Assessment of water pollution in the Tibetan Plateau with contributions from agricultural and economic sectors: a case study of Lhasa River Basin

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Abstract

The freshwater environment of watersheds in the Tibetan Plateau is bound with the safety of the Asian Water Tower. In this study, nitrogen (N) and phosphorus (P) loads delivered to freshwater and the associated gray water footprint (GWF) in the agriculture, tourism, domestic life, and industrial sectors were estimated to assess the seasonal and annual characteristics of the water pollution levels (WPLs) in the Lhasa River Basin from 2006 to 2018, and WPL calculations were compared with actual water quality measurements from 2017 to 2018. We found that more than 90% of the GWF came from anthropogenic sources. From the perspective of the whole basin, domestic life was the largest contributor to both N-related GWFs (52%) and P-related GWFs (50%), followed by agriculture for N-related GWFs (32%) and tourism industry for P-related GWFs (30%). The N emissions into the freshwater environment exceeded the maximum assimilation capacity of the watersheds in individual years at both seasonal and annual scales, while P emissions were completely within the pollution assimilative capacity. Besides, we found the serious N pollution near irrigation areas at the seasonal scale (WPL = 2.7 and TN = 1.11 mg/L). The prosperity of tourism has led to a tenfold increase in N-related GWFs and a fivefold increase in P-related GWFs for the tourism industry near the Lhasa city. The strict top-down unified management for ecological environmental protection in plateaus may be an effective method.

Keywords Gray water footprint · Water pollution level · Agriculture · Tourism industry · Lhasa River Basin

Introduction

Nitrogen (N) and phosphorus (P) are two indispensable nutrient elements for food production and the global population (Simons et al. 2014; Davidson et al. 2016; Zheng et al. 2018). However, large portions of nutrients that are applied to crops in the form of fertilizers and used for human lives and production end up in freshwater bodies as contaminants. Excessive nutrient loads that enter freshwater resources may exceed natural assimilation capacities and thus pose a threat to water availability quantities and freshwater ecosystems (Mekonnen and Hoekstra 2015; Suttle 2010; Paerl and Otten 2013), as the nutrients may not stop in aquatic media but end up in aquatic biota (Ferreira et al. 2017).

Rivers and their associated basins are carriers of concentrated pollutants (Howarth et al. 2000). Two-thirds of the world's rivers experience human-induced eutrophication (Smith 2003; Liu et al. 2012). For watersheds with ecological protection requirements, it is important and difficult to

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ensure healthy water environments in the face of social development. The Lhasa River Basin is a typical water conservation area on the Tibetan Plateau and is one of the important water supply areas in Asia (Chen et al. 2019). It is also the most densely populated area on the plateau. The Lhasa River is the receiving body of many types of waste and sewage including nutrient pollution from farmland and concentrated emissions from economic sectors. To achieve the regional goal of building high-quality farmland, an increasing number of the nutrient content is applied to the infertile soil in the form of fertilizer (Li et al. 2020). The excessive fertilizer may enter the freshwater environment through leaching, resulting in the non-point source pollution. The thriving tourism industry causes more pollutant emissions from accommodation and restaurants into the water environment. By 2020, the number of tourists has reached 20.02 million. In addition, due to the backward facilities, the industrial and domestic sewage are basically discharged directly, and the overall centralized treatment rate of urban sewage is even below 50% (Ding 2005). Therefore, it is necessary to quantify the amounts of N and P emissions along the production chain and consumption amounts based on the regional nutrient budget to determine whether the pollutants emitted exceed the maximum assimilation capacity of the watershed.

The concept of the gray water footprint (GWF) solves the problem of measuring pollutant burdens in freshwater resources. This is defined as the volume of freshwater that is required to assimilate the pollutant load based on natural background concentrations and existing ambient water quality standards (Hoekstra et al. 2011). The natural concentration is a measure of the chemical components before the anthropogenic disturbances. Only the concentration in rivers beyond the natural background concentration can be regarded as contaminant (Gao et al. 2020). The GWF indicator has been widely used for examining pollutant loads in freshwater at global scales (Mekonnen and Hoekstra 2018; Ming et al. 2019), regional scales (Johnson and Mehrvar 2019; Zhang et al. 2019), and watershed scales (Pellicer-Martínez and Martínez-Paz 2016; Aldaya et al. 2020). The water pollution level (WPL) metric is often used together with the GWF to measure the waste assimilation capacity of watersheds and the WPL is calculated as the ratio of GWF to runoff (Hoekstra et al. 2011). If the WPL is greater than 1, then the pollutants emitted exceed the maximum assimilation capacity of the watershed under a defined water quality standard and violate freshwater environmental safety. The GWFs and WPLs related to individual sector (Moree et al. 2013; Hu et al. 2018) or anthropogenic loads (Bouwman et al. 2005; Mockler et al. 2017) have been addressed in the previous studies, but systematic studies that consider temporal variations in nutrient pollution loads into rivers from all sectors (i.e., agricultural, domestic, and industrial) and

identify the anthropogenic loads of watersheds are scarce. More importantly, we argue that water pollution assessments should take seasonal scales into account, to address issues in those sectors that exhibit obvious seasonality, such as the agriculture and tourism industries. Especially for plateau areas at high altitudes, low temperatures cause periods of human activity to be more concentrated and shorter, and then river loads are more likely to exceed the seasonal dilution capacity of the basin. Another novelty is that we combine the theoretical calculation indicators with measured water qualities. Considering these two, the reliability of indicator calculations can be evaluated based on actual measurements. On the other hand, water quality measurements can compensate for the shortcomings of indicators at spatial scales. Especially for watersheds, it is important to identify which sections of rivers have problems of exceeding assimilation capacities. This can lead to more targeted recommendations for watershed management.

The purpose of this research is, therefore, to provide details of the temporal variations in nutrient element emissions of the GWF and WPL for the economic (i.e., tourism, domestic, and industrial) and agricultural sectors, quantify their anthropogenic loads, assess the seasonal characteristics, compare the results with real water quality measurements in water bodies, and finally identify the specific sectors that are responsible for river pollution.

Material and methods

Study area

The Lhasa River Basin is located on the southwestern Tibetan Plateau at 90° 05' E–93° 20' E and 29° 20' N–31° 15' N and is the largest and most important tributary of the Yarlung Zangbo River (Fig. 1a). High mountains and deep gorges are distributed as a mosaic with an obvious slope from the northeast (over 7000 m) to southwest (lower than 3700 m). Human activities are concentrated in the lower reach, including six counties and one city. Farmlands are distributed on both sides of the riverbanks (Fig. 1b), which are Moda and Pengbo irrigation areas. There are two hydrological stations (i.e., Tanggya station and Lhasa station). The climate has the characteristics of a plateau temperate semi-arid monsoon climate with an annual average temperature of 7.7 °C and mean annual precipitation of 440 mm.

Modeling nutrient emissions

Nutrient emissions in the basin were modeled for both diffuse loads from agriculture and for point source loads associated with domestic life, tourism, and industry (Fig. 2). The agriculture considered in this study referred

Fig. 1 Study areas: **a** location of the Lhasa River Basin and **b** map showing water quality monitoring sites

to cropland systems. The nutrients entering the freshwater are from the leaching and runoff in cropland systems and the direct discharge from economic sectors. The nutrient flow calculations were based on nitrogen and phosphorus fertilizer application, irrigation area, sown area and grain yield of each crop type, and livestock numbers for every county in Lhasa River Basin from 2006 to 2018, sourced from the Tibet Autonomous Region Bureau of Statistics (2019) and Lhasa Municipal Bureau of Statistics (2014). A number of parameters used for depicting these processes were derived from Chinese governmental reports and published peer-review articles. Crops were aggregated into wheat, highland barley, beans, and oil crops and animals were aggregated into pigs, sheep, and cattle. N and P contents in seeds were estimated by multiplying seed doses by crop sown areas and N and P uptake per crop (Gao et al. 2018; Ma 2018). Atmospheric deposition

rates were 5.45 kg N/ha and 0.39 kg P/ha, respectively (Jia 2008). The biological N fixation rates were divided into leguminous and non-leguminous crops (Lu et al. 1996; Du et al. 2010). The N and P content rates in irrigation water and irrigation quotas per crop were sourced from Tibet Autonomous Region Water Quota (Water Resources Department of Tibet Autonomous Region 2019). Manure inputs were calculated by multiplying livestock numbers, feeding periods, nutrient excretion for every animal (kg N or P per head), and return rates to the field (Liu 2018). The recycling ratio refers to recycling due to manure burning and is based on local customs. The recycling rates of crop residues to the field were 20.2% for N and 67.4% for P, which agrees with the studies reported in Bao et al. (2014). Nutrient removal through harvested crops and crop residues was calculated by multiplying crop yields, crop-specific nutrient contents, ratios of straw to grain, and ratios

Fig. 2 Theoretical framework for nutrient emissions in the Lhasa River Basin

of straw recycling (Liu et al. 2017). N loss through gaseous emissions was estimated based on fertilizer or manure application and was 10% for the NH₃ volatilization rate, 0.86% for the N₂O emission factor of fertilizer, and 1% for manure (Ma et al. 2010). The outputs for erosion were calculated by the total N input with a 0.3% erosion factor. Soil denitrification was calculated based on the difference between N balance and NH₃ volatilization and was 60% for the denitrification fraction. Finally, leaching and runoff of N and P into surface waters were estimated by assuming balance over the long term (Mekonnen and Hoekstra 2015; Mekonnen and Hoekstra 2018).

Estimations of point source pollution were based on the Manual of Production and Discharge Coefficients of Urban Domestic Sources for the first national census of pollution sources. This manual divides the entire country into administrative divisions according to geographical environments, economic levels, and living habits, and provides the living sources, accommodation, and catering industry and industrial discharge coefficients of each region. Then the nutrient emissions from point source were determined by multiplying the discharge coefficients (i.e., domestic, tourism, and industrial sector) and the number of urban residents or tourist arrivals. Anthropogenic nutrient loads were calculated as the sum of all point source pollution and diffuse pollution from anthropogenic sources (i.e., fertilizer, manure, and straw recycling) by multiplying the total leaching, runoff volume, and fraction of nutrient input from anthropogenic sources by the total inputs.

Gray water footprints and water pollution levels

According to Hoekstra et al. (2011), the gray water footprint (GWF, m³/year) is calculated by dividing the nutrient load (*Load*, kg/year) by the difference between the ambient water quality standard for N or P (maximum acceptable concentration $C_{\max}$, mg/L) and the natural concentration of N or P in the receiving water body (C_{nat} , in mg/L):

$$\text{GWF} = \frac{\text{Load}}{(C_{\max} - C_{\text{nat}})} \quad (1)$$

The GWF is highly sensitive to ambient water quality standards (i.e., $C_{\max}$, C_{nat}) of contaminants in water resources (Muratoglu 2020). The parameters are suggested to be determined based on regional agreements or national legislation (Mekonnen et al. 2016). In this study, the natural concentrations were obtained from our water quality measurements in upper basins without human activities and exhibited values of 0.29 mg/L for total nitrogen (TN) and 0.02 mg/L for total phosphorus (TP). The maximum acceptable values were provided based on the regional requirements with values of 1.0 mg/L for TN and 0.2 mg/L for TP, as proposed by the National Department of Ecology and Environment. The water pollution level (WPL) indicates the degree of pollution in a catchment and is the ratio of the GWF in a catchment to runoff from the catchment (R_{act} , m³/year) (Eq. 2). Historical annual runoff and monthly runoff data for the 2006–2018

period were obtained from data measured at hydrological stations.

$$WPL = \frac{GWF}{R_{act}} \quad (2)$$

Water quality measurements

A total of 51 water samples were collected from the upstream watershed to the outlet of the watershed on August 20, 2017, and August 31, 2018. Sampling points included the inlets and outlets of irrigation areas and river water, which almost covered the whole area with human disturbances (Fig. 1b). Water samples from 10-cm depths were collected against the river current. They were stored in polypropylene bottles at 4 °C until laboratory analysis. TN and TP concentrations were measured to quantify their actual contents in the water body.

Results and discussion

Nutrient emissions to freshwater

The total N loss from leaching and runoff in the cropland systems exhibited a downward trend from 2.3 million kg to 1.0 million kg due to the 40% decrease in fertilizer and manure inputs, as shown in Fig. 3a. Ninety percent of the total N loss in freshwater was from anthropogenic sources. The total P output from leaching and runoff presented an increasing first and then declining pattern, with the largest value of 0.07 million kg occurring during 2011–2016 (Fig. 3b). The P fertilizer application were responsible for this variation, contributing 65–80%. Almost all the P leaching and runoff to the freshwater originated from anthropogenic sources. Table 1 shows the variations in point source emissions and total nutrient emissions to freshwater. The total N loads were basically stable at of 3.5 million kg, resulting from the opposite trends of diffuse source loads and point source loads where point source N loads increased by 50%. However, total P emissions into rivers exhibited a sharp upward trend from 1.5 million kg to 2.5 million kg due to the doubling of P emissions from point sources.

Gray water footprint

Figure 4a, b shows the variations in N-related and P-related GWFs from four sectors from 2006 to 2018. The total N-related GWFs were basically stable at 5 Gm³. The N-related GWFs from the agricultural sector were halved from 3.3 to 1.5 Gm³, while those from domestic life continued to increase from 1.8 to 2.3 Gm³ due to population

growth. The N-GWFs related to tourism industry increased by more than 10 times. The GWFs generated by industrial N emissions were stable at between 0.5 and 0.6 Gm³. Compared with N-related GWFs, the total trend of P-related GWFs was different, continuously increased from 0.8 to 1.4 Gm³, and the magnitudes of P-related GWFs were generally smaller. Similar with N-related GWFs, the P-related GWFs from domestic life and the tourism industry both increased from 0.5 to 0.7 Gm³ and from 0.04 to 0.4 Gm³, respectively, and those from industrial P emissions remained unchanged at 0.02 Gm³. The GWFs from agricultural P loads increased from 0.2 to 0.4 Gm³ and then declined to 0.3 Gm³.

The contributions of the four sectors to the total N-related and P-related GWFs are shown in Fig. 4c, d. In 2006, the N-related GWF from agriculture and domestic life accounted for 98% of the total GWF of the watershed. However, domestic life became the leader by 2018, accounting for half of the total GWF, while the contributions from agriculture decreased by nearly half. There was a rapid increase in the share of the tourism industry from 1 to 14%. The contributions of industrial emissions were generally consistent. For P-related GWFs, domestic life was the largest contributor, more than half of the share. The share from the tourism industry increased and became the second largest contributor in 2018. Contributions from agriculture and industry before and after exhibited almost consistent. Overall, domestic life is the largest contributor to both N-related and P-related GWFs. The tourism industry is an emerging sector to focus on and has become the second largest contributor to P-related GWFs. The contribution from agriculture has declined, and the share from industry was small.

Water pollution levels

Figures 5 and 6 show the annual and seasonal WPLs and those from anthropogenic and non-anthropogenic sources. The N-related WPLs were generally less than 1, except for value in 2015. The WPL values in most years remained 0.60. This indicated that the pollution assimilative capacity in the entire basin was basically sufficient to assimilate the N emissions on annual scale. The N-related WPL from anthropogenic sources accounted for about 95% of the total WPL. Human disturbances are still the most primary factor for water environment in plateau. The trends of P-related WPLs did not show the significant fluctuations, with all values less than 1 (Fig. 5b). These WPL values remained at approximately 0.2, which were far less than the N-related WPL values. It could be found that the P loads into freshwater were much lower than the pollution assimilative capacity at the annual scale. Similarly, almost all the P-related WPL was from anthropogenic sources.

The seasonal indicators were based on nutrient loads in the entire basin from April to September, during which

Fig. 3 a, b N and P balance in cropland systems

emissions from the agriculture and tourism sectors were largely concentrated. Seasonal emissions from household life and industry were determined on the proportion of seasonal months to the whole year, and total runoff was also determined from the sum of monthly runoffs in these 6 months. Seasonal N-related GWFs for non-anthropogenic sources accounted for only 6%, and the proportion of seasonal P-related GWFs for non-anthropogenic sources was less than 1%. Thus, the WPL from anthropogenic sources was nearly equal to the seasonal WPL. Generally, seasonal WPL values were larger than annual WPL values. Compared with the annual WPLs, years with seasonal N-related WPLs larger than 1 increased (i.e., 2006, 2007, and 2015), while all

seasonal P-related WPLs did not exceed 1 and were less than 0.3. Overall, the annual and seasonal WPL values related to P loads were satisfactory, which indicated that the P loads were completely within the pollution assimilative capacity. However, N emissions still need to be controlled to avoid the water pollution in some years.

Water quality analysis

Figure 7 shows the spatial distributions of TN and TP parameters in 2017 and 2018. According to the national environmental quality standards for surface water, the standard III requires TN concentrations to be less than 1.0 mg/L

Table 1 Total nutrient loads and those from point sources

	N loads (million kg)		P loads (million kg)	
	Point source	Total	Point source	Total
2006	1.37	3.70	0.11	0.15
2007	1.44	3.37	0.12	0.14
2008	1.52	3.46	0.12	0.14
2009	1.61	3.67	0.13	0.16
2010	1.52	3.05	0.12	0.15
2011	1.57	3.12	0.13	0.19
2012	1.62	3.30	0.14	0.20
2013	1.71	3.47	0.15	0.22
2014	1.76	3.53	0.15	0.22
2015	1.86	3.63	0.17	0.23
2016	1.93	3.52	0.18	0.24
2017	2.00	3.56	0.18	0.23
2018	2.14	3.16	0.20	0.25

and TP concentrations to be less than 0.2 mg/L. The water quality results in 2017 and 2018 exhibited similar spatial patterns. In the upper reaches of the mainstream, TN concentrations were below 0.5 mg/L due to the little human activities limited by the high altitude. In the middle part, TN values in the Pengbo River increased to 1.0 mg/L when through the Pengbo irrigation area but reverted to the III standard at the river confluence. The water quality of another tributary Mozhu River was relatively high, with TN values of 0.5–0.8 mg/L. Along the river flow, TN concentrations

increased to greater than 1 mg/L in the Chengguanqu District due to discharges of urban pollution sources. Finally, the TN concentrations at the outlet of the river basin were diluted to approximately 0.8 mg/L and were due to the assimilative capacity of the river itself. The TP concentrations of all water samples in the entire basin reached the III standard. The spatial trend in 2018 was similar to that in 2017, but the specific values might be higher. For example, TN values in the middle part of the Pengbo River ranged from 1.0 to 1.5 mg/L. In total, the water quality of the whole basin reached the national standard. The TP concentration presented a satisfying result. But in some river fragments, TN contents exceeded the assimilative capacity of the river, especially in the Chengguanqu District and Pengbo irrigation area.

Comparison between WPL and water quality parameters

The results of theoretical seasonal WPL calculations were consistent with those of TN and TP concentrations in freshwater systems from 2017 to 2018. Although water quality measurement took place in late August 2017 and 2018 and cannot represent seasonal averages, it represented the freshwater environment quality in the late period of pollution discharge as a preliminary attempt. By comparison, it can be found that the theoretical seasonal WPL values were generally below than 1 and that the TN and TP contents at the basin outlet also met the standard III. Both results indicated

Fig. 4 GWFs from four sectors and their contributions: **a** N-related GWFs in four sectors, **b** P-related GWFs in four sectors, **c** the contributions to N-related GWFs, and **d** the contributions to P-related GWFs

Fig. 5 a, b Annual WPLs that are related to nutrient loads from anthropogenic and non-anthropogenic sources

Fig. 6 Seasonal WPLs that are related to nutrient loads from anthropogenic and non-anthropogenic sources

that when considering the whole river basin, the freshwater environment was healthy. However, through spatial measurements, it was worth noting that there were obvious water quality problems in the Pengbo tributary and mainstream reach, which were discussed in detail below.

Nutrient pollution from irrigated areas

Through the actual water quality data, serious N pollution was found in the reach of the Pengbo River (1.0–1.5 mg/L), located at the Pengbo irrigation area. Therefore, we further explored the pressure exerted by irrigation areas on the water environment. Due to a lack of runoff data for the Pengbo tributary, we regarded the differences in river discharge between Lhasa Station and Tanggya Station as the sums

of runoff from the Pengbo tributary and Mozhu tributary (Fig. 1). These two tributaries run through Lhünzhub and Maizhokunggar Counties, both of which are the main irrigation areas. Tables 2 and 3 show the annual and seasonal GWF and WPL values in irrigation areas.

The results indicated that although the N-related GWFs in irrigation areas have continued to decline, the WPLs from N load were still larger than 1. The average annual N-related WPLs generally remained at 2.2, while seasonal WPLs were more serious with an average value of 2.7. This indicated that N-related pollution caused by agriculture in some river reaches was quite serious. Compared with the results for N, P-related WPLs were all less than 1, which means that the agricultural development has not brought P pollution to the local water environment.

Agriculture has caused severe N pollution to the freshwater environment near irrigation areas. The application of fertilizers and manure is primarily responsible for the high N loads in the local rivers. Arable land in Tibet is scarce and relatively barren, with only 2.4% soil organic matter (Dai et al., 2011). The plateau's unique dietary habits (i.e., highland barley and wheat) and remote geographical position result in difficulties in meeting local food needs by transportation (Gu 2000). Therefore, fertilizers and manure become the only guarantees for food production, but they also lead to higher losses of N and P into freshwater due to severe soil erosion. Chen et al. (2018) found that significant portions of fertilizer were lost and were not absorbed by crops. Although the diffuse source pollution from agricultural sector has no obviously negative impact on the freshwater quality from the perspective of the whole basin, these pollution problems in some reaches are prominent, especially near the Pengbo tributary in Lhünzhub County, which is also the habitat of an endangered species called *Grus nigricollis* (Bishop and Li 2002).

Nutrient pollution from tourism industry

The tourism industry is the pillar industry of the plateau economy, and the tourist population is mainly concentrated in Lhasa city. Table 4 shows the seasonal WPLs and annual WPLs in Chengguan District. The results of WPL calculations were consistent with the measured water quality, and the annual and seasonal WPLs were all lower than 1 for both N and P. However, as the urban area is close to the river outlet to the Yarlung Zangbo River, it is necessary to understand the emission intensity of tourism.

GWFs due to tourism have increased exponentially in recent years (Fig. 8). For the N-related GWFs, although those from domestic life ranked at the top, those from tourism increased by more than 10 times and their proportion increased from 5.1 to 31.6% (Table 5). Especially for P-related GWFs, tourism has outpaced domestic life and has increased from 10.8 to 54.6%. Since the location of Lhasa city is near the outlet, the tourism poses a potential threat to the water environment security of the Yarlung Zangbo River. Even during the COVID-19 outbreak in 2020, tourism visitation in Lhasa reached a new high of 2.02 million, which was 40 times larger than the resident population (People's Government of Lhasa Municipality 2017). The large tourist flow inevitably caused considerable food consumption and waste in restaurant and accommodation services, especially for large restaurants (Gao et al. 2017). In addition, the low rate of urban sewage treatment, slow construction of facilities, and direct discharge of wastewater increased the burden on the water environment in the basin (Zhang et al. 2016).

Implications for basin management

Our findings indicate that N concentration in the rivers near the irrigation area exceed the national standard. This requires the local government to strictly control N emissions. It is the most direct and effective method to continuously reduce the amount of N fertilizer and adjust the structure of chemical fertilizers, despite the Tibet has met the target of the “Zero Increase Action Plan” (Jin and Zhou 2018). The unified management for chemical fertilizer application should also be implemented. Due to the relatively low soil fertility, the government should conduct soil testing and set fertilizer standards (Sun and Huang 2012). Farmers can apply to their local committees for the required fertilizer based on the amount of arable land. The committees should keep detailed records of fertilizer sources and strictly control the channels through which farmers buy fertilizer themselves. The fertilization schedule can be made in each village. In addition to controlling at the source, it can also reduce N loss via erosion and runoff at the endpoint. For example, the appropriate conversion between farmland and forest will contribute to the decrease in the number of fertilizers used. The reforestation on river floodplains may have positive impacts on soil erosion, improving the soil fertility (Liu et al. 2017). They both can stop nutrient pollutants entering the freshwater body, further decreasing the GWFs. These projects also meet the plateau's requirements for wind protection and sand fixation.

Tourism contributes enormously to economic growth on the plateau. However, there are some potential hazardous effects on the water environment that require government attention, although P emissions from tourism are within the assimilation capacity of rivers. Ensuring sustainable development of the tourism industry is the primary issue (Timur and Getz 2009; Zhang et al. 2015). Comprehensive and efficient tourism administration is of great significance for sustainability due to the closed geographical location on the plateau. Due to the fragile ecological environment, reasonable control of tourist flow should be considered. It is also important to increase publicity for tourists to enhance their environmental protection awareness, such as the “clear your plate” action. Additionally, dumping untreated waste directly into surface water still exists based on local surveys. Thus, wastewater technology improvements and centralized garbage treatment should be strengthened to avoid the direct discharge of domestic sewage into surface water.

Uncertainties and limitations

There are inevitable uncertainties in the process of the estimated N and P loads to freshwater and WPL calculations. These uncertainties mainly arise from the

Fig. 7 Distributions of TN and TP in the mainstream, Pengbo River, and Mozhu River in 2017 and 2018. **a–c** TN distribution in 2017. **d–f** TP distribution in 2017. **g–i** TN distribution in 2018. **j–l** TP distribution in 2018

data uncertainties and the simplifying calculations. Firstly, the used parameters such as N or P uptake per crop, biological N fixation rates, and nutrient excretion for livestock are basically taken from province level or national level, which actually vary among the

cities or producing areas (Hu et al. 2018). Moreover, we assume a steady-state condition during the period of 2006–2018 for some temporal differentiated parameters (Aldaya et al. 2020), because it is difficult to collect these parameters for different time periods. For example, atmospheric deposition rates can vary with annual climate conditions and the ratio of straw recycling is consistently changing promoted by governmental departments (Li et al. 2011). On the other hand, due to current data availability, we have to make

Table 2 N-related GWFs and WPLs in irrigation areas

	Gray water footprint in two irrigation districts (billion m ³)				Annual runoff (billion m ³)	Annual WPL	Seasonal runoff (billion m ³)	Seasonal WPL	TN (mg/L)
	Lhünzhub	Maizhokunggar	Annual total	Seasonal total					
2006	1.20	1.02	2.22	1.74	0.53	4.20	0.24	7.19	–
2007	1.26	0.76	2.02	1.52	1.62	1.25	0.46	3.31	–
2008	1.29	0.79	2.08	1.58	3.72	0.56	2.38	0.66	–
2009	1.40	1.01	2.41	1.89	0.84	2.86	0.46	4.13	–
2010	1.25	0.93	2.18	1.65	1.44	1.51	0.38	4.30	–
2011	1.16	0.86	2.01	1.48	1.51	1.34	1.04	1.42	–
2012	1.14	0.79	1.93	1.39	1.12	1.72	0.75	1.85	–
2013	1.14	1.00	2.14	1.58	1.20	1.78	0.68	2.33	–
2014	1.09	0.90	1.99	1.43	1.69	1.18	1.41	1.01	–
2015	1.11	0.87	1.98	1.41	0.42	4.69	0.48	2.96	–
2016	1.00	0.80	1.80	1.22	0.62	2.90	0.47	2.60	–
2017	1.06	0.62	1.68	1.09	0.61	2.78	0.46	2.38	1.08
2018	0.67	0.60	1.27	0.66	0.70	1.81	0.52	1.26	1.14
Average	1.14	0.84	1.98	1.43	1.23	2.20	0.75	2.72	1.11

Table 3 P-related GWFs and WPLs in irrigation areas

	Gray water footprint in two irrigation districts (billion m ³)				Annual runoff (billion m ³)	Annual WPL	Seasonal runoff (billion m ³)	Seasonal WPL	TP (mg/L)
	Lhünzhub	Maizhokunggar	Annual total	Seasonal total					
2006	0.11	0.08	0.19	0.11	0.53	0.36	0.24	0.44	–
2007	0.11	0.08	0.20	0.11	1.62	0.12	0.46	0.23	–
2008	0.11	0.10	0.21	0.12	3.72	0.06	2.38	0.05	–
2009	0.12	0.13	0.25	0.15	0.84	0.29	0.46	0.34	–
2010	0.10	0.12	0.23	0.13	1.44	0.16	0.38	0.34	–
2011	0.21	0.15	0.36	0.26	1.51	0.24	1.04	0.25	–
2012	0.21	0.15	0.36	0.26	1.12	0.32	0.75	0.35	–
2013	0.22	0.18	0.40	0.30	1.20	0.33	0.68	0.44	–
2014	0.22	0.17	0.39	0.29	1.69	0.23	1.41	0.20	–
2015	0.22	0.17	0.38	0.28	0.42	0.90	0.48	0.58	–
2016	0.22	0.16	0.38	0.28	0.62	0.61	0.47	0.60	–
2017	0.18	0.14	0.32	0.21	0.61	0.52	0.46	0.46	0.04
2018	0.18	0.15	0.33	0.22	0.70	0.47	0.52	0.42	0.02
Average	0.17	0.14	0.31	0.21	1.23	0.35	0.75	0.36	0.03

Table 4 Seasonal and annual WPLs in Chengguan District

	WPL for N		WPL for P	
	Seasonal WPL	Annual WPL	Seasonal WPL	Annual WPL
2006	0.16	0.19	0.09	0.06
2007	0.29	0.20	0.09	0.05
2008	0.09	0.11	0.04	0.03
2009	0.22	0.26	0.09	0.07
2010	0.16	0.16	0.09	0.05
2011	0.11	0.15	0.06	0.05
2012	0.16	0.19	0.07	0.06
2013	0.19	0.21	0.08	0.07
2014	0.17	0.17	0.06	0.05
2015	0.37	0.39	0.14	0.13
2016	0.21	0.23	0.08	0.08
2017	0.23	0.24	0.08	0.08
2018	0.23	0.24	0.09	0.09

some assumptions for simplifying calculations. Firstly, we only take the main categories of crops and live-stock into account. That means our results for N or P loads to freshwater may be slightly underestimated. Secondly, the emissions from point sources are based on statistical discharge coefficients and populations. The specified processes such as protein consumption and wastewater treatment coverage are not refined. Thirdly, when calculating the seasonal emissions from point source, we assume the seasonal values are based on the proportion of seasonal months to the whole year due to the lack of seasonal data. Besides, we regard the difference in runoff between two hydrological stations as the sum of river discharge from the Pengbo tributary and Mozhu tributary. Interactions between groundwater and surface water and evapotranspiration are ignored, which may also introduce uncertainties for WPL calculations.

Fig. 8 GWF trends for four sectors in Chengguanqu: **a** N-related GWF and **b** P-related GWF

Table 5 GWF proportions for the tourism industry

	GWF for N			GWF for P		
	Tourism industry (billion m ³)	Total (billion m ³)	Proportion (%)	Tourism industry (billion m ³)	Total (billion m ³)	Proportion (%)
2006	0.05	1.05	5.09	0.04	0.33	10.80
2007	0.09	1.50	5.76	0.06	0.35	16.23
2008	0.04	1.37	3.09	0.03	0.35	7.97
2009	0.10	1.54	6.54	0.07	0.40	16.56
2010	0.13	1.12	11.61	0.09	0.38	22.94
2011	0.16	1.17	13.77	0.11	0.41	26.35
2012	0.20	1.39	14.67	0.14	0.44	30.69
2013	0.25	1.50	16.78	0.17	0.49	33.64
2014	0.29	1.66	17.49	0.19	0.52	36.71
2015	0.38	1.80	21.00	0.25	0.59	42.54
2016	0.43	1.84	23.62	0.29	0.63	45.52
2017	0.48	1.94	24.68	0.32	0.67	47.74
2018	0.63	2.00	31.59	0.42	0.77	54.06

Conclusion

In this study, we estimated the nutrient loads delivered to freshwater and the associated GWFs in four sectors (i.e., agriculture, domestic life, tourism industry, and industry) to assess the seasonal and annual characteristics of the WPLs in a plateau basin from 2006 to 2018 and compared the WPL calculations with actual water quality measurements. The values of N-related GWFs in the basin were basically stable at 5 Gm³, while the P-related GWFs nearly doubled from 0.8 to 1.4 Gm³. More than 90% of the GWF came from anthropogenic sources. For the entire basin, domestic life was the largest contributor to both N-related GWFs (52%) and P-related GWFs (50%). The agriculture and the tourism industry have become the second largest contributor to N-related GWFs (32%) and P-related GWFs (30%), respectively. The share of industry was generally small. The N emissions into the freshwater environment exceeded the maximum assimilation capacity of the watershed in individual years (WPL > 1), and the values at the seasonal scale were more serious than those at the annual scale. P-related WPLs were far less than 1 at both seasonal and annual scales. The results of water quality measurements were generally consistent with WPL calculations. The serious N pollution was found near irrigation areas with average seasonal WPL value of 2.7 and average TN concentration of 1.11 mg/L. The prosperity of tourism has led to a 10-time increase in N-related GWFs and 5-time increase in P-related GWFs for the tourism industry near the Lhasa city. As a result, a strict top-down unified management suggestion for agricultural operation and tourism management was proposed for water pollution mitigation.

Author contribution DL: conceptualization, writing—original draft preparation; PT: writing—review and editing; DS: conceptualization, visualization; TH: funding acquisition, validation; HL: validation, data curation; BD: formal analysis, visualization; SK: conceptualization, validation; YC: supervision, validation; YL: conceptualization, supervision, methodology.

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Data availability The datasets used and/or analyzed during the 618 current studies are available from the corresponding author on reasonable request.

Declarations

Ethics approval and consent to participate Not applicable.

Consent for publication Not applicable.

Competing Queryinterests The authors declare no competing interests.

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