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Geophysical Research Letters

RESEARCH LETTER
10.1029/2024GL109621

Sandy Subterranean Estuaries Minimize Groundwater Nitrogen Pollution Impacts on Coastal Waters

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Key Points:

- A 10% increase in economic output enhanced coastal groundwater nitrate by ~24%
- subterranean estuaries (STEs) attenuated 45%–85% of nutrients, yet groundwater remains a crucial nutrient source for coastal waters
- Integrated management is vital to safeguard coastal water quality against polluted groundwater seepage

Supporting Information:

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

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Citation:

Chen, X., Santos, I. R., Du, J., Xu, B., Tamborski, J. J., He, D., et al. (2025). Sandy Subterranean estuaries minimize groundwater nitrogen pollution impacts on coastal waters. *Geophysical Research Letters*, *52*, e2024GL109621. https://doi.org/10.1029/2024GL109621

Received 5 APR 2024
Accepted 13 JAN 2025

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Abstract Widespread anthropogenic activities pollute groundwater that eventually seeps out to the coastal ocean. Here, we resolve nutrient transformations and fluxes in 11 sandy subterranean estuaries (STEs) with contrasting nutrient sources and development trajectories. Coastal groundwater nitrogen pollution stems from sewage discharge and land use change. Anthropogenically derived groundwater nutrient fluxes with high N/P ratios (~170) accounted for 22%–61% of riverine inputs into China's coastal waters, providing an additional source of nutrients that can fuel coastal eutrophication and algal blooms. Sandy STEs remarkably attenuated ~84% of nitrogen pollution, minimizing the impact of submarine groundwater discharge (SGD) on coastal water quality. Hence, STEs deliver an overlooked ecosystem service that is particularly important in highly polluted coastal aquifers. Protecting STEs and recognizing the integrated nature of groundwater and seawater is thus important in coastal water quality management initiatives.

Plain Language Summary The extensive pollution of coastal groundwater is a risk to coastal ecosystems. Subterranean estuaries filter groundwater nutrient pollution, yet their function remains poorly quantified. Our study highlights the influence of anthropogenic activities such as sewage discharge and urbanization in driving coastal groundwater nitrate pollution. Anthropogenically derived nutrients from groundwater contribute significantly to budgets in China's coastal waters. Nevertheless, sandy STEs attenuate up to 84% of nitrogen pollution, effectively mitigating the impact of SGD on coastal water quality. Accordingly, it is imperative to implement targeted measures that go beyond current legislation and initiatives focused on surface water only to ensure the protection of coastal groundwater.

1. Introduction

Coastal eutrophication caused by terrestrial nutrient pollution is prevalent along coastlines worldwide (Dai et al., 2023). About 40% of the world's population lives within 100 km from the coastline. In China, ~44% of the population lives on the coast, contributing to ~60% of the national gross domestic product (GDP) (Wang, Liu, et al., 2014). Due to rapid economic development, most Chinese coastal waters have already reached a state of eutrophication (Wang et al., 2021).

Many earlier studies focused on riverine input and atmospheric deposition as sources of nutrients to coastal waters (Liu et al., 2009; Zhang et al., 2010). However, submarine groundwater discharge (SGD) can also be an important nutrient source (Santos et al., 2021; Slomp & Van Cappellen, 2004). Nutrient fluxes from SGD can exceed rivers on local (Chen et al., 2018; Luijendijk et al., 2020; Xu et al., 2024), regional (Rodellas et al., 2015; Yu et al., 2024; Zhang et al., 2020), and global scales (Cho et al., 2018; Wilson et al., 2024). Large SGD-derived nutrient fluxes

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can trigger and sustain coastal algal blooms (Chen, Cukrov, et al., 2020; Hu et al., 2006; Luo & Jiao, 2016). Such blooms have caused significant social, environmental and economic impacts worldwide (Xiao et al., 2021).

Subterranean estuaries are locations where fresh and saline groundwater mix in coastal aquifers (Duque et al., 2020; Moore, 1999; Rocha et al., 2021). Nutrient enrichment in STEs can be driven by both natural (e.g., seawater infiltration into coastal aquifers; A. M. Wilson and Morris, 2012; Anwar et al., 2014), and anthropogenic factors (e.g., sewage and aquaculture inputs, and land use change; Corbett et al., 2002; Bishop et al., 2017; Chen et al., 2018). These natural and anthropogenic drivers interact to control nutrient speciation and concentrations (Chen, Ye, et al., 2020; Robinson et al., 2018).

Nitrogen attenuation in STEs is crucial to mitigate nitrogen pollution in coastal waters (Kroeger & Charette, 2008; Loveless & Oldham, 2010; Schutte et al., 2015). Previous studies investigated various factors influencing nitrogen attenuation in coastal groundwater, including drivers such as tidal pumping (Heiss et al., 2017; Liu, Su, et al., 2017), fluxes (Spiteri et al., 2008; Talbot et al., 2003), and mechanisms involving denitrification, adsorption, and biota assimilation (Erler et al., 2014; Kim et al., 2017; Xiao et al., 2018). Nevertheless, these earlier studies have mostly been focused on a site-by-site basis.

Here, our comprehensive investigation across 11 sites along the Chinese coastline allows direct comparisons across pollution gradients, thereby facilitating mechanistic assessments at both regional and national scales. Although sandy coastlines account for only 9.8% of China's total coastline, their high permeability makes them significant pathways for transporting groundwater and pollutants to the ocean (Hou et al., 2016). Pollutants, such as nutrients, undergo biogeochemical reactions within the sandy subsurface. Therefore, we hypothesize that sandy STEs attenuate groundwater nitrogen pollution seeping out to the ocean. We resolve natural (e.g., climate and hydrology) and anthropogenic (e.g., socio-economic factors) drivers of groundwater nutrient transformations in sandy STEs across pollution gradients (Figure S1 in Supporting Information S1). Given the long-term economic growth in coastal China (Figure S1 in Supporting Information S1) and other regions, understanding how anthropogenic activities influence SGD-derived nutrient fluxes is crucial for protecting coastal water quality and developing effective management policies.

2. Method

2.1. Study Sites

The 11 sandy STEs sampled here span three climatic zones on the China coast. Dashawa, Shanhaitian, Jinshatan, Dashawan and Zaihaiyifang in the north are located along the Yellow Sea (YS) coast; Nanchangtu and Jihu are located along the East China Sea (ECS) coast; and Jintan, Bailangtan, Yintan and Gaolong Bay in the South are located along the South China Sea (SCS) coast (Figure S1 in Supporting Information S1). These STEs exhibit extensive spatial variations in both natural and anthropogenic factors (Figure S1 in Supporting Information S1), facilitating the assessment of nitrate pollution across pollution gradients. Sandy beach details (including locations, beach characteristics, sediment type, natural or artificial beach, and sampling date) are provided Table S1 in Supporting Information S1.

2.2. Sampling and Analysis

Field investigations were mainly conducted in 2015 and 2016 (Table S1 in Supporting Information S1). Porewater profile samples (0–1.5 m below surface) were collected using a push-point piezometer (Solinst, Charette & Allen, 2006) and a peristaltic pump (Watson-Marlow) at low, mid, high, and above high tide marks at each site. We also collected seawater at each site. ^{222}Rn samples (including surface water) were collected into 40 or 250 mL glass bottles and immediately analyzed using a RAD7 detector (DURRIDGE, Burnett & Dulaiova, 2003). ^{222}Rn concentration was determined based on the measured ^{222}Rn concentration in air using the air-water partitioning of ^{222}Rn (Schubert et al., 2012). Nutrient samples were filtered with 0.45 μm cellulose acetate filters and stored in polyethylene bottles. The nutrient samples were then treated with saturated $HgCl_2$ to prevent biological activity. Nutrient concentrations were analyzed using a flow injection analyzer (SKALAR SAN++) (Liu et al., 2009). Dissolved inorganic nitrogen (DIN) represents the sum of nitrate (NO_3^-), nitrite (NO_2^-) and ammonium (NH_4^+). Dissolved inorganic phosphorus (DIP) and dissolved silicon (DSi) represent the concentrations of phosphate (PO_4^{3-}) and silicic acid ($Si(OH)_4$), respectively. The analytical precision of NO_3^-, NO_2^-, NH_4^+, DIP and DSi was 0.06, 0.01, 0.09, 0.03 and 0.15 μmol L^{-1}, respectively (Liu et al., 2009). The analytical uncertainties of NO_3^-, NO_2^-, NH_4^+, DSi are <1%, and the uncertainty of DIP is <3%. The salinity and temperature of porewater and surface water were measured using a YSI-EC300 A probe (Chen et al., 2019). In addition, sediment cores (1–1.5 m below grade) were collected from each site with a push core approach to determine ^{222}Rn activities (Rn_{eq}) using a sediment equilibration experiment (Corbett et al., 1998). The cores were sliced on-site at 10 cm intervals, and sediment equilibration experiments were conducted on each slice separately.

2.3. ^{222}Rn Residence Time Calculations

Residence time refers to the time elapsed since porewater or groundwater entered the subsurface (Bethke & Johnson, 2008). ^{222}Rn released by sediment ^{226}Ra decay enters the porewater, resulting in enriched ^{222}Rn activity relative to coastal seawater. The ^{222}Rn production rate refers to the rate at which ^{222}Rn is generated from the ^{226}Ra decay in the environment. At equilibrium, the ^{222}Rn production rate equals the rate at which ^{222}Rn decays (Goodridge & Melack, 2014). Thus, ^{222}Rn residence time in STEs can be obtained by the following equation if the magnitude of terrestrial groundwater input to the STE is minor compared to seawater recirculation through the STE (Bertin & Bourg, 1994; Colbert et al., 2008; Goodridge & Melack, 2014):

$$\tau = -\frac{1}{\lambda}\ln\left(1 - \frac{Rn_{pw}}{Rn_{eq}}\right) \quad (1)$$

where τ is the ^{222}Rn residence time (day); λ is the decay constant of ^{222}Rn (0.181 $days^{-1}$); Rn_{pw} is porewater ^{222}Rn activity (Bq m^{-3}); and Rn_{eq} is mean equilibrium ^{222}Rn activity (Bq m^{-3}), determined from sediment equilibration experiments (Corbett et al., 1998). Sandy STEs commonly have significant inputs of terrestrial groundwater, thus requiring the consideration of this mixing process between terrestrial groundwater and seawater. Here, we use the equation of Tamborski et al. (2017b) to consider mixing and calculate ^{222}Rn-derived residence times in the STE. Assuming steady state, the analytical solution yields Rn_{pw} as a function of τ as:

$$Rn_{pw} = Rn_{eq}\left(1 - e^{-\lambda\tau}\right) + Rn_{sw}e^{-\lambda\tau} + f_{gw}\left(Rn_{gw}e^{-\lambda\tau}\right) \quad (2)$$

where Rn_{sw} and Rn_{gw} are the initial ^{222}Rn activities (Bq m^{-3}) in seawater and terrestrial groundwater, and f_{gw} is the fraction of terrestrial groundwater (i.e., fresh groundwater), which can be calculated by a two-endmember mixing equation:

$$\begin{aligned} f_{sw} + f_{gw} &= 1 \\ f_{sw}S_{sw} + f_{gw}S_{gw} &= S_{pw} \end{aligned} \quad (3)$$

where f_{sw} is the fraction of seawater, and S_{sw}, S_{gw} and S_{pw} are the salinity of seawater, terrestrial groundwater and porewater. Here, we calculated residence times using two methods based on Goodridge and Melack (2014) and Tamborski et al. (2017b). Only the latter one considers terrestrial fresh groundwater. Despite the significant positive correlation ($R = 0.98$, $p < 0.0001$) between the results of the two methods, the residence time will be overestimated by ~20% if mixing between fresh groundwater and seawater is not considered (Figure S2 in Supporting Information S1).

2.4. Anthropogenic and Climate Factors and Statistical Analysis

Anthropogenic (e.g., tourism activity, wastewater discharge, industrial output, grain output, agricultural output, ammonia emission, N_2O emission, sewage, GDP, mariculture, urbanization and population) and climate factors (e.g., rainfall, temperature, water resource, sea level, tidal range, tidal level and STE residence time) were collected or estimated from governmental statistical yearbooks, China Ports and Harbors Association, China Shipping Service and China Meteorological Data Service Center as detailed in Table S2 in Supporting Information S1. Pearson's correlation coefficients and corresponding heatmaps were calculated and plotted using Origin software.

2.5. Groundwater Nutrient Endmember

The SGD nutrient flux is equals to the SGD water flux multiplied by the groundwater nutrient endmember in the STE. Salinity exceeding 10‰ (Cho et al., 2018) in 96% of the porewater samples indicate the dominance of saline rather than fresh SGD. The groundwater endmember was determined by calculating the range of nutrient concentrations between the first and third quartiles of the groundwater nutrient dataset, and then subtracting the corresponding seawater concentration under the assumption that saline SGD dominates total SGD (e.g., Chen, Cukrov, et al., 2020; Rodellas et al., 2015). This approach thus offers conservative estimates of SGD-derived nutrient fluxes. These calculations and detailed methodologies are provided in the Supplemental Information.

3. Results and Discussion

3.1. Nutrient Distribution Patterns in Sandy Subterranean Estuaries

Nutrient concentrations and N/P ratios in groundwater are typically higher than those in surface seawater (Santos et al., 2021). Here, coastal groundwater DIN, phosphorus (DIP) and silicon (DSi) values were 189 ± 383, 2.4 ± 3.5 and 114 ± 124 μmol L^{-1}, respectively (Figure 1). Nitrate (NO_3^-), ammonium (NH_4^+) and nitrite (NO_2^-) accounted for 71%, 25% and 4% of DIN in these STEs, respectively. The mean N/P (174 ± 423) and Si/N (1.9 ± 3.5) ratios in groundwater exceeded Redfield proportions (Si:N:P = 15:16:1) (Redfield, 1958). Hence, these groundwater nutrients have the potential to modify biomass and phytoplankton community structure once they reach seawater (Howarth & Marino, 2006). For example, groundwater-derived nutrient fluxes with a higher Si/N ratio promote the growth of diatoms (e.g., *Skeletonema costatum*), which are unicellular phytoplankton with silica cell walls, but inhibit dinoflagellates (e.g., *Prorocentrum dentatum*), which are a type of single-celled marine algae (Chen et al., 2019; Kristiansen & Hoell, 2002).

Nutrient concentrations and ratios showed large spatial variability (Figures 1 and 2a). The mean groundwater DIN concentrations across the 11 sandy STE sites ranged from 148 to 405 μmol L^{-1}. NO_3^- (~88–91%) was the major DIN species in the YS and ECS, while NH_4^+ (~59%) was dominant in the SCS. NO_2^- (<10%) had a small contribution to DIN in all regions. Groundwater DIP concentration in the YS and ECS (1.1 ± 1.0 μmol L^{-1}) was significantly lower than that in the SCS (3.5 ± 4.8 μmol L^{-1}; One-Way ANOVA, $R^2 = 0.09$, $p < 0.01$). Groundwater DSi concentration did not show clear spatial gradients due to a lack of major anthropogenic sources and similar sand composition (Van Gosen et al., 2019). Overall, nutrient concentrations in groundwater at the 11 STEs were 22 ± 42 times greater than those in coastal seawater (Figure 2b).

Based on the Redfield ratio (N/P/Si = 16/1/15), Si-enriched conditions occurred in 51% of groundwater samples, while N-enriched and P-enriched conditions occurred in 36% and 13%, respectively (Figure 2c). Groundwater along the highly populated YS and ECS coastal zones had similar nutrient ratios, with 66%–78% of the groundwater samples having N-enriched conditions, 22%–33% being Si-enriched, and only 0%–1% being P-enriched (Figure 2c), suggesting substantial nitrogen sources. The coastal ocean is usually characterized by N-limited conditions in the absence of anthropogenic nitrogen sources (Santos et al., 2021). Our results demonstrated that nitrogen-rich groundwater with high N/P ratios (~83% of the samples had ratios greater than 16) flows into coastal waters. These nutrients thus fuel primary production, change biological communities, and potentially enhance red tides and seasonal hypoxia (e.g., Chen, Cukrov, et al., 2020; Tamborski et al., 2017a).

3.2. Nutrient Transformations Driven by Hydrological and Biogeochemical Processes

The mixing of nutrient-rich coastal groundwater with comparatively nutrient-deplete seawater provides opportunities to assess biogeochemical processing within STEs (Chen, Ye, et al., 2020). Mixing models along salinity gradients have been widely used to examine nutrient behaviors in estuaries and define source or sink trends (e.g., Boyle et al., 1974; Reithmaier et al., 2021). For example, NO_3^- versus salinity scatter plots showed NO_3^- attenuation in contaminated STEs (Loveless & Oldham, 2010; Oehler et al., 2021), while NO_3^- was produced in an uncontaminated STE (Santos et al., 2009) due to organic matter remineralization (Robinson et al., 2018).

Here, groundwater NO_3^- exhibited a significant exponential decay trend with salinity which implies removal within STEs (Figure 2d). High concentrations of NO_3^- in terrestrial groundwater were attenuated during mixing in 10 out of 11 STEs (Figure 3) prior to discharge to coastal waters. The removal of nitrate is likely attributed to microbial denitrification under anaerobic conditions and high organic matter content (Adyasari et al., 2019;

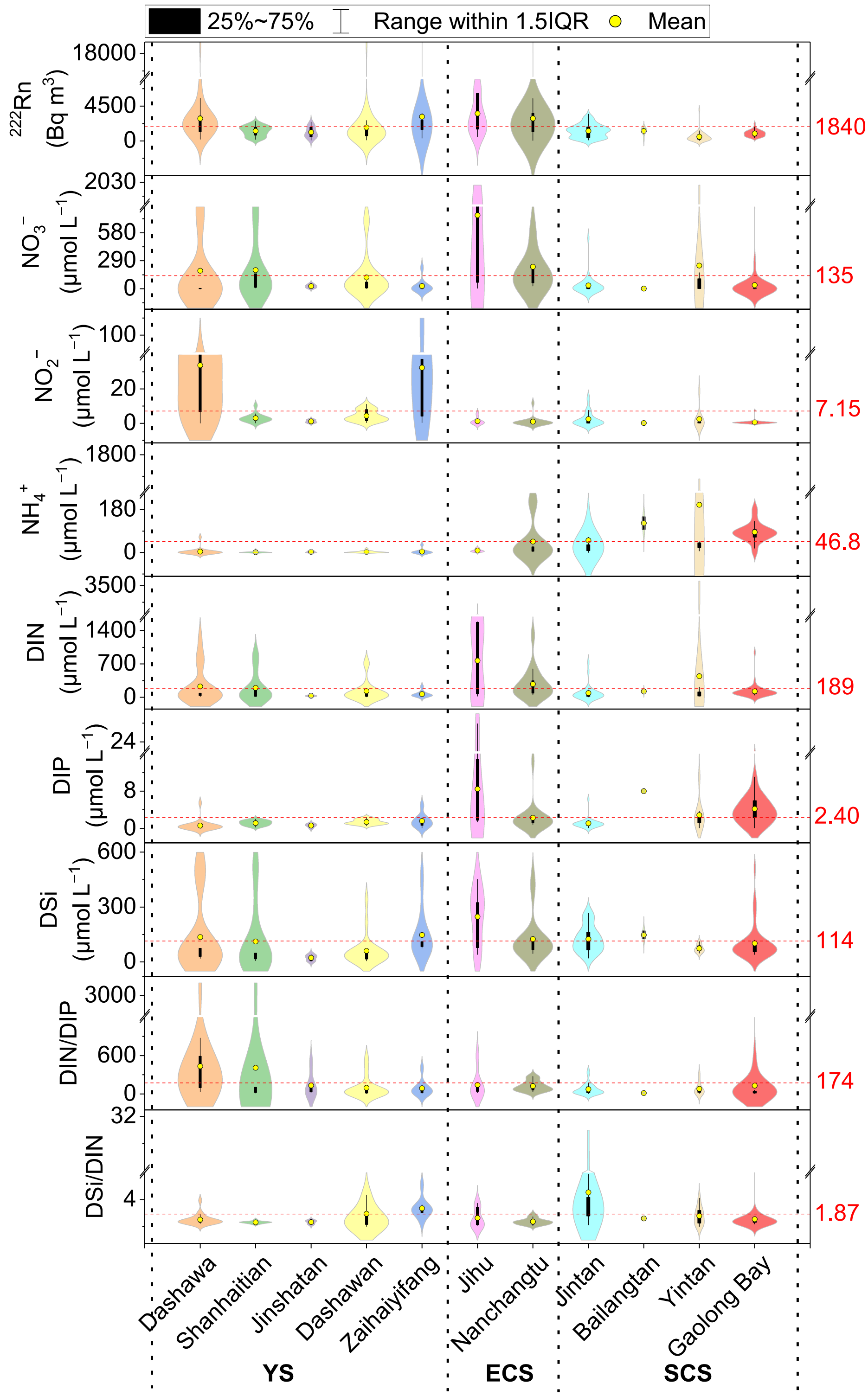

Figure 1. Violin plots for porewater nutrient concentrations and ratios at the eleven sandy beach sites. Red dotted lines represent the mean porewater nutrient concentrations and ratios of the eleven sandy beach sampling sites. Yellow Sea (YS), East China Sea (ECS), and the South China Sea respectively represent the YS, ECS, and South China Sea.

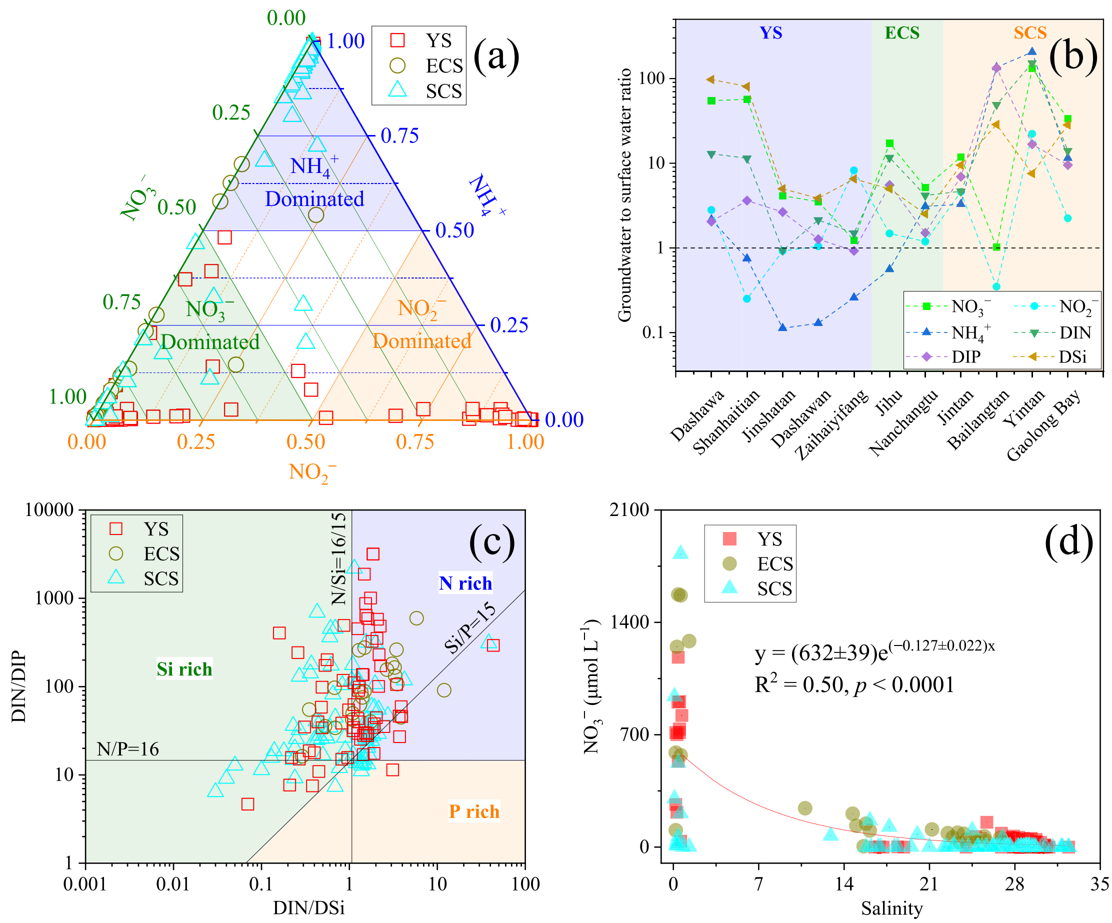

Figure 2. Groundwater nitrogen speciation (a), groundwater to surface water ratios (b), and groundwater nutrient limitation (c) in the eleven sandy beach subterranean estuaries. The relationship between groundwater nitrate and salinity, the fitting curve shows a significant exponential decay of all coastal groundwater nitrates with salinity (d).

Chen, Ye, et al., 2020; Slomp & Van Cappellen, 2004). While fresh groundwater is also enriched in silicon, DSi showed conservative behavior in 6 out of 11 STEs (Figure 3). Dissolved silicon addition in some STEs may be due to lithogenic dissolution of coastal sediments (Rahman et al., 2019).

Quantitative information about biogeochemical reaction rates and residence times in STEs remains limited (Goodridge & Melack, 2014; Ibánhez & Rocha, 2017). We coupled ^{222}Rn-derived groundwater residence times with nutrient distributions to assess how groundwater flow through the STE modifies nutrient behavior (Figure S4 in Supporting Information S1). The mean residence time across the 11 sandy STE sites (0.6 ± 0.4 to 5.3 ± 4.4 days) showed large spatial variability. Overall, DIN, NO_3^-, NH_4^+ and DIP concentrations decreased with longer residence times that enable microbial reactions. Therefore, the rapid attenuation of groundwater nitrogen and phosphorus in STEs further indicates significant removal during transport from land to the ocean. In addition, DSi showed a significant increase with residence times indicating the dissolution of coastal sediments (Rahman et al., 2019).

3.3. Anthropogenic Groundwater Nutrient Enrichment

We explore how nutrient enrichments in STEs are influenced by 17 anthropogenic factors and 10 natural factors (Figure 4), building on similar large-scale approaches focusing on surface estuaries (Bricker et al., 2008). Groundwater NO_3^- was enhanced by sewage (r = 0.75; $p < 0.01$) and GDP (r = 0.75; $p < 0.01$) (Figure 4), indicating that the development of China's coastal economy impacts coastal groundwater NO_3^- pollution (Bu et al., 2021). A positive correlation was also found between groundwater NO_3^- and annual tourist volume (r = 0.70; $p < 0.05$) and population density (r = 0.68; $p < 0.05$) (Figure 4). For example, the 6% increase in coastal

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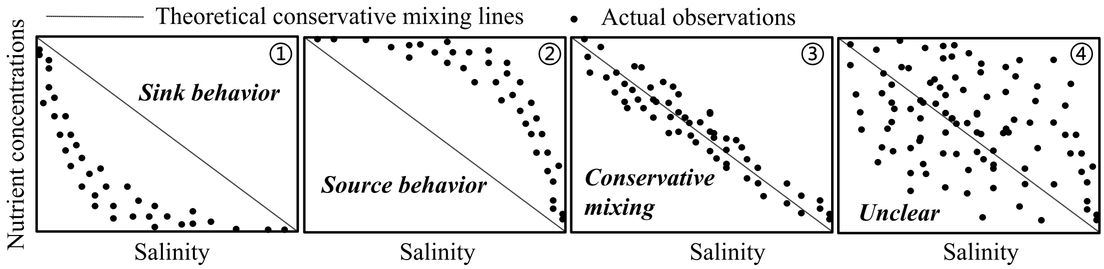

Site	NO_3^-	NO_2^-	NH_4^+	DIN	DIP	DSi
YS						
Dashawa	①*****	④	②	①*****	①*****	③*****
Shanhaitian	③*****	④	④	③*****	②****	③*****
Jinshatan	①*****	①	④	①*****	①	①***
Dashawan	①	②*****	④	①*****	④	①*****
Zaihaiyifang	①*****	①**	②	①*	④	①**
ECS						
Jihu	①	②	①	①	①	③*
Nanchangtu	①*****	②	②	①*****	①**	①*****
SCS						
Jintan	①*	④	①**	①****	①****	③*****
Bailangtan	①	①*	①	①	①	③*
Yintan	①*****	②	①*****	①*****	③****	④
Gaolong Bay	①*****	①****	②*****	①	①*****	③*****

Figure 3. Nutrient source-sink behaviors in the eleven sandy beach subterranean estuaries. Statistical levels were associated with the following p-values <0.05*, <0.01**, <0.001***, <0.0001****, and <0.00001*****. The multiple nutrient source-sink behaviors of each study site are shown Figure S3 in Supporting Information S1.

population density and 100% increase in China's GDP in the last decade enhanced coastal groundwater NO_3^- by ~10% and ~240%, respectively. The links between groundwater nutrients and Chinese socio-economic indicators are much more distinct than those found in global datasets, which often use diverse methods and inconsistent sampling densities (Wilson et al., 2024).

Coastal groundwater NH_4^+ was related to both natural factors such as rainfall and temperature, and human activities such as sewage and wastewater discharge (Figures S5, S6 in Supporting Information S1). Increases in rainfall and temperature (r = 0.71; p < 0.05) promote dissimilatory nitrate reduction to ammonium (Lai et al., 2021) and degradation of organic matter (AminiTabrizi et al., 2022), enhancing groundwater NH_4^+ (Figures S5, S6 in Supporting Information S1). In addition, there is a strong negative correlation between NH_4^+ and marine drivers (e.g., tides, sea level) and anthropogenic factors (e.g., tourism, wastewater, population) (Figures S5, S6 in Supporting Information S1). Hence, marine driving forces release NH_4^+ to groundwater, while anthropogenic factors primarily elevate nitrate.

Terrestrial phosphorus can be removed in sandy STEs through precipitation onto Fe-oxides or co-precipitation with dissolved Al and Ca minerals (Slomp & Van Cappellen, 2004). Here, groundwater DIP correlated with rainfall (r = 0.61; p < 0.05) and per capita water use (r = 0.65; p < 0.05). Rainfall flushes terrestrial fresh water with high DIP through coastal aquifers (Greenaway & Gordon-Smith, 2006). There was also a strong positive correlation (r = 0.74; p < 0.01) between groundwater DSi and rainfall (Figures S5, S6 in Supporting Information S1.) due to the delivery of silicon from terrestrial mineral weathering. Groundwater DSi was also related to urbanization likely due to enhanced soil erosion and rock weathering (Maguire & Fulweiler, 2019). Nitrogen fertilizers, industrial wastewater discharge and land use change enhance DIN/DIP and DSi/DIN ratios in coastal groundwater relative to those in coastal seawater, threatening the nutrient structure of coastal waters (Figure 4). For example, large amounts of groundwater nutrients with high N/P ratio (~174) entering the YS fuel large-scale

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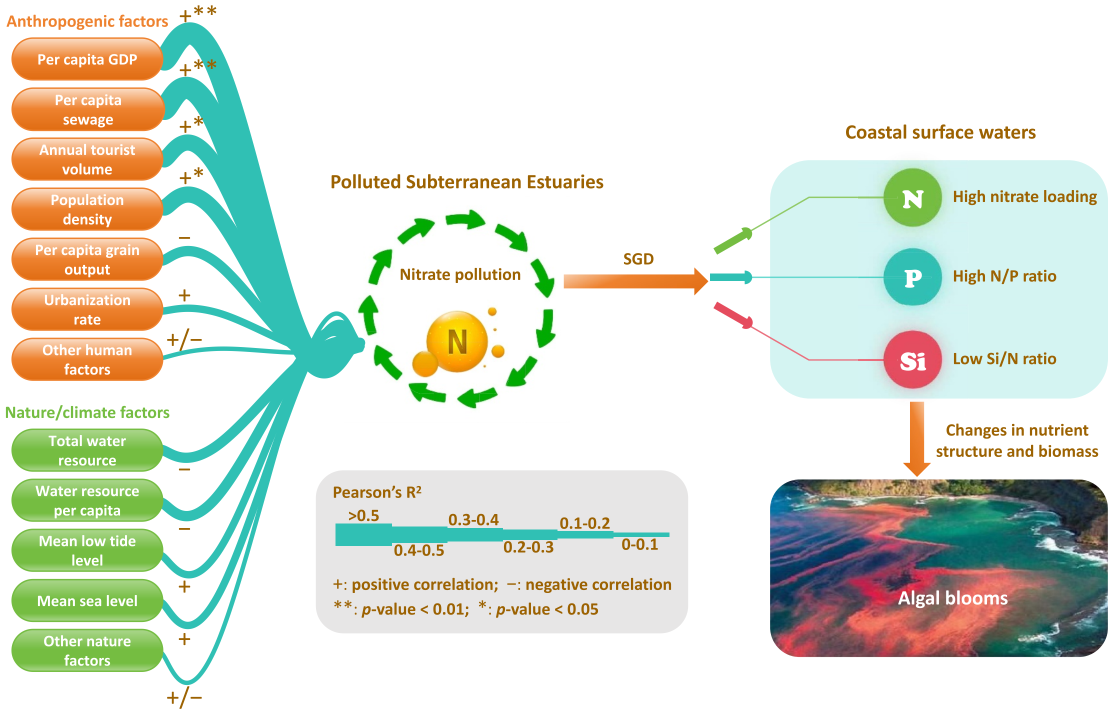

Figure 4. Impacts of anthropogenic and natural drivers of nitrate concentration in eleven sandy beach sites and their potential environmental responses. The "+" and "−" signs indicate whether a specific driver is positively or negatively correlated with nitrate concentration. The width of the blue flow corresponds to the Pearson's R^2 values. Statistical significance are assumed at *p*-values <0.05* or <0.01**.

bloom outbreaks of *Ulva prolifera*. This elevated N/P ratio, exceeding the optimal ratio for phytoplankton growth, creates a nutrient imbalance that encourages growth of *Ulva prolifera* (Liu, Jiao, et al., 2017; Wang et al., 2021; Zhao et al., 2021).

3.4. Groundwater-Derived Nutrient Fluxes

What is the fate of groundwater-borne nutrients as they traverse sandy STEs and enter coastal waters? Zhang et al. (2020) summarized coastal groundwater data for the entire coastline of China and estimated a coastal groundwater flux of 7.8×10^{12} m^3 yr^{-1}. Sandy coastlines account for only 9.8% of the total Chinese coastline (19.7×10^3 km) (Hou et al., 2016). Because sandy coastal sediments usually have high permeability, we conservatively assume that at least 9.8% of this SGD occurs through sandy coasts. By combing the groundwater flux through Chinese sandy coastlines and groundwater nutrient endmembers, we estimated that the DIN, DIP and DSi fluxes through coastal sandy aquifers at the China-scale were $(1.60–3.81) \times 10^{10}$ $(0.04–0.08) \times 10^{10}$ and $(2.91–5.13) \times 10^{10}$, respectively. Despite most samples being collected from November to June, the lower SGD nutrient fluxes typically observed during the dry season (November to April) along the Chinese coast (e.g., Chen, Jiang, et al., 2024; Liu et al., 2018) suggest that our calculations are conservative. Based on our nutrient source-sink analysis (Figure 3), if there were no nutrient attenuation during mixing within STEs, nutrient concentrations would increase significantly, leading to a 1–5 times increase in the flux of these nutrients.

These coastal groundwater-derived DIN, DIP, and DSi fluxes were equivalent to approximately 13%–31%, 39%–83% and 18%–32% of riverine inputs into the coastal ocean in China, respectively (Liu et al., 2009). Hence, SGD through sandy coastlines represents another significant source of nutrients to Chinese coastal waters and add to large fluxes observed in muddy coastlines dominated by saltmarshes (Chen et al., 2021, 2024a; Wang et al., 2022). Compared with river water and seawater, coastal groundwaters have much higher N/P ratios (∼174). Therefore, coastal groundwater (i.e., SGD) may contribute to persistent water quality issues in the coastal ocean (Van Meter et al., 2018), and should be considered in water quality management plans (Santos et al., 2021) particularly in sandy coastlines that are highly permeable and attractive to urban expansion.

3.5. Implications for Coastal Groundwater Quality and Management

Our observations from 11 STEs in China revealed complex source-sink nutrient behavior and large fluxes of groundwater nutrients driven by anthropogenic and climate factors. Accelerating industrial and agricultural activities drive nitrogen pollution in coastal groundwater. While STEs attenuate approximately 84%, 85% and 45% of the fluxes of DIN, DIP, and DSi, respectively, SGD is still a major nutrient source to the coastal ocean. As a result, it is necessary to minimize sources and develop efficient sewage treatment. For example, fertilizers should be used rationally in agricultural production (Schlesinger, 2009). The booming tourism industry also has a major impact on groundwater nutrient pollution. Urban sewage treatment is required especially during the peak tourist season. Long-term coastal groundwater monitoring programs should be established particularly in areas with high industrial and agricultural production to inform water quality management programs that consider groundwater-surface water connectivity.

Coastal groundwater flow and related nutrient fluxes have historically been overlooked (Burnett et al., 2003; Moore, 2010). Quantitative investigations have revealed that coastal groundwater releases nutrients that adversely impact coastal ecosystems. The effects include eutrophication and red tide outbreaks (e.g., Chen, Cukrov, et al., 2020; Lecher et al., 2015; Lee et al., 2010), coastal hypoxia (e.g., Guo et al., 2020; McCoy et al., 2011) and seawater acidification (e.g., Cardenas et al., 2020; Wang, Jing et al., 2014). Coastal groundwater nutrient transport may also have positive impacts such as enhancing primary productivity, fish production or coral calcification (Santos et al., 2021). Therefore, coastal groundwater quality management needs to consider both positive and negative impacts. To our knowledge, coastal groundwater has not been considered in legislation and major water quality management initiatives worldwide (Santos et al., 2021). In September 2021, the State Council of China promulgated the first regulation on groundwater management (State Council of China, 2021). However, the regulation does not explicitly address the management of groundwater connectivity with the ocean, except for monitoring to prevent seawater intrusion. To achieve economic, social, and ecological goals, it is essential to implement targeted measures that address pollution prevention and groundwater treatment, guided by an understanding of coastal groundwater connectivity.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Data Availability Statement

Data are available in the Figshare repository at https://doi.org/10.6084/m9.figshare.25256395.v1 (Chen, Santos, et al., 2024). All anthropogenic and climate factors in Section 2.4 can be found from governmental statistical yearbooks (http://www.stats.gov.cn/sj/ndsj/), the China Ports and Harbours Association (http://www.chinaports.com/), China Shipping Service (https://www.cnss.com.cn/), and the China Meteorological Data Service Center (http://data.cma.cn/).

Acknowledgments

This research was supported by the National Key R&D Program of China (2022YFE0209300) and Natural Science Foundation of China (42476151).

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