Quantifying the benefits of Nature-based Solutions in urban drainage on headwater stream water quality

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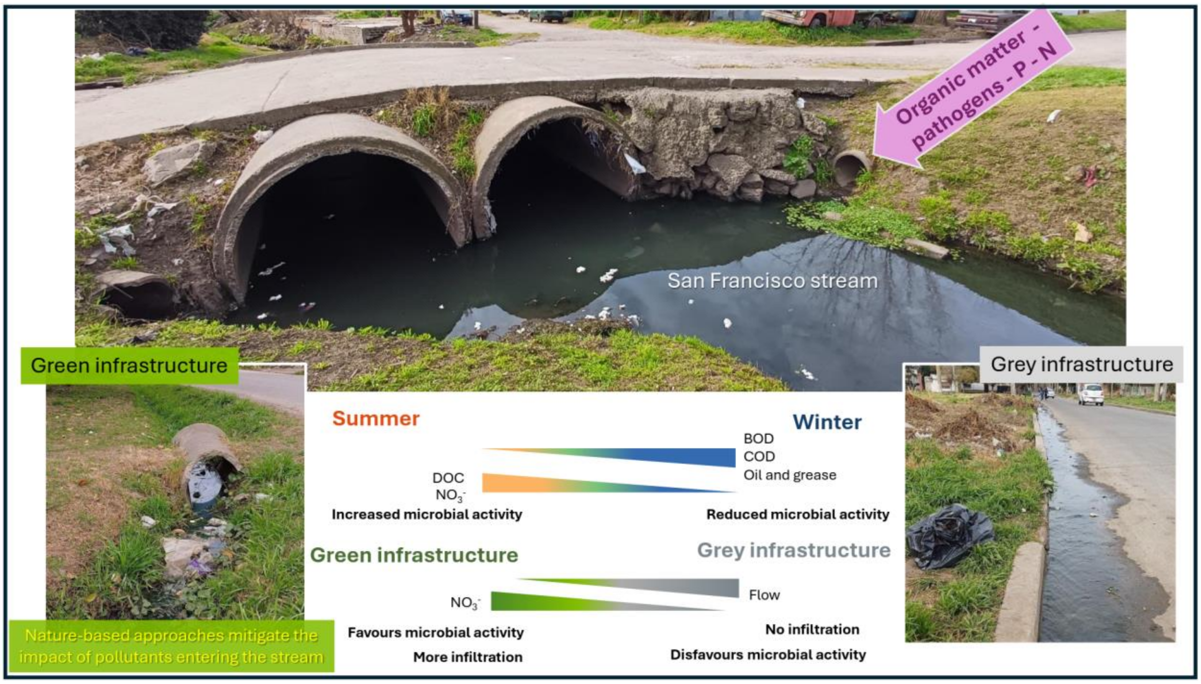
Highlights

- In areas with poor sanitation infrastructure, storm drains discharge domestic effluent into urban streams.
- Seasonality and type of infrastructure act independently and differentially on effluent composition and flow
- Concrete ditches and the winter season have the greatest impact on pollutant loads entering the stream
- Vegetated ditches reduce flow and pollutant load compared to grey infrastructure
- Vegetated drain ditches encourage infiltration and microbial activity

Graphical abstract



Keywords

- Nature-based Solutions (NbS)
- Urban water quality
- Green infrastructure
- Vegetated drainage ditches
- Pollutant load
- Urban stream syndrome

Abstract

In urban areas lacking adequate sanitation infrastructure, greywater flows into nearby urban streams through the stormwater drainage network. This study evaluates the impact of this drainage on water quality in a headwater stream in the Metropolitan Area of Buenos Aires (AMBA), Argentina. We analysed and model pollutant loads — organic matter, nutrients, and allochthonous faecal bacteria — entering the stream via the stormwater drainage network, comparing vegetated systems (green infrastructure) and concrete systems (grey infrastructure) across seasons. We observed the relative impact of domestic effluents on stream water quality, with significant input flows (3-16% across seasons) and high mass loads of Chemical Oxygen Demand (COD), Total Suspended Solids (TSS), Biochemical Oxygen Demand (BOD), Oil and Grease, nutrients, and allochthonous faecal bacteria, which, in some cases, double the concentrations of these substances in stream water over a one-kilometre reach. Linear models showed that both features, season and type of infrastructure, have mostly independent and differential effects on effluent quality. Grey infrastructure showed a significant increase in flow across seasons, together with a reduction in nitrate and total phosphorus levels. On the other hand, an increase in effluent BOD and COD levels was observed in winter, while DOC and nitrate concentrations increased in summer. Finally, bayesian models of effluent impact on stream water quality showed that during the most critical season (winter), the impact on the organic load of the stream is predominantly higher in a scenario with only grey infrastructure, with estimated increases between 1-96% for the different pollutants compared to green infrastructure. Overall, our results suggest that higher temperatures and the presence of green infrastructure, such as vegetated ditches, result in a lower impact of domestic effluents on stream water quality through mechanisms such as improved infiltration and enhanced biotic activity that contribute to metabolizing the organic load present in the effluent. This study highlights the ecological benefits of Nature-based Solutions (NbS) for urban water management, underscoring the potential of green infrastructure to improve water quality, support biodiversity, and strengthen the resilience of urban ecosystems.

Introduction

Urbanisation is one of the most impactful forms of land use and land cover because it has significant effects on the pattern, dynamics, and functionality of ecosystems, particularly aquatic ecosystems (Cerqueira et al., 2020; Elmqvist et al., 2013). Latin America has undergone a significant urbanisation process, with over 80% of its population residing in urban areas (UNDESA, 2019). In many cases, this rapid population growth has not been accompanied by essential sanitation infrastructure. The lack of basic services impacts not only the health and quality of life of affected populations, but also negatively affects the quality of water bodies that receive untreated or inadequately treated industrial and domestic effluents (Cirelli & Ojeda, 2008; Ramírez et al., 2008). Although the environmental consequences of inadequate urban planning are well known (Capps et al., 2016; da Cruz e Sousa & Ríos-Touma, 2017; Meyer et al., 2005; Ramírez et al., 2008; Walteros & Ramírez, 2020) as well as the quantitative impact of point discharges into urban streams (Bernal et al., 2020; Martí et al., 2004; Comber, Gardner & Ellor, 2020), understanding the spatial and temporal patterns of pollutant fluxes to urban streams, as well as the interactions between natural and engineered hydrological cycles, remains a critical challenge that requires further research. In particular, tracking pollutant sources is a primary need for managing pollutant loads in stormwater, as it will allow differentiation of pollutant types associated with specific urban land uses (McGrane 2016). How much pollution is caused by these discharges? What are the management implications? What are the alternatives for mitigation?

Latin America, which holds 33% of the world's water resources, faces significant challenges related to water availability, sanitation access, and environmental quality (Trimble et al., 2021). In Argentina, although it is a water-rich country, surface water resources are unevenly distributed, being concentrated in the Pampas and Mesopotamia regions (Cirelli & Ojeda, 2008). These regions include the Metropolitan Area of Buenos Aires (AMBA), the country's largest urban agglomeration, where 66% of the population resides (INDEC, 2011) and which encompasses some of the most degraded river basins (Basílico et al., 2022; Castañé et al., 2016; Graziano, Giorgi & Feijoo, 2021) The river systems in AMBA have been extensively altered through channelling and pipelining, leading to severe ecological simplification of key habitats (Rodrigues Capítulo et al., 2010; Graziano et al., 2019). Additionally, rapid and unplanned population growth has resulted in urban areas lacking adequate sanitation infrastructure, with untreated wastewater from over five million people being directly or indirectly discharged into rivers and streams (Cirelli & Ojeda, 2008; Öberg et al., 2014). Consequently, urban streams in AMBA exhibit poor water quality, with high levels of organic matter, nutrients, and microbiological contamination (Capítulo et al., 2001; López et al., 2013; Rigacci et al., 2013).

The grey infrastructure traditionally used for stormwater management has generated negative impacts by removing vegetation and altering hydrology. This has an impact on the natural structures of ecosystems, reducing their ecosystem services such as flood control, habitat supply and aesthetic benefits, accentuating the socio-ecological disconnection between cities and their natural environment (Dhakal & Chevalier, 2016). In contrast, approaches based on green infrastructure, or “Nature-Based Solutions” (NbS) promote ecosystem integrity and functionality, integrated into urban development (Mercado et al., 2024). An example is vegetated drainage ditches (VDD), which offer a viable approach to mitigating a wide range of contaminants, like constructed wetlands, intercepting and purifying runoff before it enters water bodies (Kumwimba et al., 2018). Although VDDs were initially used to treat agricultural runoffs, they are now successfully applied to manage domestic, aquacultural, and other types of wastewaters (Kumwimba et al., 2018). This type of green infrastructure addresses habitat and stream degradation by maintaining ecosystem integrity and connectivity while providing ecosystem services beyond stormwater management (Lovell & Taylor, 2013; Tzoulas et al., 2007).

In this context, the aim of this work is to analyse the inflow of domestic effluent through the stormwater drainage system into an urban headwater stream, and to assess the impact of the type of drainage infrastructure (green or grey) on it. We characterize the composition of urban effluents in the drainage network, quantify their impact on the stream's pollutant load, and examine how stormwater infrastructure type affects pollutant load and flow in the stream, along with inferring the ecological processes driving their seasonal variability. The results support the hypothesis about the benefits of green infrastructure to reduce the impact of urban drainage on water bodies, allowing the design of policies to protect urban water bodies within the framework of NbS approaches.

Material and methods

Area of study

The San Francisco stream belongs to the San Francisco-Las Piedras basin located within the Metropolitan Area of Buenos Aires (Argentina) and is categorized as a first-order lowland stream with a total length of 15 kilometres that flows to the Río de la Plata Estuary after its confluence with Las Piedras stream (Fig 1A). The creek is piped at its origins, in the peri urban area of the Almirante Brown District, emerging to surface at the locality of Claypole (Lat: 34°49'14"S; Long: 58°21'45"W). It has an estimated flow of 20–30 L sec^{-1} and a mean wetted width of 2–5 m (Efron et al., 2014). The stream exhibits clear eco-hydrological alterations and evidence of pollution along its course, with high levels of organic and microbiological contamination (Saraceno et al., 2021; Re et al. 2022; Efron et al., 2014). In turn, the city of Claypole presents socio-demographic characteristics of concern given the lack of sewerage coverage and drinking water supply (Graziano et al., 2019). Due to the lack of infrastructure, domestic wastewater is discharged through the stormwater drainage system. This network consists of two types of drainage systems: vegetated ditches running parallel to the road, which are found in unpaved streets, and paved streets where water is conveyed along the surface via the ditch until it reaches stormwater inlets (Fig. 1B). Both systems ultimately discharge into the stream via concrete pipes placed approximately every 100 meters. The study was conducted along a section of the San Francisco Stream, from where it emerges from the underground pipe system to 1.25 km downstream (Fig. 1C).

Based on these characteristics, two sub-basins were delineated using a Digital Elevation Model (DEM) of the area, obtained from the National Geographic Institute (IGN), and processed using QGIS (Imran et al., 2019) (Fig. 1C). One sub-basin (SB_1) includes the contributing area up to the point where the stream emerges, while the second sub-basin (SB_2) encompasses the surface runoff contributing to the study area. The delineation of these sub-basins was cross-referenced with data from the 2011 census (INDEC, 2011) to determine the population and number of households potentially discharging effluents into the stormwater system. Additionally, the stormwater network was classified as vegetated (green infrastructure) or cemented (grey infrastructure) (Fig. 1C). Since hybrid systems are often encountered, a drainage network was classified as green infrastructure (n=15) if at least the final 150 meters before discharging into the stream were vegetated, and as grey infrastructure (n=15) if this condition was not met.

Sampling and characterization of domestic effluents

Three seasonal samplings were conducted in winter (7/10/2019 and 07/18/2019), spring (10/21/2019 and 11/05/2019) and summer (02/11/2020 and 02/13/2020) between 11 am and 2 pm. The samplings included the midday hours to cover one of the peak times for household demand. On each sampling date, samples were collected from 5 discharge pipes spatially distributed along the study section. A total of 10 discharges were sampled per season. Time-integrated samples were taken in triplicate from each discharge, and one stream sample was also collected per season by triplicate. We considered that it had not rained for at least 48 hours prior to the measurements. The seasonally accumulated precipitation, recorded at the meteorological station (875760 SAEZ; Lat: 38° 48' 36"S, Long: 58° 31' 48"W) located 35 km from the study area, was as follows: 138.4 mm, 282.2 mm and 443.6 mm in winter, spring and summer, respectively. The water flow rate was measured at each intake by quantifying the volume of water discharged and the duration of time (Gore & Banning, 2017; Michaud & Wierenga, 2005). Streamflow was estimated by the velocity-area method, measuring the flow velocity in triplicate by the floating method (Gordon et al., 2004; Gore & Banning, 2017). For each sample, the following parameters were determined in situ: pH, temperature (T) and Electrical Conductivity (EC) using portable equipment (SensION 156

Hach), dissolved oxygen (DO) with an optical sensor (YSI PRO DO) and turbidity with a portable turbidity meter (2100P Hach).

Physicochemical samples were collected in polypropylene containers of 20 L. Once in the laboratory, it was fractionated and preserved in amber glass bottles (organic determinants) or plastic bottles (inorganic determinants). The characterization of the samples included: Total Suspended Solids (TSS), Biochemical Oxygen Demand (BOD), Chemical Oxygen Demand (COD), Dissolved Organic Carbon (DOC), Chlorides, Anionic Surfactants, Oil and Greases (determined as Soluble Substances in Ethyl Ether -SSEE-) and nutrients (Total Phosphorus -TP-, Soluble Reactive Phosphorus -SRP-, Ammonium, Total Kjeldahl Nitrogen -TKN-, and nitrates). TSS were determined by filtration onto a pre-weighed glass fibre filter paper (Whatman GF/C 1.2 μm pore) and determined gravimetrically after washing and drying at 105 ± 5 °C (APHA, 2018). Limits of quantification (LOQ) were 1 mg L^{-1}. BOD was determined within 48 hours of sample collection by the 5-Day BOD Test method (APHA, 2018), where sample dilutions were incubated for 5 days at 20 ± 3 °C (LOQ = 5 mg L^{-1}). COD was determined using the closed reflux procedure and the colorimetric method (APHA, 2018) (LOQ = 10 mg L^{-1}). DOC was filtered by 0.7 μm pore glass fibre (Microclar), preserved with sulfuric acid (Merck), and determined by the high-temperature combustion method using a Shimadzu 5000A carbon analyser (APHA, 2018) (LOQ 0.1 mg L^{-1}). In turn, chlorides and nutrients (except TKN and TP) were analysed within 48 hours of sample collection after filtration with a 0.7 μm glass fibre (Microclar). Chlorides were determined by the argentometric method (APHA, 2018) (LOQ = 4 mg L^{-1}), nitrates and nitrites by the cadmium reduction method (LOQ = 0.3 mg L^{-1}) (Company, 2007), ammonium by distillation (APHA, 2018) (LOQ = 0.1 mg L^{-1}), and SRP by the ascorbic acid method (APHA, 2018) (LOQ = 0.01 mg L^{-1}). TP was determined by the ascorbic acid method after acid digestion with nitric acid and sulfuric acid (Merck) (APHA, 2018) (LOQ = 0.01 mg L^{-1}). TKN was determined by distillation and colorimetry after Kjeldahl digestion (APHA, 2018) (LOQ = 0.1 mg L^{-1}). The “Kjeldahl nitrogen” is the sum of organic nitrogen and ammonia nitrogen, to calculate the Nitrogen organic (Norg) the difference between TKN and ammonium was made. Anionic surfactants were determined by the method of methylene blue active substances (MBAS) (APHA, 2018) (LOQ = 0.02 mg L^{-1}). SSEE were determined by gravimetry after extraction with sulfuric acid (Sintorgan) (APHA, 2018) (LOQ = 1.4 mg L^{-1}). Measurements were realized by colorimetric analysis with a spectrophotometer Shimadzu UV 2450.

For bacteriological analyses, sterile 1 L polypropylene containers were used to collect samples. Bacteriological analyses were conducted within a timeframe of 24 to 48 hours after sample collection. The abundance of Total Coliforms and *E. coli* was determined in each sample using the membrane filtration method. Serial dilutions in sterile ultrapure water were performed for each sample in triplicate, followed by filtration through a sterile nitrocellulose membrane with a pore size of 0.45 μm (Sartorius). Subsequently, the membranes were placed onto plates containing the selective medium Chromocult® Coliform Agar (MilliporeSigma). Incubation of the plates was carried out for 24 hours at 37°C in a dark environment (Alonso et al., 1998; Manafi & Kneifel, 1989). Colony counts were recorded at the optimal dilution, and an average count was calculated per sample. The limits of quantification were set at 1 CFU ml^{-1} for both determinations.

Data analysis

First, to compare the levels of contaminants present in the sampled discharges, we conducted a comparative analysis of the observed discharge concentrations against those permitted by the Argentine regulations (ACUMAR, 2019). Subsequently, a linear discriminant analysis (LDA) was conducted to identify the variables that allow the differentiation of effluents by season. Multicollinearity was prevented by removing variables with a Pearson correlation coefficient (ρ) greater than 0.90. The following variables, previously standardized, were utilized for the LDA: Norg, DOC, *E. coli*, nitrates, chlorides, TP, MBAS, and COD.

In order to analyse the effect of *season* and *type of infrastructure*, generalized linear models (GLMs) and generalized linear mixed models (GLMMs) were employed, with *season* and *type of infrastructure* as explanatory factors, incorporating the identity of the discharge as a random effect. The distribution of the response variable was examined to select the appropriate family and link function for each model. In cases where normality or homoscedasticity assumptions were violated, alternative distributions such as Gamma (log link) or Tweedie, suitable for continuous positive data with non-constant variance, were applied. Normality and homogeneity of variance assumptions were evaluated using diagnostic plots and statistical tests (Shapiro-Wilk). When assumptions were not met, alternative models were fitted, or heterogeneous variance was explicitly modelled. Significant differences in main effects and interactions were analysed through ANOVA, while multiple comparisons between factor levels were performed using *post hoc* tests (Tukey HSD).

Furthermore, a Bayesian model (warmup = 1000, chains = 3, iter = 5000, thin = 3) was employed to develop a theoretical flow distribution by season and by type of infrastructure, separately for each factor. The empirically determined flow rates were employed as the input for the model. To ascertain the prior distribution, the empirical flow data were fitted to three potential distributions: lognormal, normal, and gamma. The lognormal distribution exhibited the most optimal fit and was therefore selected. The model was employed to generate the probability distribution curve of domestic discharge flow. A total of 40 random values were selected from the theoretical probability distribution of discharge flow rates for each analysed condition. This allowed for the estimation of the theoretical cumulative flow of discharges entering the stream after 1 km, under the assumption of four active discharges every 100 meters. The mean theoretical mass loads of each contaminant were calculated after 1 km. To assess the influence of discharges on the stream water quality discriminated by season, the proportion of each contaminant load in the stream flow was estimated. Furthermore, the differential contribution between grey and green infrastructure was evaluated through the simulation of two contrasting scenarios: one in which all discharges originated from grey infrastructure and another in which all originated from green infrastructure. The comparison between these scenarios enabled the relative contribution of each type of infrastructure to the stream's water quality to be quantified.

Statistical analyses were carried out using the R packages MASS (Venables & Ripley, 2002), fitdistrplus (Delignette-Muller & Dutang, 2015), lme4 (Douglas et al. 2015), gmmlTMB (Mollie et al. 2017), emmeans (Searle et al. 1980), DHARMa (Hartig, 2022) and brms (Bürkner, 2021) in R (version 4.3.1) (R Core Team, 2023).

Results

Characterization of effluents from domestic discharges

To assess the organic load, nutrients and bacterial load entering the San Francisco stream through the storm drainage network, and their impact on the receiving water body, 10 time-integrated samples were collected per season from discharges along a 1.25 km stretch of the stream, a drainage area with 6,938 households and 27,115 inhabitants (SB_2). In addition, a stream sample was collected upstream of the discharge points, covering the drainage of 11,777 households and a total of 41,058 inhabitants (SB_1). Spatial and seasonal variability was evident in the flow rates of the discharges, as well as in the physicochemical and bacteriological parameters analysed (Fig. 2 and Table 1).

Table 1. Characterization of urban effluents. The Mean±SD for each parameter evaluated per season (n=10) are reported.

	Flow	*E.coli*	Total coliforms	Turbidity	Norg	Nitrates	Ammonium	SRP	TP	TSS	MBAS	SSEE	BOD	COD	DOC	Chlorides	pH	T	EC	DO
	ml seg^{-1}	CFU ml^{-1}	CFU ml^{-1}	NTU	$mgNL^{-1}$	mg $N\text{-}NO_3^- L^{-1}$	mg $N\text{-}NH_4^+ L^{-1}$	mg $P\text{-}PO_4^{3-} L^{-1}$	mg $P\text{-}PO_4^{3-} L^{-1}$	mgL^{-1}	mgL^{-1}	mgL^{-1}	mgL^{-1}	mgL^{-1}	mgL^{-1}	mgL^{-1}	units pH	º C	μScm^{-1}	mgL^{-1}
Winter	97.2±174	9.2E+03±6.3E+03	1.5E+05±8.5E+04	116±48	6.37±2.85	1.4±0.5	14.5±1.9	1.16±0.65	2.11±0.96	76.5±61.8	6.07±1.97	42.2±52.7	167±155	434±333	56.8±17.0	127±12	7.96±0.27	14.3±1.5	1595±180	3.95±1.18
Spring	138±224	6.1E+03±5.5E+03	2.6E+05±2.7E+05	64±44	14.7±21.7	2.4±0.5	10.9±2.0	2.86±5.07	3.86±5.80	60±49	5.74±2.29	34.5±17.3	102±120	282±217	54.3±20.1	140±42	7.96±0.14	26.5±3.03	1690±404	2.28±2.48
Summer	50±78	6.1E+03±4.4E03	1.5E+05±1.7E+05	51±30	5.40±3.29	4.7±2.2	9.8±2.2	1.30±1.05	1.70±1.10	41.1±31.2	3.42±2.04	19.5±18.3	43±31	192±96	95.4±27.8	141±40	8.30±0.12	28.1±1.6	1618±301	4.01±1.88

Effluent concentrations exceeded the limits established by Argentine regulations (Res. ACUMAR, 2019) for discharge into stormwater systems (Fig. 3a). Several parameters, primarily related to suspended organic matter, consistently exceeded regulatory limits across all sampling periods (BOD, COD, MBAS, and *E. coli* as a proxy for faecal coliforms), indicating the high contribution of domestic effluents into the stormwater network. Other parameters, including total nitrogen and dissolved ammonium, exceeded limits only during spring.

A linear discriminant analysis (LDA) was conducted to determine seasonal differences in effluent composition (Fig. 3b). Group means for the parameters indicated that spring was generally characterized by intermediate values for most parameters, such as nitrates, TP, and MBAS. During summer, the highest DOC values were observed, along with the lowest MBAS and TP values, whereas winter showed elevated levels of DOC and *E. coli*. In turn, the coefficients of the two linear discriminant axes revealed the most influential variables in the differentiation of the seasonal groups. The first discriminant axis (LD1) accounted for 89.1% of the variability, while the second discriminant axis (LD2) explained the remaining 10.9%. The variables contributing most to LD1 were Norg, DOC, and TP, indicating their critical role in seasonal discrimination. In LD2, *E. coli* and DOC concentrations also had a significant influence, albeit to a lesser extent. The predominance of LD1 in capturing variability suggests that seasonal differentiation is primarily driven by nutrient and organic matter concentrations. The classification error was 16.67%.

Influence of seasonality and type of infrastructure on pollutant dynamics.

An ANOVA analysis revealed that different variables have a differential response according to season or type of infrastructure (Fig. 4). Both flow and TP varied significantly as a function of infrastructure type ($p<0.05$), with higher flow rates observed in cemented ditches compared to vegetated ditches. This suggests a higher infiltration capacity in vegetated ditches. On the contrary, TP concentrations were higher in ditches with green infrastructure.

In the case of nitrate, it was influenced independently by both the type of infrastructure and seasonality, with significantly higher concentrations in the warm seasons (spring and summer), and in the presence of the vegetated ditches ($p<0.05$). This suggests a higher activity of nitrifying organisms during these seasons, also favoured by the green infrastructure.

On the other hand, organic matter-related variables responded differentially to seasonality. BOD was significantly lower as temperatures increased (summer < spring < winter) ($p<0.05$). Similarly, COD concentrations were higher in winter. SSEE also followed a seasonal pattern, with significantly lower concentrations in summer. In contrast, the highest

DOC concentrations were observed in summer, suggesting processes of decomposition of organic matter into smaller or soluble fractions.

In contrast, the levels of *E. coli*, total coliforms, chloride, MBAS, TSS, SRP, Norg, TKN and ammonium showed no significant influence of season or infrastructure type.

Impact of effluents on stream water quality: Benefits of Green over Grey Infrastructure

Applying a Bayesian model, we estimated the distribution of discharges along a one-kilometre stretch of the San Francisco stream, assuming the presence of four active discharges per 100 meters of the stream (Fig. 5). This approach was relevant for assessing the relative contribution of domestic discharges to the total stream flow. Analysing the contribution by season, the highest contributions were observed in spring and winter, representing 16% and 9% of the total stream flow, respectively, while in summer this proportion decreased to 3% (Fig 5a). By integrating the modelled flow with pollutant data obtained during sampling, the increase in the mass load of different pollutants compared to the levels originally transported by the stream was estimated. The results indicate that the most critical conditions occur in winter (Fig. 5b), with 60% increases in the mass load of BOD and MBAS after traveling 1 km. For *E. coli*, total coliforms, DOC, and chlorides, increases ranged from 20% to 50%, while for nutrients an increase between 10% and 25% was estimated. In addition, TSS and SSEE doubled their downstream load. In spring, the increases were reduced to the 6%-50% range, while in summer they were even smaller, between 3% and 15%.

When comparing the theoretical flow distributions according to infrastructure type (Fig. 5c), flow rates were estimated to be three times higher for grey infrastructure compared to green infrastructure (2.7 L seg^{-1} km^{-1} for grey infrastructure and 0.9 L seg^{-1} km^{-1} for green infrastructure). Assuming two extreme scenarios-with all discharges being grey or all green-the results show considerable differences in pollutant contribution (Fig. 5d). During the most critical season (winter), total coliforms would double if all discharges were grey, anionic detergents would increase by more than 50%, and TSS would increase by almost 50%.

Discussion

Our study highlights that in neighbourhoods lacking adequate sanitation infrastructure, the stormwater drainage network becomes a point source of pollution, introducing domestic effluents with high loads of organic matter, nitrogen, phosphorus, and allochthonous faecal bacteria. We found that this situation is aggravated in winter, when effluent flows are higher, and lower temperatures reduce microbial activity. In turn, when comparing grey infrastructure with green infrastructure, we identified that green infrastructure attenuates the impact of effluents on stream water quality by promoting infiltration processes and favouring microbial activity. The influence of type of infrastructure and season is mainly independent of each other. These findings underscore the relevance of implementing NbS in urban settings to mitigate pollution and strengthen ecological resilience, suggesting that even in other contexts with better infrastructure these actions can help reduce pollutants entering urban water bodies.

Characterization of urban effluents in informal settlements

In our study, urban effluents presented a significant pollutant load, exceeding the limits established by Argentine regulations. Although the physicochemical composition of these effluents varied greatly, the analysed discharges showed patterns similar in composition and quantity to greywater (kitchen, bathroom, laundry) (Eriksson et al., 2002), and given the levels of *E. coli* found, possibly combined with blackwater. It is known that in contexts of social vulnerability, there may be situations of greywater/blackwater mixing, or even contamination of the water source used with high levels of *E. coli* (Di Pace et al., 2012). It should be also noted that the residents of the neighbourhood, which doesn't have a drinking water supply, are forced to meet their needs through water withdrawals from polluted aquifers (Merlinsky et al., 2012; Tobías, 2015).

Regarding the levels found for the main components of the effluents, we found a higher load of total nitrogen (min-max: 6 - 214 $mgNL^{-1}$) than reported by other authors in greywater (Eriksson et al., 2002; Henze et al., 1997) We consider that these levels could be related with blackwater/greywater mixing, adding additional sources of N such as urine (Eriksson et al., 2002; Jefferson et al., 2004). On the contrary, expected values for total phosphorus (0.34-19.1 $mgP\text{-}PO_4^{3-}L^{-1}$) were found, compared with other studies (Eriksson et al., 2002; Eriksson et al.2009; Li et al., 2009), where P containing detergents are one of the main sources of this pollutant. Finally, the COD: BOD ratio was 2.6, in the winter; 2.8 in the spring, and 4.5 in the summer, indicating a higher proportion of non-biodegradable organic matter with respect to wastewater (Jefferson et al., 2004; Metcalf et al., 2004).

Impact of Green versus Grey Infrastructure and season on Stream Water Quality

The experimental design allowed us to discriminate the seasonal influence from that of the type of infrastructure present, finding mostly independent, though comparable, effects of both factors. We develop a conceptual model synthesizing such effects and the possible mechanisms that support it (Fig. 6). Our study demonstrated that in a hypothetical scenario with 100% vegetated ditches, the impact of pollutants would decrease significantly.

Concrete ditches, designed to maximize the transport and speed of stormwater, do not prioritize ecological functions or peak flood attenuation (Shen et al., 2021). In our study, the impact of grey ditches on water quality tends to be greater, in a similar fashion compared with the influence of cold season. We suggest that both factors impact urban effluents by reducing their microbial degradation rates and the capacity of water infiltration or evapo/transpiration, thus increasing flow and reducing the capacity of nutrient removal (Zhao et al., 2019).

In particular, if we focus in terms of nitrogen removal, numerous studies have indicated that nitrification/denitrification is the dominant nitrogen removal process in vegetated ditches (Kumwimba et al., 2018; Soana et al., 2017; Zhang et al., 2020). Temperature affects plant growth and microbial degradation rates (Zhao et al., 2019). Specifically, it influences nitrification/denitrification rates, as nitrifying and denitrifying microorganisms have been observed to be inhibited below 10 °C (Cookson et al., 2002; Hwang and Oleszkiewicz, 2007). In our study, higher nitrate concentrations were observed during the warm seasons, which may be attributed to nitrification of remineralized NH_4^+. On the other hand, vegetated ditches showed higher nitrate concentrations in all seasons. In this regard, it is important to note that plant-mediated nitrification is a key source of NO_3^- for denitrifiers in saturated soils where NH_4^+ is the dominant form of inorganic N (Tatariw et al., 2021).

Using a Bayesian model and the theoretical flow distribution in each season, we estimated the mass loads entering the stream through the stormwater network over a 1 km stretch, representing between 3% and 16% of the total load depending on the season. The impact of these effluents on the San Francisco stream was especially notable in winter, when low flows reduced dilution capacity, exacerbating pollutant concentrations. This phenomenon aligns with observations in other urban water bodies, where the combination of low dilution capacity and high pollutant loads leads to significant water quality degradation (Moeder et al., 2017). Moreover, when analysing flows according to the type of infrastructure, it was observed that vegetated ditches, by promoting infiltration processes and enhancing microbial activity, reduce the contaminant loads entering the stream, thereby resulting in a lesser impact on its water quality.

Vegetated Ditches as a Nature-Based Solution to Mitigate Pollution in Urban Streams

Vegetated drainage ditches (VDDs) have been widely studied as effective NbS systems to control diffuse pollution in agricultural settings (Kumwimba et al., 2018; Rizzo et al., 2023). However, although stormwater drainage systems are common elements in the landscape, few studies have quantified their biogeochemical potential (Tatariw et al., 2021).

In our work, vegetated ditches were observed to provide superior benefits compared to concrete ditches, especially in contexts of social vulnerability and lack of sanitation infrastructure, where they acquire even greater relevance.

Shen et al. (2021) compiled several findings showing that vegetated ditches have better nutrient removal rates compared to non-vegetated ditches and that concrete ditches exhibit significantly lower removal rates than vegetated ones. Vegetation plays a fundamental role in enhancing nutrient removal compared to non-vegetated ditches (Kumwimba et al., 2017; Shen et al., 2021). These ditches function as an integrated plant-sediment-microbe system, which is crucial for nutrient uptake, sediment interception, and microbial decomposition (Zhang et al., 2020). Additionally, vegetation directly or indirectly influences nitrification and denitrification processes in the rhizosphere (Tang et al., 2020). Studies have evaluated various emergent macrophyte species, such as *Myriophyllum, Typha, Canna, Hydrocotyle, Pontederia, and Phragmites* (Zhang et al., 2016; Kumwimba and Zhu, 2017; Vymazal and Brezinová, 2018; Moore et al., 2020; Zhang et al., 2020). In our study, the ditches were mostly vegetated with grasses, and in some cases, with macrophytes of the genus *Hydrocotyle*, while some ditches lacked vegetation. Evaluating different species and selecting those native to the region with greater removal capacity could be a future line of research.

While vegetated ditches may be perceived as a "precarious" solution from a conventional perspective, they hold great potential. This potential is especially relevant considering that many cities in the Global North are adopting more natural, less concrete-dependent systems, whereas in the Global South, the idea of progress remains linked to large-scale, cement-based infrastructure. An integrated approach could reduce construction and investment costs, allowing resources to be reallocated to critical needs such as sewage systems and drinking water networks. Improving the current infrastructure and increasing selected vegetation could bring numerous benefits, as documented in the literature. This approach is essential, especially in neighbourhoods where access to treatment systems is a priority, and stormwater runoffs carry high contamination loads.

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Figures

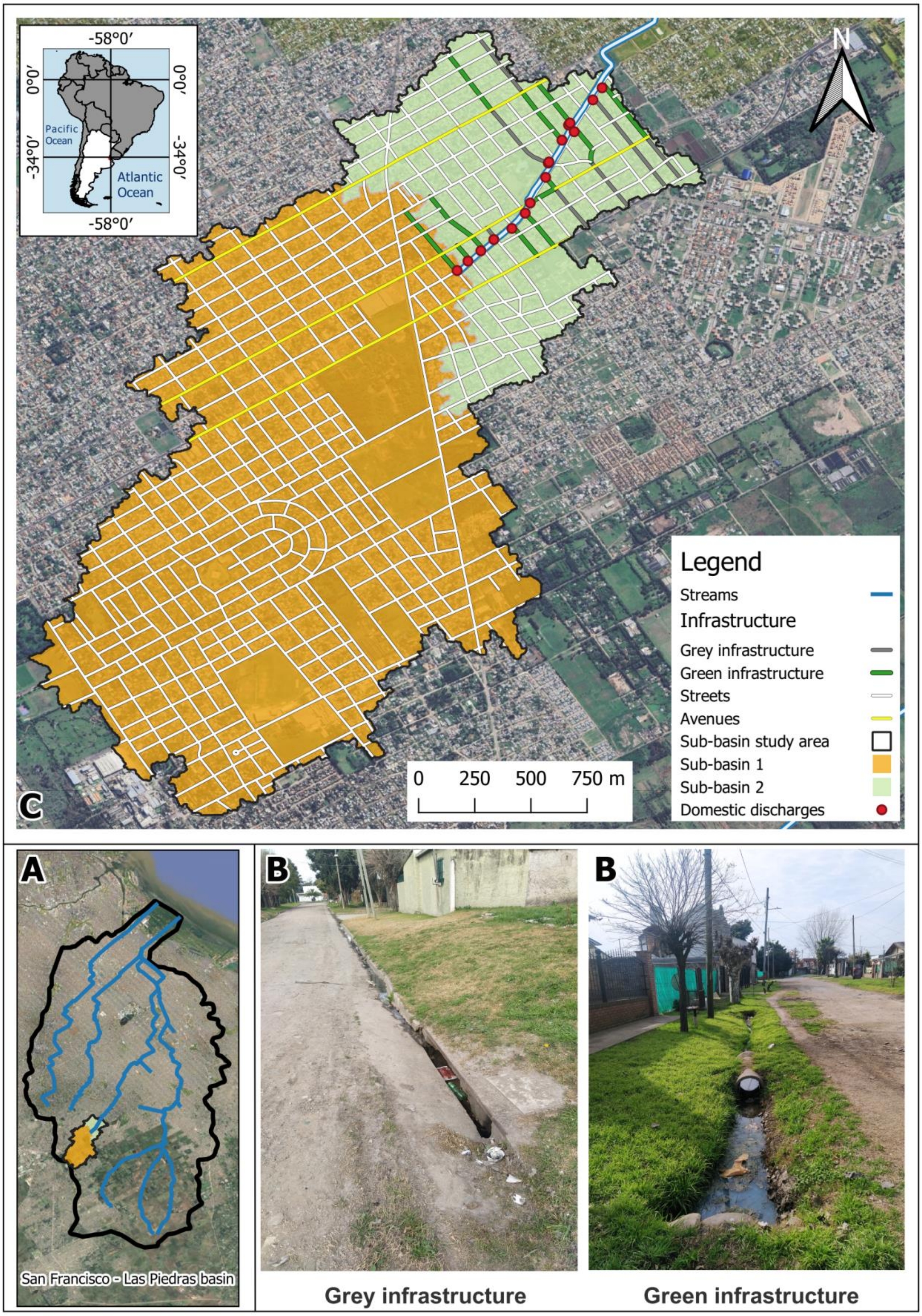


Figure 1. Study area. A) San Francisco-Las Piedras watershed, located in the Metropolitan Area of Buenos Aires (Argentina). B) Characterization of the storm drainage network in the

neighbourhood: vegetated ditches parallel to the road, present in unpaved streets, and paved streets where water flows superficially through the ditches to storm drains. C) Watershed delineation of San Francisco Creek in the study area: sub-basin 1 (SB1) includes the piped contributing area, while sub-basin 2 (SB2) encompasses the study area, from where the creek emerges from the underground pipe system up to 1.25 km downstream.

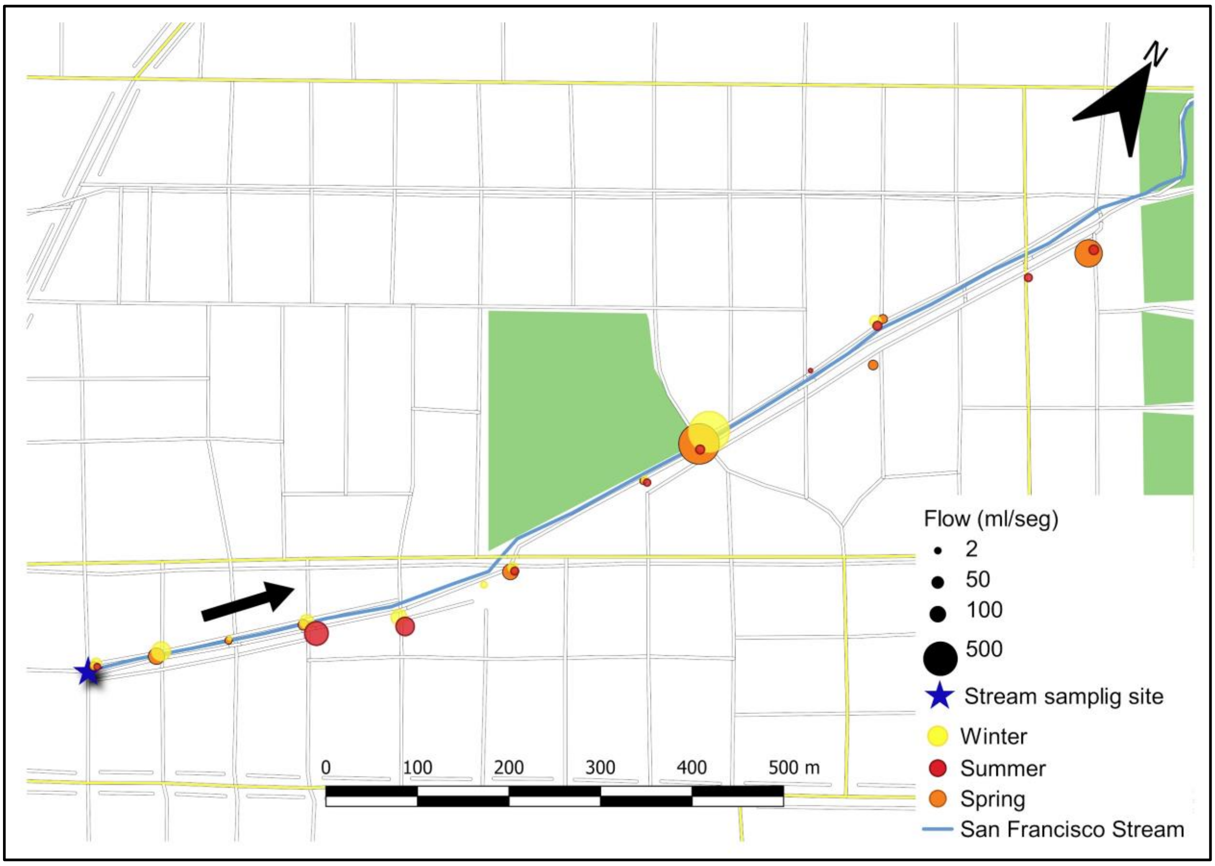


Figure 2. Map with the area of study. The blue star indicates the stream sampling site. The black circles indicate the analysed domestic discharges where the size of the circle is proportional to the measured flow (red= winter, yellow= summer, orange= spring). The arrow indicates the direction in which the stream water flows.

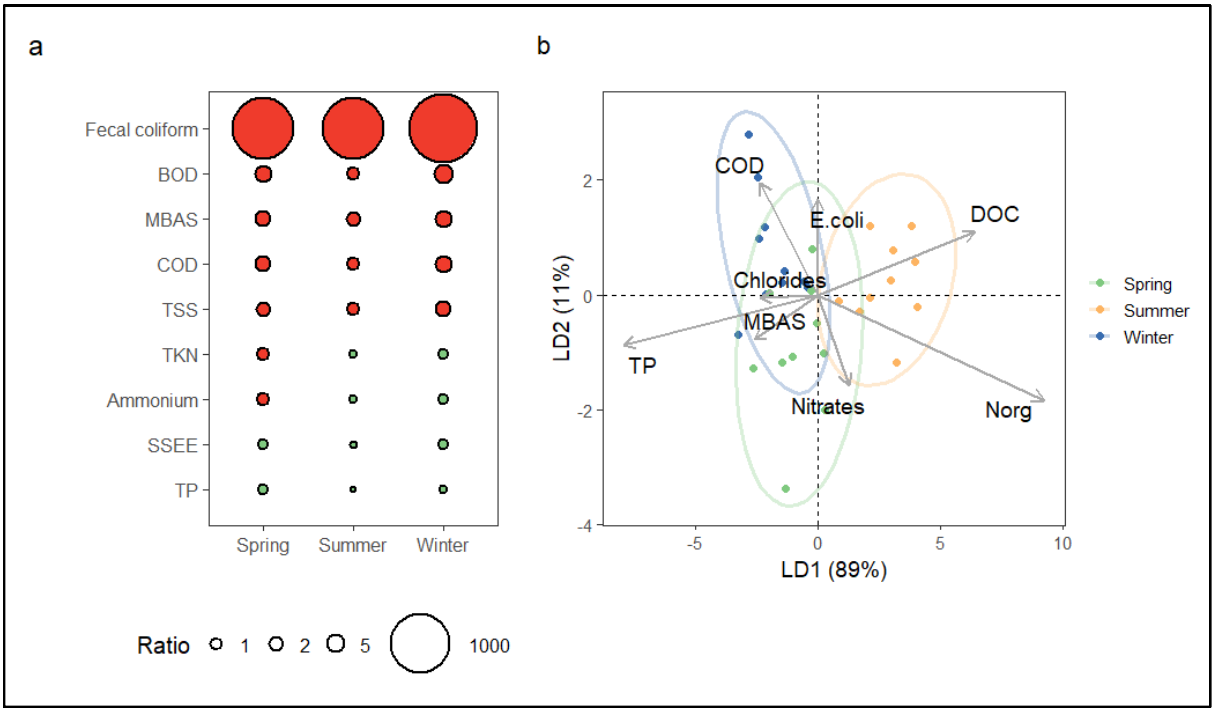


Figure 3. Characterization of urban effluents. a) Values of discharge allowed to a pluvial network with respect to Argentine regulations (ACUMAR, 2019). The size of the circle represents how much the parameter is exceeded, and the colour whether it complies with the regulation (green) or not (red). b) Linear Discriminant Analysis (LDA) according to the composition of the effluent with respect to season.

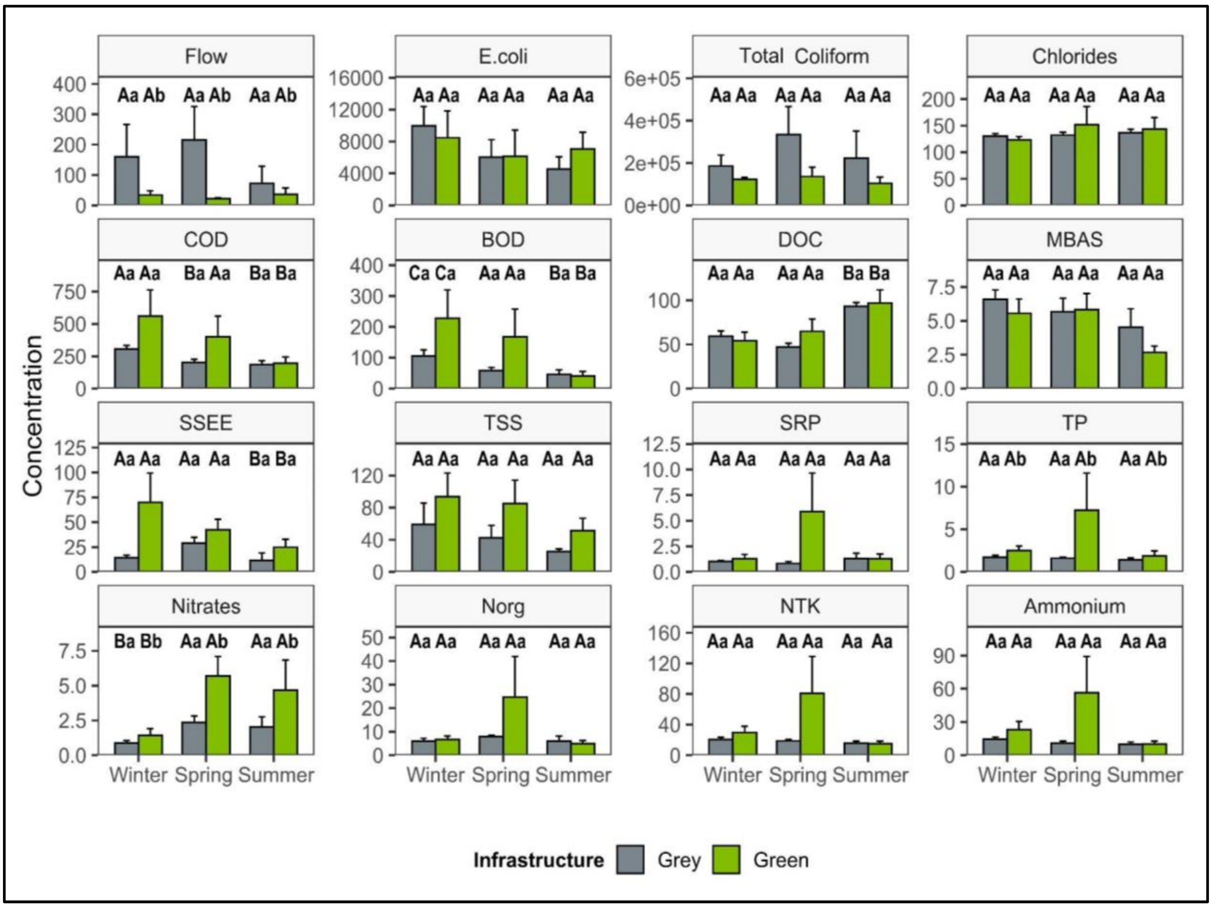
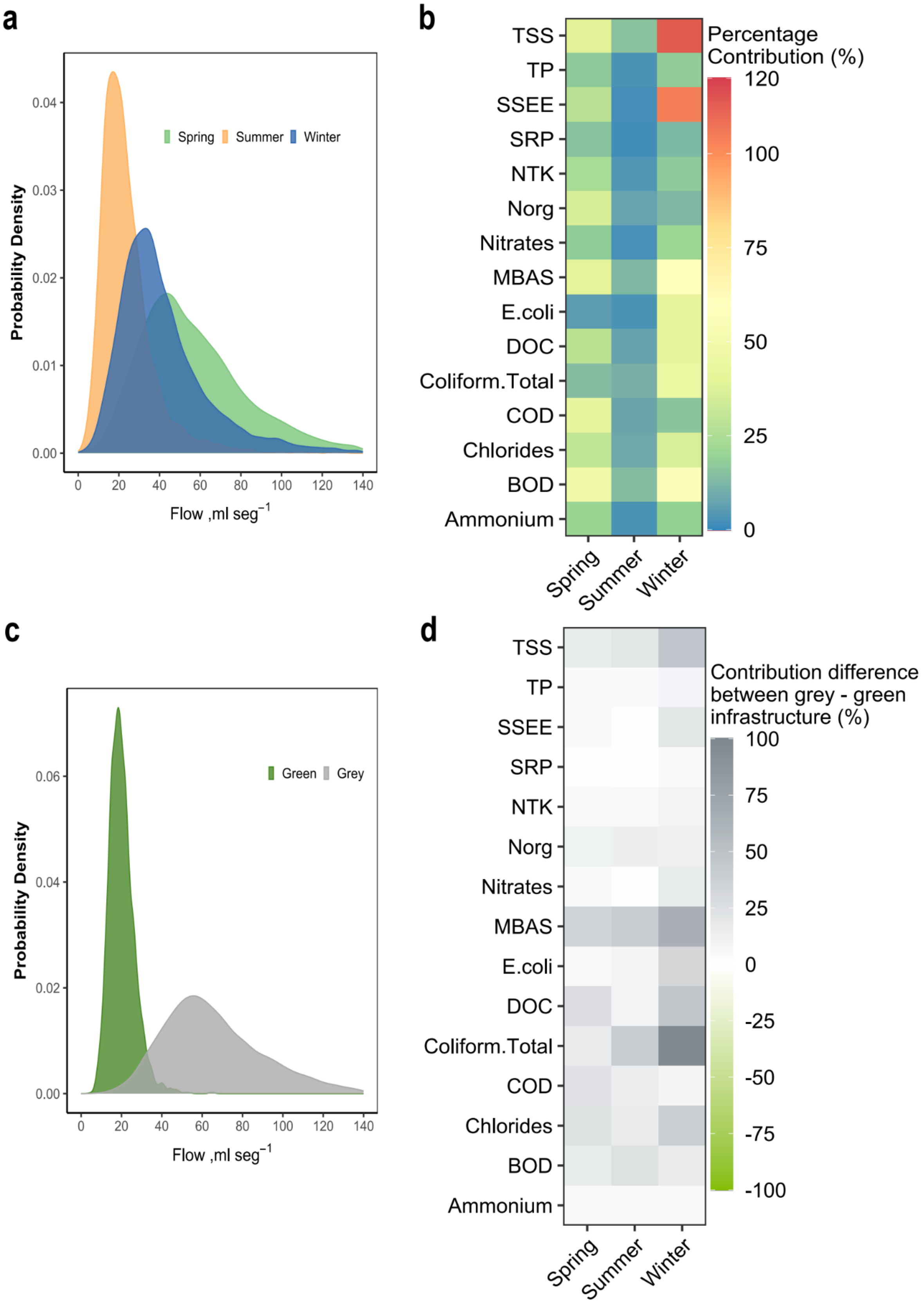


Figure 4. ANOVA analysis comparing the flow and pollutant concentrations by season and infrastructure type (grey vs. green). Different capital letters indicate post-hoc significant differences ($p < 0.05$) between stations. Different lower-case letters indicate significant differences ($p < 0.05$) between infrastructure types. Flow is expressed as mL seg^{-1}, and *E. coli* and total coliforms are expressed as CFU/ml.



a
Probability Density
0.04
0.03
0.02
0.01
0.00
Spring
Summer
Winter
0
20
40
60
80
100
120
140
Flow ,ml seg^{-1}
b
TSS
TP
SSEE
SRP
NTK
Norg
Nitrates
MBAS
E.coli
DOC
Coliform.Total
COD
Chlorides
BOD
Ammonium
Spring
Summer
Winter
Percentage Contribution (%)
120
100
75
50
25
0
c
Probability Density
0.06
0.04
0.02
0.00
Green
Grey
0
20
40
60
80
100
120
140
Flow ,ml seg^{-1}
d
TSS
TP
SSEE
SRP
NTK
Norg
Nitrates
MBAS
E.coli
DOC
Coliform.Total
COD
Chlorides
BOD
Ammonium
Spring
Summer
Winter
Contribution difference between grey - green infrastructure (%)
100
75
50
25
0
-25
-50
-75
-100

Figure 5. Modelling of urban discharge flow and pollutant mass load estimates along the stormwater drainage network. (a) Probability distribution of flows by season from Bayesian modelling. (b) Estimated contribution of domestic discharges relative to stream flow over a one-km reach by season. (c) Probability distribution of flows by infrastructure type from Bayesian modelling. (d) Percent differential contribution between grey and green infrastructure.

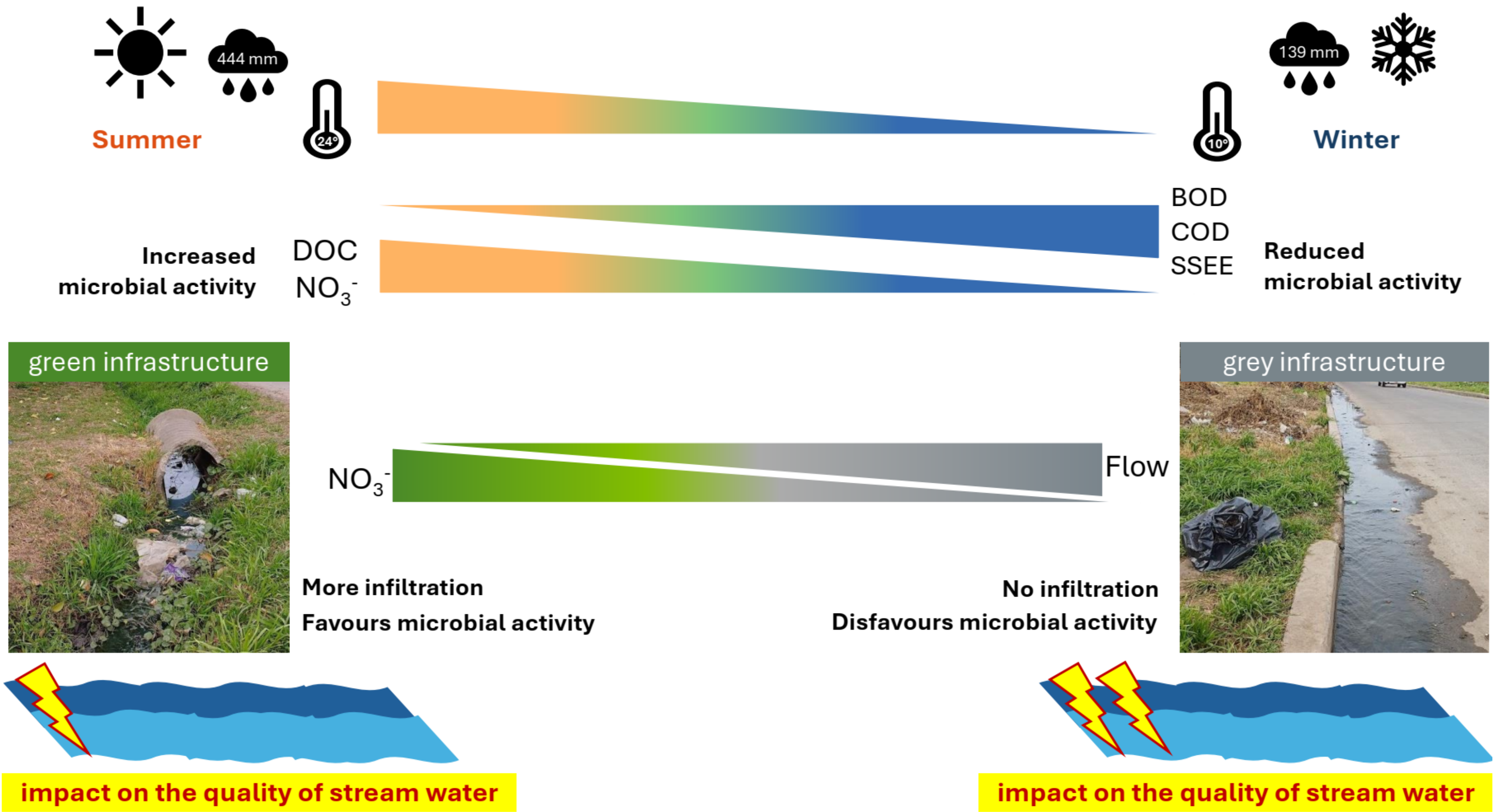


Figure 6: Conceptual model illustrating pollutant dynamics by season and infrastructure type (green - grey), and the impact (lightning bolt) on stream water quality.