers. Many technologies exist to support these policies. For example, animal manure can be recycled on cropland after treatment (e.g., composting) (Dong et al., 2018; Hou et al., 2017; Jia, 2014). Nutrient use efficiency of crop production can be improved by fertilizing crops according to their needs for nutrients (Zhang et al., 2012). According to Chinese statistics (MEP, 2013), more than 80% of the wastewater treatment is the secondary treatment in 2012. Nutrients inputs to rivers from sewage can be much reduced by improved wastewater treatment (e.g., tertiary and quaternary treatments) (van Puijenbroek et al., 2019). River export of N and P can be reduced by up to 90% depending on the nutrient forms in China between 2012 and 2050 for SSP1-RCP2.6, in which we assumed direct discharge of manure is completely prohibited. This confirms that current Chinese policies on restricting manure discharge (MOA, 2017) and increase manure recycling (MOA, 2015) are effective and necessary to improve water quality in China.
Climate mitigation is a second way to reduce coastal eutrophication. As indicated above, coastal water pollution may increase as a result of climate change-induced changes in river discharge. This is indicated by the higher river export of TDN (24% higher) and TDP (16% higher) for the higher climate change scenario (SSP2-RCP8.5 vs. SSP2-RCP2.6) in 2050. This is in line with Sinha et al. (2017), who concluded that climate change-induced precipitation changes will likely increase future riverine nitrogen loading to seas in the United States (by about 20%). Thus, to improve water quality in China, not only improved nutrient management in China is needed but also climate change mitigation worldwide. Our most optimistic scenario (SSP1-RCP2.6) indicates that it is possible to reduce nutrient pollution to relatively low levels to avoid coastal eutrophication and to achieve climate mitigation and sustainable socio-economic development.

Global change impacts coastal water pollution not only in China but also in other world regions. For example, Wang, Tang, Burek, Havlík, et al. (2019) applied the MARINA 1.0 model to the Indus River and explored future trends in river export of N for selected SSPs and RCPs. A similar conclusion has been drawn by Wang, Tang, Burek, Havlík, et al. (2019), who conclude that global change can increase coastal eutrophication at the river mouth of Indus. Suggestions for improved policies and technologies were given to the Indus basin based on the modeling results. Strokal et al. (2019) developed and applied the MARINA-Global model for rivers worldwide and quantified river export of multiple pollutants (e.g., nutrients and plastics) in 2010. Applying our approach of scenario analysis based on SSPs and RCPs using MARINA-Global can be the next step to analyze the impacts of global change on coastal water pollution and to identify improved water pollution management options in other world regions.

5. Conclusions

In this study, we described and applied the newly developed MARINA 2.0 model to estimate nutrient export by 12 Chinese Rivers for 2012 and to project nutrient export for 2050. We explore future trends in nutrient export by Chinese rivers as affected by socio-economic and climate change for the SSPs and RCPs. In the following, we summarize our findings for China, which may also be relevant for other world regions.

Rivers and coastal waters in China are highly polluted in 2012. Rivers export 4,854 kton of TDN and 706 kton of TDP to Chinese seas. In the future, rivers and coastal waters may be considerably more polluted or considerably cleaner than in 2012, depending on socio-economic and climate changes. River export of TDN and TDP is projected to be 52% and 56% higher in 2050 than today, respectively, in the scenarios with high challenges toward sustainability (e.g., low nutrient use efficiency in agriculture and poor sewage treatment) (SSP3-RCP8.5 and SSP4-RCP2.6). River export of TDN and TDP is projected to be 56% and 85% lower in 2050 than today in the scenarios assuming more attention for sustainability (SSP1-RCP2.6, SSP2-RCP8.5, and SSP5-RCP8.5). Two thirds of the rivers in 2050 in SSP1-RCP2.6 are projected to have low risks for coastal eutrophication.

Future nutrient management is expected to have a large impact on future river export of TDN and TDP, as indicated by our analysis of future hotspots. The largest hotspot area is found in scenario SSP3-RCP8.5. Our scenario analysis shows that direct discharge of manure may be an important source of water pollution in 2050 if manure management does not improve as shown in SSP3-RCP8.5. This confirms that current policies in China on restricting manure discharge are effective in reducing nutrient water pollution. In addition to implementing and complying to current policies, we suggest further reductions in synthetic fertilizer use, increases in recycling of manure, and improvements in wastewater treatment.

An important conclusion of our study is that global change can make coastal eutrophication control in China more difficult. Our scenarios indicate that climate change may increase nutrient pollution in rivers and coastal seas. Taking, for instance, the storylines for nutrient management in SSP2, we projected river export of TDN is 24% and TDP 16% higher in SSP2-RCP8.5 (more climate change) than in SSP2-RCP2.6 (less climate change). Thus, nutrient management may be less effective in a future world with increased river discharge under severe climate change. Climate mitigation may, therefore, help to improve water quality in rivers and coastal waters of China, and likely of other countries, in the future.

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