Water Air Soil Pollut (2019) 230: 262
https://doi.org/10.1007/s11270-019-4314-7

Assessment of Pollution Sources, Fate of Pollutants, and Potential Instream Interventions to Mitigate Pollution of Earthen Canals of Urban to Rural-Urban Fringe

Pattiyage I. A. Gomes ·
Bothalage A. V. W. Fernando · **Ganegeoda K. Dehini**

Received: 8 July 2019 / Accepted: 16 October 2019 / Published online: 9 November 2019

Abstract Three representative earthen canals from urban, peri-urban, and rural-urban fringe of Sri Lanka were studied for a 2-year period against different seasons to capture insights important in ecological rehabilitation. Only the canal from rural-urban fringe showed a better water quality in wet season; elucidating, the impact of contaminated catchment runoff in the other canals. At a given sampling session, one or two peaks (relative maxima) were observed in urban and peri-urban canals for pollution representative parameters such as nitrate nitrogen and soluble reactive phosphorus. Those peaks were highly localised, an indication of poor advection. In general, two-dimensional variations of electrical conductivity and turbidity in dry season were uniform in urban and peri-urban canals, an indication of dominant molecular diffusion. This was further evidenced via physical models for different flow stages (low, high, and bankfull). Therefore, fate of contaminants had to be mainly governed by assimilation via sediments. However, grey water footprint analyses showed urban and peri-urban canals have over utilised the natural assimilation capacity of many water quality parameters by several folds. This study proved the importance of inducing attenuation by instream physical heterogeneity similar to natural streams or naturalised canals such as the canal from the rural-urban fringe of this study.

Keywords Advection · Assimilation · Grey water footprint · Molecular diffusion · Spatiotemporal water quality variations · Urban canals

P. I. A. Gomes (✉) · B. A. V. W. Fernando · G. K. Dehini
Department of Civil Engineering, Sri Lanka Institute of Information Technology, New Kandy Road, Malabe 10115, Sri Lanka
e-mail: ishan_gomes@yahoo.com
e-mail: ayantha.g@sliit.lk

B. A. V. W. Fernando
Department of Civil Engineering, Curtin University, Bentley, WA 6102, Australia

1 Introduction

Man-made lotic waters such as canals are built mainly for irrigation, transportation, and flood control. In many urban areas, the contemporary use of canals (even natural waterways) is flood control. This is why most of the urban waterways are straight with homogenous cross sections (Kim and Jang 2019). Man-made implies canals had to be considered as a disturbance to the natural environment. However, people would consider canals to be part of the natural environment with time, and value them same as natural streams and rivers (Nakamura et al. 2006; Gomes et al. 2016). Furthermore, lotic waters (whether man-made or natural) provide, perhaps, the only recreational space in urban areas (Nakamura et al. 2006). Even though the demand for naturalised lotic waters in developing countries is not given equal status similar to other socio-economic factors such as in developed countries, some developing countries have taken initiations to attend such goals (e.g. Megapolis plan of Sri Lanka 2018).

The rapid development in urban catchments results in higher sediment yield (Kiat et al. 2008). The fate of contaminants carried by suspended and dissolved sediments greatly depends on the assimilation and attenuation potential of the stream, whereas assimilation and attenuation mainly depend on the transport and mixing signature of the stream. No doubt the contaminant load to lotic waters needs to be curtailed, same time the capacity of the stream/river to transform contaminant to less or no harmful state should be enhanced. Aeration-induced oxidation of dissolved hydrogen sulfide to sulfite, thiosulfate and sulfate ions (Zhang and Millero 1993) is one such example. In addition, transportation of the contaminant to a less sensitive ecosystem with high carrying capacity is important too.

Transport and mixing in lotic waters depend on several hydraulic factors such as three-dimensional distribution of velocity, wetted depth and width, channel slope, and channel bed properties. Usually transport and mixing studies are based on field experiments, supported by physical models and numerical simulations. Numerical simulations are usually conducted for extreme flow scenarios such as low frequency but high-intensity flood events (e.g. a 100-year return period flood). One major problem of transport and mixing studies of lotic waters is difficulties associated in field experiments in terms of large labour, time, and cost. One strategy in this regard is incorporation of easy to measure, low cost, and less time-consuming variables representative of the experimental scenarios, instead of the conventional hydraulic variables. As examples, turbidity and electrical conductivity (EC) observations instead of total suspended (TSS) and total dissolved solids (TDS), respectively, can be given.

Canals in general straight and have homogeneous cross sections. Such characteristics could be observed in regulated streams also (e.g. straightened and/or concrete-lined streams in metropolises such as Hong Kong and Singapore). In many parts of the world, canals are constructed to convey flood waters, as such they have large flow rates, and flows are supercritical or near supercritical in nature. However, the canals in Sri Lankan urban areas are characterised by slow-moving and subcritical flows. This is particularly the case of the characteristic Dutch canal system (built about 300 years back during Dutch colonisation) within the Colombo Municipality. Also, a few recently built canals (e.g. Talangama canal built in 1995) too have such flow conditions. Slow-moving flow conditions in Sri Lankan canals are mostly because of the flat terrain and the canal elevation relative to the mean sea level. If these canals are made steep, the downstream will be a few meters below the mean sea level and would aid more saline water intrusion.

First objective of this study was to identify the role of pollution sources (point and non-point) and impact of season on water quality of urban canals of Sri Lanka. The second objective was investigation of transport and mixing signature of canals and relates it with water quality. In this regard, two scales, longitudinal and reach wise, were considered in sampling. Longitudinal sampling will be used to understand the one-dimensional variations, whereas the reach wise will be used to study the two-dimensional (longitudinal and lateral) variations. The two seasons considered were dry and wet as defined by the rainfall intensity.

Insight into transport and mixing signature would reveal the fate of contaminants, and the contribution of assimilation and attenuation in that regard. Attenuation is the reduction in contaminant concentration due to dilution, mixing, and/or dispersion, whereas assimilation mainly refers to the association of contaminants by sediments (Gomes and Wai 2014). Knowledge on attenuation and assimilation processes of canals is important in rehabilitation designs. We hypothesised assimilation to play the major role in the studied canals, and testing it was the third objective. Final and fourth objective was to assess the pollution level via grey water footprint (GWF) approach and to assess the utilisation of canal natural assimilation capacity.

2 Methodology

2.1 Study Area

Gomes et al. (2014) observed a clustered response of Colombo Municipal limits canals (including Dutch canals), flood control canals of suburban Colombo, and abandoned canals of rural-urban fringe in their dentrended correspondence analysis ordination plots drawn for water quality variables. Therefore, this study selected one canal from each category. Dutch canal (DC) system is mainly confined within the Colombo

Municipality with a population density of 18,350/km^2 (Statistics.gov.lk 2016). The part of the DC system spanning from The Open University of Sri Lanka, Nawala, and flowing through Havelock town area and split Colombo 4 and 6 wards was considered for sampling (Fig. 1a). Talangama canal (TC) flows across suburbs of Colombo Municipality (e.g. Kaduwella with a population density of 2926/km^2 (Statistics.gov.lk 2016)). TC originates from Talangama Tank (built around 1550 A.C.) and converges with Kelani River, just after the Ambatale water intake. Heen canal (HC) an abandoned irrigation canal from the rural urban fringe and is in Hanwella (a population density of 775/km^2 (Statistics.gov.lk 2016)) (Fig. 1a).

2.2 Field Sampling

All three canals were sampled at every 500 m, and hereinafter will be referred as 1D or longitudinal sampling. In this regard, for every 500 m, a 50–70 m reach was selected and observations for water quality and hydraulic parameters were made in duplication or triplication along the thalweg. Water quality parameters included turbidity, EC, dissolved oxygen (DO), pH, TSS, TDS, nitrate-nitrogen, nitrite-nitrogen, ammoniacal-nitrogen, and soluble reactive phosphorus (SRP). Then a reach of 100-m length was selected that had approximately the average EC of the canal obtained by 1D sampling for a grid-based sampling to observe variations in longitudinal

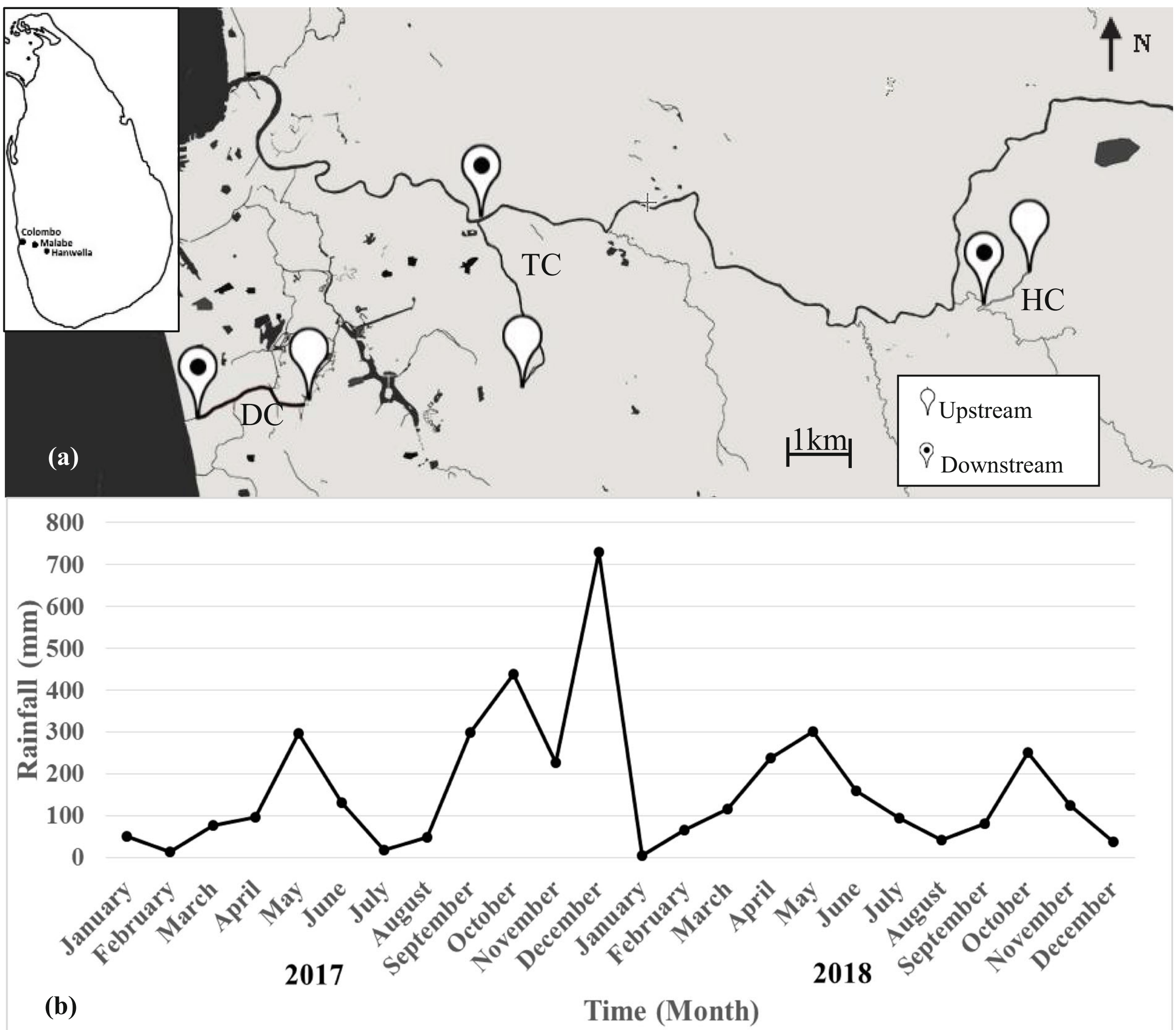

Fig. 1 **a** Locations of Dutch, Talangama, and Heen canals. **b** Monthly rainfall observed in Colombo from 2017 to 2018

and lateral directions (hereinafter referred as 2D sampling). Within this 100 m reach, there were no point source water or wastewater inputs. The nearest inputs were at least 20-m upstream. This ensured the variations within the reach were without the interference of external factors. For 2D sampling, the reach was divided into a mesh of 100 squares. Each square had dimensions of 4 m × 2 m, 4 m × 3 m, and 4 m × 1 m in TC, DC, and HC, respectively. Observations were made for each square for EC and turbidity. 2D sampling was conducted only once for a given canal in dry season; thus, the observations are rather a snapshot of a temporally dynamic process. Also, dry season characterised by low flow shows impacts of physical heterogeneity better than the wet season (Gomes et al. 2014).

1D sampling was conducted in wet and dry seasons in 2017 and 2018 (three sampling sessions for each season). Dry season sampling was carried out in July to August and/or January to March (these are typical dry seasons of the year and on average each month get less than 5% of the total annual rainfall). Wet season sampling was conducted in April to June and/or September to December: during these periods, each month get more than 10% of the total annual rainfall. Figure 1b shows the monthly rainfall observed in Colombo for 2017 and 2018 (Accuweather 2019). It should be noted the wet season's stream flow rate as well as water depth in most part of the season was about two to three times of dry season.

2.3 Laboratory and Field Water Quality Observations

pH, EC, and DO were measured in situ using meters (pH: HACH HQ411D Laboratory Single Input pH meter; EC and DO: HACH Sension + EC7; sensION+ DO6 portable DO field kit with 5130 electrode). Turbidity was measured using a turbidity meter (HACH 2100Q Portable Turbidity meter). Nitrogen species were measured by HACH methods using HACH multi-parameter portable colourimeter (DR/900, USA). Nitrate-nitrogen and nitrite-nitrogen were measured by cadmium reduction and diazotisation methods, respectively. The HACH ammonia salicylate and ammonia cyanurate reagent powder pillows that incorporate salicylate method were used for ammoniacal nitrogen measurements (APHA 1998). SRP was measured by ascorbic acid method using HACH multi-parameter portable colourimeter (DR/900, USA). TSS and TDS were measured according to gravimetric method (APHA 1998).

2.4 Physical Modelling

Physical modelling was carried out in a laboratory flume of Sri Lanka Institute of Information Technology for a straight section of Dutch canal. The flume was made of acrylic, 5-m long, 0.15-m wide, and 0.5-m deep (Rex Group, Negombo, Sri Lanka). The slope was adjustable and the flume could produce a maximum flow rate of 0.08 m^3/s with a 0.01% slope. It was necessary to use a distorted model due to the geometric dissimilarity of the canal and the flume. The similarity between the model and the prototype was realised by dynamic similarity based on the Froude number (*Fr*). The lateral scale and the vertical scales were taken as 1:120($(Lr)_H = 120$) and 1:40 ($(Lr)_V = 40$), respectively (Ettema et al. 2000). The prototype's transport pattern was observed for a 100-m distance using 'Domba' (*Calophyllum inophyllum*) seeds (diameter ~ 25 mm) released at the centre of the straight reach, and in the physical model, plastic beads (diameter 3 mm) were used. Model was executed for three flow conditions: low, high, and bankfull flows. Low flow had half the discharge of high flow. Validation of the physical model was done for the high flow.

2.5 Data Analysis

Contour diagrams of EC and turbidity in 2D sampling were drawn using Surfer 16. Significant differences between two and more than two groups were realised by *t* test and one-way ANOVA, respectively. In this regard, IBM SPSS (version 20) was used. *Fr* number was calculated using Eq. (1) where U is the average velocity, g is the gravitational acceleration, and D is the hydraulic depth.

$$F_r = \frac{U}{\sqrt{gD}} \tag{1}$$

GWF is the ratio between pollution load to critical pollution load (Eq. 2). The GWF can express in m^3/s, when Eq. (2) is multiplied by the flow rate (Q) (Eq. 3). L is the pollution load (mg/L) and $L_{critical}$ is the load (mg/L) of a pollutant that will fully consume the assimilation capacity of the receiving water body (Eqs. 4 and 5, respectively).C_{max} is the maximum acceptable concentration (also known as critical concentration). The natural concentration of a pollutant is denoted by $C_{natural}$ and is outlined in Table 1. $C_{natural}$ values are the typical

Table 1 Ambient and natural concentrations of pollutants

Parameter	Critical concentration	Reference	Natural	Reference
Nitrate-nitrogen (mg/L)	13	CCME (2013)	0.4428	Chapman et al. (1996)
Nitrite-nitrogen (mg/L)	0.06	CCME (2013)	0	Franke et al. (2013)
Ammoniacal nitrogen (mg/L)	0.025	CCME (2013)	0.015	Chapman et al. (1996)
Turbidity (NTU)	15	Franke et al. (2013)	0	Franke et al. (2013)
Total suspended solids (mg/L)	25	CCME (2013)	15	Chapman et al. (1996)
Total dissolved solids (mg/L)	600	Franke et al. (2013)	0	Franke et al. (2013)

natural concentrations of pristine water bodies (Chapman et al. 1996; Franke et al. 2013).

$$\text{GWF (as a ratio)} = \frac{L}{L_{\text{critical}}} \tag{2}$$

$$\text{GWF (volume/time)} = \frac{L}{L_{\text{critical}}} \times Q \tag{3}$$

$$L = (C_{\text{actual}} - C_{\text{natural}}) \times Q \tag{4}$$

$$L_{\text{critical}} = (C_{\max} - C_{\text{natural}}) \times Q \tag{5}$$

3 Results and Discussion

3.1 Seasonal Water Quality Variations and Contribution of Non-point Pollution Sources

Surface runoff is a major source of non-point pollution and a major factor that govern lotic water quality (Vidal and Melgar 2000). Quality and quantity of runoff are governed by factors such as the land use, land cover, and rainfall intensity (Wang et al. 2015). Table 2 shows the water quality observations made for DC, TC, and HC for both seasons. The inorganic nitrogen and SRP of DC were significantly (one-way ANOVA; $P < 0.05$) higher than TC in both seasons. A similar observation was observed in dry season; however, the differences were not statistically significant (one-way ANOVA; $P > 0.05$) for some parameters.

Nitrogen can be in several forms in surface runoff including particulate (organic and inorganic), and dissolved nitrogen. Dissolved nitrogen is in the forms of nitrate-nitrogen, nitrite-nitrogen, and ammoniacal nitrogen. Nitrite is the most unstable, and its concentration in surface runoff considered to be insignificant (Veuger et al. 2004). Out of total nitrogen, only a small fraction is bioavailable; these include nitrate-nitrogen, ammoniacal nitrogen, and some low molecular weight organic nitrogen (Veuger et al. 2004). Out of inorganic nitrogen in DC, ammoniacal nitrogen represented more than 50%. In TC, the dominant nitrogen species was nitrate, and it contributed about 60% of inorganic nitrogen. Nitrate-nitrogen concentrations can be largely attributed to biogenic waste such as human and animal excreta as well as products of nitrification. Since it exerts an oxygen demand, it can potentially result in oxygen depletion (Jinadasa et al. 2012).

TC and DC had the highest and lowest DO values, respectively. However, in general, the DO concentrations did not show significant differences between canals for a given season (one-way ANOVA; $P < 0.05$). When the sewage and effluents are released into a canal, the active degradation (oxidation) of organic matter in the water consumes DO, leading to its rapid depletion (Kumar and Reddy 2008). High DO can be a reason for high nitrate fraction in TC as ammonia and nitrite rapidly oxidised into nitrate. One of the reasons DC having high concentrations of ammoniacal nitrogen was due to its exposure to the sewage and other effluents from the surrounding high-density low-income human settlements. In addition, quick drainage across impermeable and/or non-vegetated riparian zones of urban streams and canals may lower the nitrification and lead to high ammoniacal nitrogen concentrations (Kaye et al. 2006). In comparison, the catchment modifications and exposure to the sewage in TC were less than DC. Even

Table 2 Water quality of Dutch, Talangama, and Heen canals in wet and dry seasons. Parentheses show standard deviation

	DC		TC		HC	
	Wet	Dry	Wet	Dry	Wet	Dry
Dissolved oxygen (mg/L)	4.4 (2.7)	5.1 (1.2)	5.4 (1.3)	5.2 (0.88)	4.8 (0.69)	5.3 (1.2)
pH	7.0 (0.24)	6.7 (0.55)	6.7 (0.39)	6.5 (0.64)	6.6 (0.48)	7.1 (0.60)
Electrical conductivity (μS/cm)	259 (96)	261 (44)	131 (23)	150 (28)	50 (2.5)	50 (3.5)
Total solids (mg/L)	226 (106)	313 (47)	264 (136)	275 (110)	129 (65)	138 (53)
Total suspended solids (mg/L)	34 (13)	25 (12)	57 (57)	76 (71)	23 (13)	38 (12)
Total dissolved solids (mg/L)	124 (32)	265 (46)	351 (183)	200 (118)	30 (8.6)	51 (1.5)
Temperature (°C)	21 (0.54)	24 (1.6)	23 (0.43)	24 (0.65)	28 (0.48)	27 (0.59)
Turbidity (NTU)	21 (9.3)	14 (5.1)	53 (52)	96.55 (80.79)	20 (15)	45 (18)
Nitrate-nitrogen (mg/L)	1.6 (1.0)	0.79 (0.37)	2.3 (0.85)	1.76 (2.20)	0.34 (0.51)	0.49 (0.27)
Nitrite-nitrogen (mg/L)	0.02 (0.02)	0.03 (0.02)	0.02 (0.02)	0.02 (0.01)	0.001 (0.0011)	0.03 (0.01)
Ammoniacal-nitrogen (mg/L)	2.3 (2.6)	1.3 (0.93)	0.28 (0.19)	0.43 (0.19)	0.14 (0.21)	0.37 (0.22)
Soluble reactive phosphorus (mg/L)	0.75 (0.71)	0.28 (0.20)	0.49 (0.28)	0.29 (0.07)	0.18 (0.04)	0.25 (0.04)

though parameters that are surrogates of nutrients showed DC to be inferior in water quality, solid-related parameters in certain cases showed otherwise. As an example, TSS of TC was about 1.5 to 2 times as of DC, and the differences were statistically significant too (one-way ANOVA; $P < 0.05$).

Dry season showed a better water quality than wet season with respect to DC and TC for most of the parameters. As examples, nitrate nitrogen and SRP were 20–60% higher in wet season, and in many cases, the differences were statistically significant (one-way ANOVA; $P < 0.05$). Same pattern was observed for solid-related parameters such as TS and turbidity. Ongoing constructions in canal bed and banks (bank modifications with gabions) can be a reason for unusually high values of turbidity, TS, and TSS of TC. High concentration for EC in dry season could be due to concentrated dissolved solids (Alam et al. 2007).

The water quality of HC gave opposite observations to those of DC and TC. Firstly, its wet season showed a better water quality for almost all parameters. These observations elucidated the role of the catchment in governing the water quality with the change of season. DC and TC characterised by urban and peri-urban catchments resulted in poor water quality in wet season, and this should be due to a contaminated surface runoff. On the other hand, HC in a relatively rural catchment resulted in a diluted canal due to less or uncontaminated surface runoff.

3.2 Longitudinal Water Quality Variation and Contribution of Point Pollution Sources

In general, no significant variation of EC was observed in all three canals (Fig. 2a–f) along the longitudinal direction in both seasons. However, one or two peaks were observed in many sampling sessions of DC and TC (e.g. stations 3 and 5 of Fig. 2b and d, respectively) and seemed to be random occurrences as those peaks did not show up at the same station in previous and/or subsequent sampling sessions. HC showed a unique variation in wet season, such that the relatively high EC value at the first station was gradually reduced up to a particular downstream station and afterwards was constant with distance (Fig. 2f). This we attributed to a dilution signature due to increased flow rate as a result of unpolluted catchment runoff of HC. In urban canals, point source inputs (e.g. expedient connections) often attributed to the EC variations, and EC is considered as an easy low-cost observation to identify expedient connections (Gomes and Wai 2014). Urban water bodies are adversely affected by point sources of contamination as results of direct or indirect human activities (e.g. discharge of domestic effluents) and industrial practices (Urbaniak et al. 2013; Lenart-Boroń et al. 2015). We expected EC to show peaks in its longitudinal profile due to point source pollutant discharges, and non-existence of such was rather surprising. However, the variation of nitrate nitrogen and SRP showed signs of sampling station-specific peaks (Fig. 3a–f).

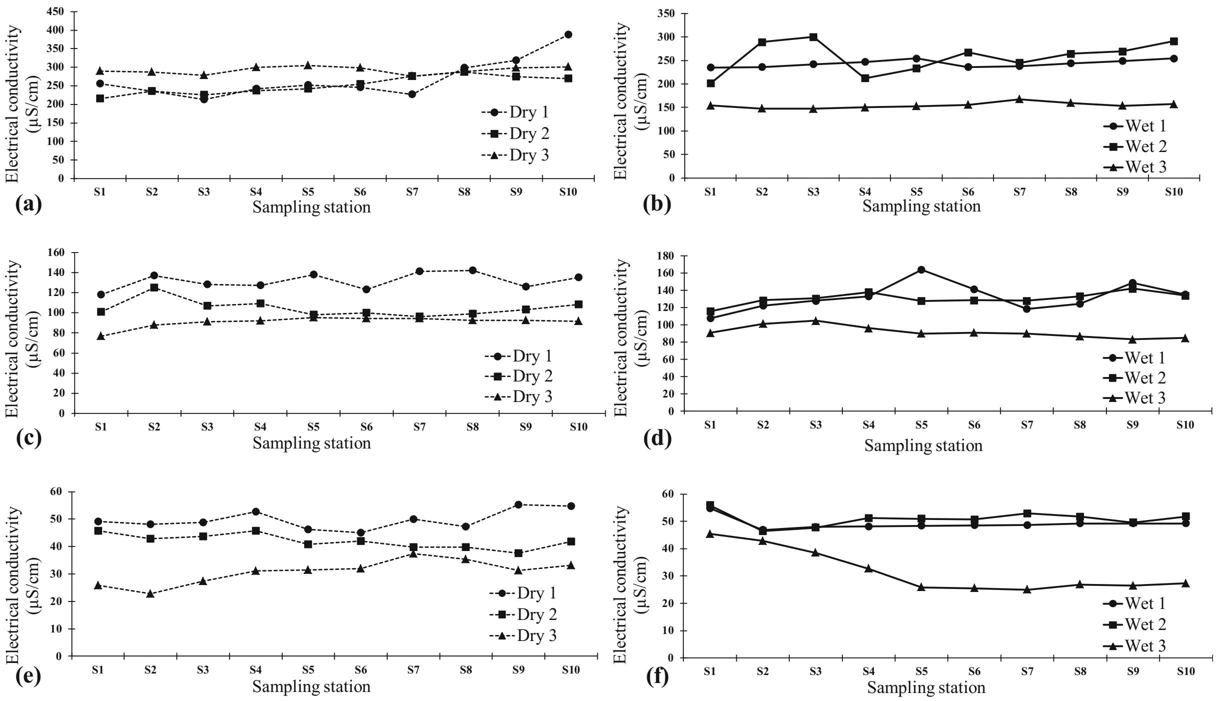

Fig. 2 Electrical conductivity variation in Dutch (**a**, **b**), Talangama (**c**, **d**), and Heen (**e**, **f**) canals in dry and wet seasons (S_1 is the upstream reach)

Nitrate variation of DC in both seasons was without a proper pattern (Fig. 3a and b). In dry season other than the first, and the last two reaches, and in wet season, the mid reaches (fourth to sixth), the nitrogen variation was ad hoc with peaks and lumps emerged without a consistent spatiotemporal pattern. This could be due to several point source pollution paths that release contaminants on ad hoc basis. TC showed relatively high nitrate concentrations from second to fourth reaches in dry season, and second to fifth in wet season (Fig. 3c and d); other than that, the nitrogen variation was more or less the same between different sampling sessions of a given season. HC's nitrate variation in both seasons showed a decreasing trend with distance downstream, and this was obvious in wet season (Fig. 3e and f). The high concentrations at the first reach indicated the influence of a human settlement about 1-km upstream (note that EC concentration was also relatively high at the beginning).

Unlike nitrate variation, SRP variations showed a canal specific, but season independent trends (Fig. 4a–f). As an example, DC showed relatively high concentrations in and around third, seventh, and eighth reaches in both seasons (Fig. 4a and b). Similarly, in and around the third reach of TC, both seasons gave high concentrations (Fig. 4c and d). However, in wet season, TC and HC showed relatively similar concentrations at a given reach amongst different sampling sessions than the dry season. (Fig. 4e and f).

The nitrate-nitrogen and SRP variation seemed to be more analogues to point source pollution than EC. Diffusivity of ions in a solution governs EC. Organic compounds such as oil, grease, and petrochemicals having low diffusivity are more resistive to electrical current and may weaken the EC observations (Martin et al. 1996). Also, several past studies referring open channels (e.g. Gali et al. 2012) observed a positive relationship between EC and nitrate nitrogen, whereas a negative relationship between EC and SRP. Furthermore, ammoniacal nitrogen can show a negative relationship with EC (Gali et al. 2012). Therefore, the combined effects of different constituents seemed made EC variation less responsive to point source pollution sources. In this regard, EC as a

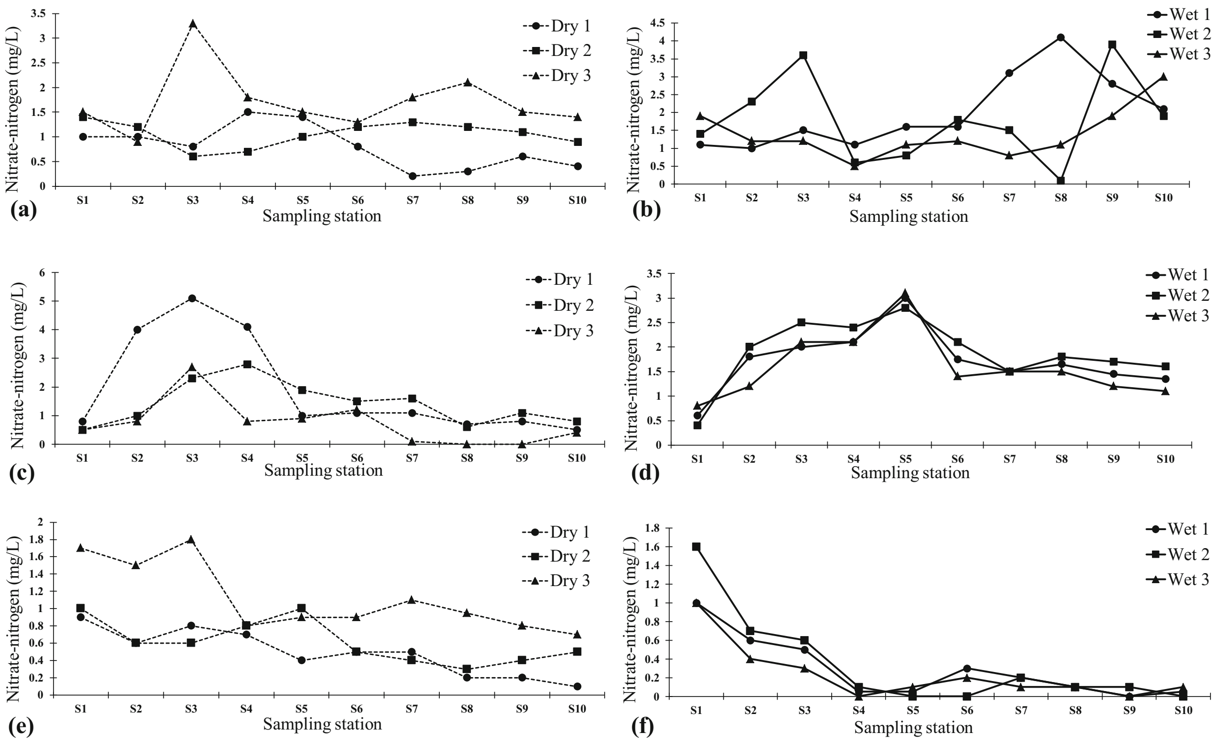

Fig. 3 Nitrate nitrogen variation in Dutch (**a**, **b**), Talangama (**c**, **d**), and Heen (**e**, **f**) canals in dry and wet seasons

method of point source pollution source identification seemed not applicable.

3.3 2D Variation of Electrical Conductivity: Short-Scale Mixing and Transport Signature

In TC, the 2D variation of EC (a surrogate of TDS) at the extreme (i.e. considering the difference between highest and lowest concentrations) was 12 μS/cm, which is about 9% from the reach mean (Fig. 5a). For DC and HC, this was about 60 μS/cm (about 20% from the mean) and 4 μS/cm (about 8% from mean), respectively (Figs. 6a and 7a). The contours indicate the extent of the mixing zones for pollutants of different concentrations (McCorquodale et al. 1983). The EC along the left bank was relatively high in TC and it could be related to the road running parallel to the canal that acted as a non-point pollution source (Walker et al. 1999). Furthermore, the EC patches of TC seemed originated from the left bank and elongated in the longitudinal direction: this was an indication of a relatively weak diffusion in the lateral direction. The turbidity variation of TC (Fig. 5b) was analogues to its EC variation, suggesting the impact of road as a pollution source as well as poor mixing in lateral direction. DC too had a similar parallel road in its left bank. The variation of EC of DC was different to that of TC. DC showed a uniformly distributed zone at the end of the reach. It resembles a mixing scenario of a prismatic straight channel where the release point is middle of an upstream cross section (Beltaos 1979). The reasons for different variations of EC between TC and DC could be due to the influence of a high strength wastewater discharge in DC's upstream that overridden the contribution made by the road on EC of the left bank. However, DC's turbidity variation was similar to that of TC (Figs. 6b and 5b), an indication that the road is a main contributor of TDS.

In case of HC, the EC variation was uniform in its 2D profile and indicated a strong diffusion and mixing signature in lateral and longitudinal directions (Fig. 7a). Therefore, with respect to dissolved solids, it was certain that TC (and also to a certain extent DC) did not show appropriate mixing. Poor mixing meant the expected dilution and the subsequent assimilation is weakened (McCorquodale et al. 1983). The variation of turbidity of HC was patchy and was evenly

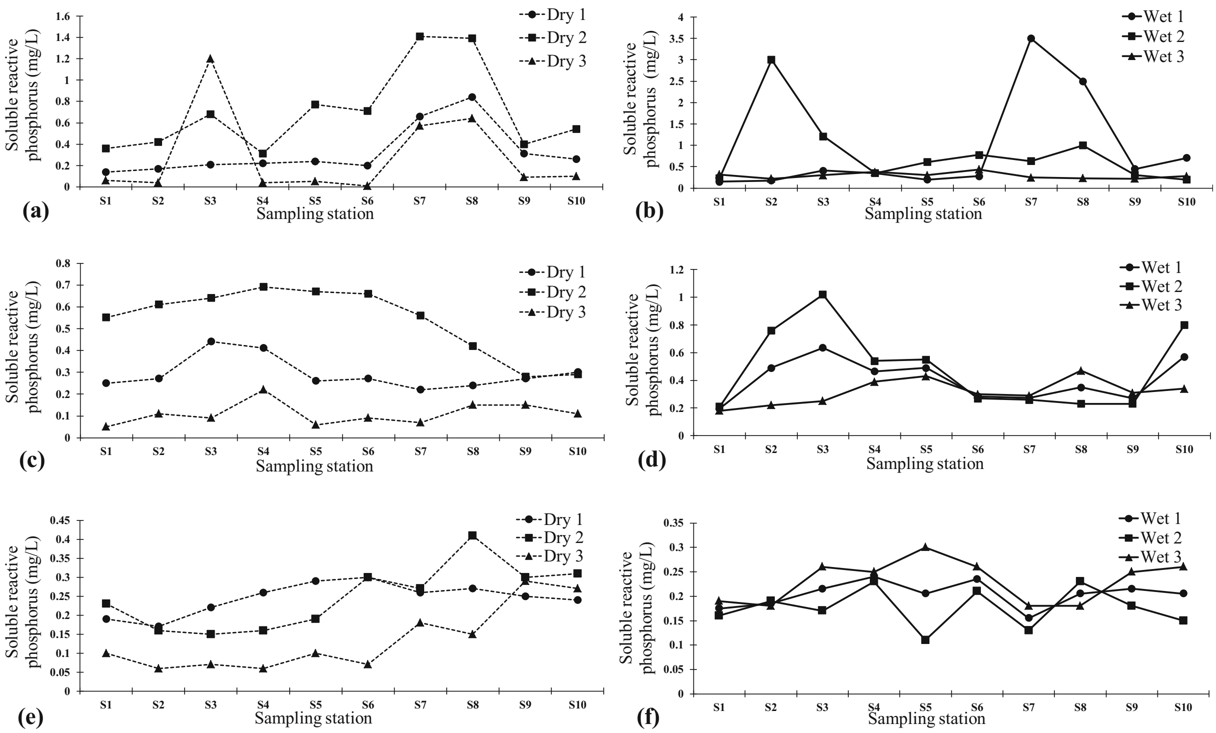

Fig. 4 Soluble reactive phosphorus variation in Dutch (**a**, **b**), Talangama (**c**, **d**), and Heen (**e**, **f**) canals in dry and wet seasons

distributed in lateral and longitudinal directions (Fig. 7b). This in turn explained the uniform distribution of EC in HC. Several patches of turbidity meant there were hydraulic patches as

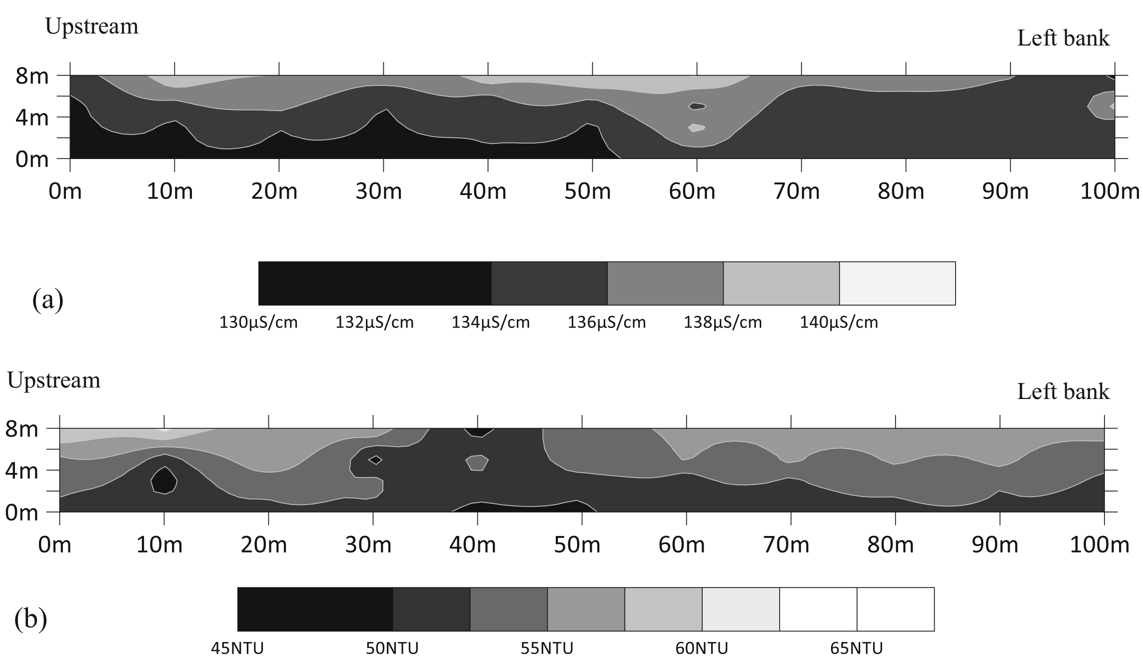

Fig. 5 Longitudinal and lateral variations of electrical conductivity (**a**) and turbidity (**b**) in a mid-reach of Talangama canal in dry season

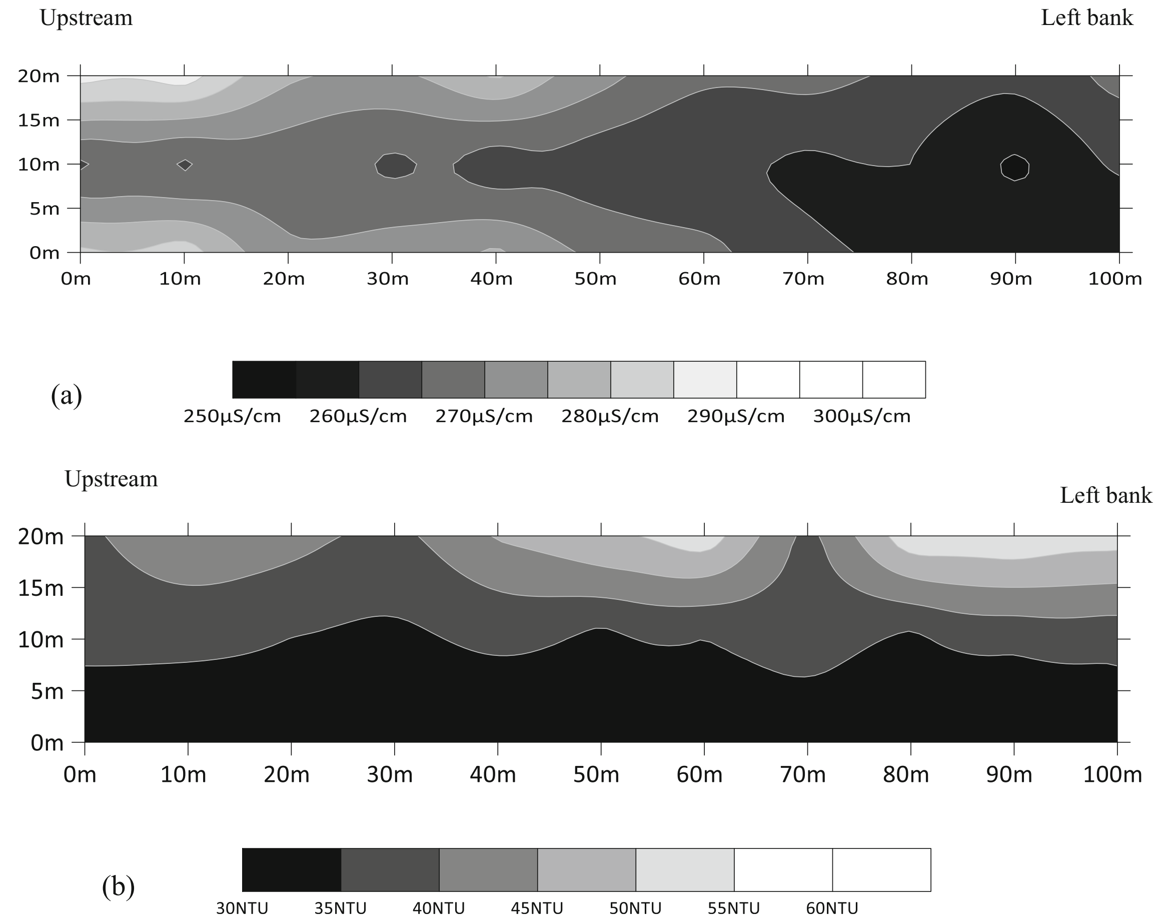

Fig. 6 Longitudinal and lateral variations of electrical conductivity (**a**) and turbidity (**b**) in a mid-reach of Dutch canal in dry season

results of high and low velocities and/or depth zones (i.e. due to bed/instream heterogeneity). Velocity and wetted depth are directly related to mixing (McCorquodale et al. 1983), and such hydraulic patches promoted mixing of dissolved solids effectively by way of turbulent diffusion. HC's turbidity values in the extreme showed a 50% difference, which is relatively significant for a short reach of 100 m. HC being an abandoned canal (i.e. with less to no maintenance such as dredging) seemed to have a heterogeneous instream environment with epi-faunal structures, sediment depositions, bank collapses, etc.

In general, no differences were observed between dry and wet season of a given canal in 2D variations (data not shown). However, the patches observed in HC were less in wet season as a result of more homogenous flow conditions due to the less influence from the canal bed heterogeneity with the increased water depth (Gomes et al. 2014).

3.4 Transport of Constituents Against Flow Stage

The dominance of molecular diffusion in the longitudinal direction can be further validated by the investigation of advection, the strength of mass transport of constituents (Bottacin-Busolin et al. 2011). Table 3 shows the results of a distorted scale physical model of the Dutch canal conducted for floating constituents. The weakest advection should be from the lowest possible flow (taken as half of the wet season flow), whereas the strongest advection should be from bankfull stage. For a minimum water depth of 0.9 m, floating

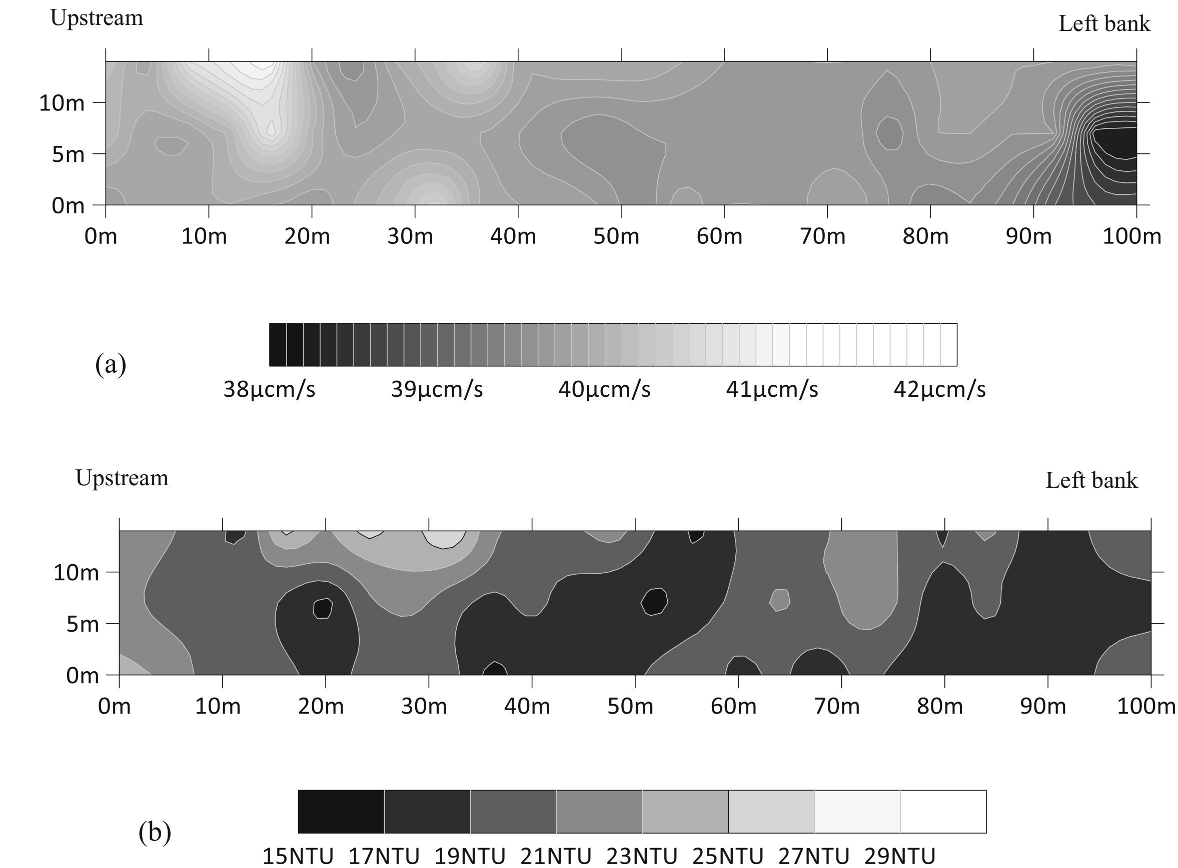

Fig. 7 Longitudinal and lateral variations of electrical conductivity (**a**) and turbidity (**b**) in a mid-reach of Heen canal in dry season

constituents showed a longitudinal velocity of 0.22 m/s. The high flow (i.e. wet season flow with a depth of 1.8 m) showed 0.40 m/s longitudinal velocity for floating constituents. The bankfull water depth was 4.2 m, and that stage gave floating constituents a 0.58 m/s longitudinal velocity. These results elucidated that advection transport was ineffective in transporting constituents in all flow stages other than the bankfull. However, the probability of annual occurrence of a bankfull stage is about 2%. Therefore, it was conspicuous natural cleansing of the canal system by way of flushing is practically inexistent. This could be problematic as accumulated pollutants as well

Table 3 Prototype and physical model results of the Dutch canal

Parameter	Wet season flow	Bankfull flow	Dry season flow
Manning's roughness coefficient ($m^{-1/3}s$)	0.065	0.04	0.065
Model velocity (velocity of water) as per Froude similarity (m/s)	0.22	0.1	0.037
Related model wetted depth when substrate is gravel (mention under methodology) (mm)	77	107	30
Velocity of floating beads in the model (m/s)	0.53	0.092	0.034
Prototype wetted depth (m)	1.8	4.2	0.9
Velocity of floating beads in the prototype (m/s)	0.4	0.58	0.22
Froude number of the prototype	0.09	0.09	0.07

Table 4 Grey water foot print as a ratio of Dutch (DC), Talangama (TC), and Heen (HC) canals

	DC		TC		HC	
	Wet	Dry	Wet	Dry	Wet	Dry
Total suspended solids	1.4 (1.5)	0.95 (1.2)	4.2 (5.7)	6.1 (7.1)	0.77 (1.3)	2.3 (1.2)
Total dissolved solids	0.16 (0.10)	0.44 (0.08)	0.59 (0.31)	0.33 (0.20)	0.05 (0.01)	0.09 (0.003)
Turbidity	1.4 (0.62)	0.92 (0.34)	3.6 (3.5)	6.4 (5.4)	1.3 (1.0)	3.0 (1.2)
Nitrate-nitrogen	0.09 (0.08)	0.03 (0.03)	0.14 (0.07)	0.10 (0.18)	0.01 (0.04)	0.004 (0.02)
Nitrite-nitrogen	0.35 (0.30)	0.46 (0.27)	0.36 (0.39)	0.26 (0.13)	0.02 (0.02)	0.44 (0.15)
Ammoniacal-nitrogen	231 (259)	130 (92)	27 (19)	41 (19)	12 (21)	35 (22)
Soluble reactive phosphorus	7.6 (28)	11 (7.9)	19 (11)	11 (2.8)	6.6 (1.6)	9.4 (1.7)

continuously supplied contaminants would remain in the system for prolonged periods.

3.5 Dominant Pollutant Removal Mechanisms and Assimilation Capacity of Canals

It was obvious the poor advection and dominant molecular diffusion signature of DC and TC. Therefore, the fate of contaminants lies on assimilation capacity. Assimilation can be through sediments, algae, and plants (Eppley and Rogers 1970). In DC, other than floating macrophyte, water hyacinth (*Eichhornia crassipes*) no plants were observed. TC had patches of emergent plants (*Eleocharis quadrangulata*, *Nymphaea* sp.), but were being removed (dredged out) frequently, so do the floating macrophytes. Therefore, the dominant assimilation path had to be the streambed sediments and filamentous algae. Any ecosystem (in this case canal) has its own natural assimilation capacity, and it could be assessed and approximated via GWF.

Table 4 shows the GWF as a ratio and the main contributor of GWF in all streams for both seasons was ammoniacal nitrogen. Similar to water quality results, wet season showed significantly high GWF values relative to dry season (one-way ANOVA; $P < 0.05$), irrespective of conveying a twofold flow rate. The contribution of ammoniacal nitrogen was several folds (200–600 times) higher than the other nitrogen species (nitrate-nitrogen and nitrite-nitrogen). In HC, the contribution from nitrate or nitrite on GWF was marginal. The second highest contributor of GWF in all canals was SRP, wherein DC showed values of 11 (dry) to 7.6 (wet). TC's wet season gave the highest GWF for SRP. In HC, the GWF by SRP was similar to values of other two canals. Even though DC showed disproportionately high values of GWF pertaining to nutrients, the sediment related, or surrogates of sediment parameters, the differences were not contrasting. In fact, it was TC that showed significantly high values (one-way ANOVA; $P < 0.05$) for GWF of TSS.

The GWF as a ratio indicates the appropriated (used) natural assimilation capacity of the canal. DC, ammoniacal nitrogen assimilation capacity had been exceeded by 23,000 times in wet season whereas in dry, it was by a margin of 13,000. However, in other canals, it was less than 3000 times exceedance. Even though the water quality suggested wet season had a poor in water quality in DC and TC, with respect to assimilation capacity for many parameters, TC and HC showed dry season to have exceed more assimilation capacity than wet season. This implied a dilution signature by way of increased flow and/or increased assimilation by way of sediment reactivation took place in wet season, in line with the conventional understanding.

Relative to nitrogen species, the GWF from SRP was very low, probably due to association of phosphates by sediment (Table 5 shows relatively high values of soil phosphorus). Unlike nitrogen,

Table 5 Soil nutrients levels of Dutch (DC), Talangama (TC), and Heen (HC) canals

	Nitrate-nitrogen (mg/kg)	Soluble reactive phosphorus (mg/kg)
DC	0.27 (0.07)	1.2 (0.45)
TC	0.21 (0.10)	1.7 (0.60)
HC	0.03 (0.06)	0.21 (0.18)

phosphorus is well associated with stream/river transport and mixing mechanisms, as such it gets bound with sediments easily (Yuan et al. 2018). Also, phosphorus is usually a limited nutrient in aquatic ecosystems as they are readily consumed by plants and algae (Edwards et al. 2000).

Furthermore, the representation of sediment parameters by GWF was relatively weak, and indicated sediment pollution of these canals was less. However, this could be also due to the fact that sediment deposition (settlement) due to subcritical flow conditions, poor advection, and dominance molecular diffusion.

3.6 Management Implications

The flow conditions of DC and TC were well within subcritical range for both seasons. Also, the *Fr* number showed similar values in its 2D profile. Even though the average Fr number of HC was in subcritical range, it showed a high variance ranging from 0.1 to 1.1 (data not shown), elucidating hydraulic heterogeneity. For DC, even the bankfull *Fr* number was well within subcritical. Homogeneous and subcritical flow conditions would not trigger attenuation, and considering the already over utilised assimilation capacity of DC and TC, this can be problematic. Therefore, it is essential to create a heterogeneous environment characterised by several mesoscale physical habitats (e.g. pools and riffles). One option in this regard is placing deflectors at appropriate orientations and distances (Gomes and Wai 2014). Undoubtedly such would increase bed friction. Considering the fact that most of these canals did not overflew over the bank during last 50 years or so, increase in bed friction to a certain level should be acceptable. However, flood safety should be validated via physical model experiments and numerical simulations.

Furthermore, instead of complete removal such as in TC, controlled removal of emergent vegetation is recommended. In recent years, during the rainy season (North-West monsoon), TC overflew over the banks. This was as a result of backflow created by the rise in water level of the Kelani River. Therefore, overflow of TC is due to a larger factor, and impact from instream structures would be negligible. Flood plain of TC has unutilised space, therefore, could be used to create flood retaining areas (by way of wetlands).

HC, an abandoned irrigation canal, gave insights to naturalisation potential of canals; it was simply left unattended. Obviously, urban canals such as DC would not naturalise without catchment and instream management and/or engineering interventions. In this regard, pollutant inputs via point and non-point sources have to be curtailed (catchment scale intervention), in addition to creation of instream heterogeneity (instream intervention).

4 Conclusion

The subcritical flow canals of urban, peri-urban, and rural-urban fringe studied for a 2-year period revealed urban and peri-urban canals to show much inferior water quality in wet season, elucidating impact of catchment laden pollutant runoff. This was in contrast to the common local perception that rainy season would flush out pollutants. Nevertheless, the canal from rural urban fringe was healthier in wet season.

Point source pollution was more evident in urban and peri-urban canals, and the input sources were spatially spread yet discontinuous with time. At a given sampling session, a peak or two (relative maxima) were observed for water quality parameters that are representatives of poor water quality (e.g. EC and nitrate-nitrogen). Impact of such peaks was highly localised, as no elevated concentrations were observed in the nearest upstream or the downstream reaches. This elucidated the response time of a downstream reach to show up an upstream pollutant release was relatively high. Two-dimensional variation of EC and turbidity of urban and peri-urban canals suggested the dominance of molecular diffusion in longitudinal as well as lateral directions. However, the canal from the rural urban fringe showed a patchy distribution of turbidity and was obvious in the dry season; this indicated attenuation also playing a major role on the fate of pollutants.

The dominance of molecular diffusion was further evidenced by physical models, where it proved not only low flow conditions but also bankfull conditions would not induce advective transport and mixing. This meant, the constituents would remain in the canal for prolonged periods. Domination of molecular diffusion meant, pollution removal must be assimilation dominated.

The GWF analyses showed the urban and semi-urban canals have already exceeded their natural assimilation capacities for most parameters by at least 1000 times, and the situation was worst in DC. Therefore, it is suggested to improve attenuation processes as a potential rehabilitation option.

Funding Information Authors wish to thank the National Research Council of Sri Lanka for funding this research project (Grant number NRC 17-066).

Data Availability The data that support the findings of this study are available from the corresponding author upon reasonable request.

References

Accuweather. (2019). https://www.accuweather.com/en/lk/colombo/311399/march-weather/311399?monyr=3/1/2019&view=table. Accessed 23 Mar 2019.

Alam, M. J., Islam, M. R., Muyen, Z., Mamun, M., & Islam, S. (2007). Water quality parameters along rivers. *International Journal of Environmental Science & Technology, 4*(1), 159–167.

American Public Health Association A.P.H.A. (1998). *Standard methods for the examination of water and wastewater.*

Beltaos, S. (1979). Transverse mixing in natural streams. *Canadian Journal of Civil Engineering, 6*(4), 575–591.

Bottacin-Busolin, A., Marion, A., Musner, T., Tregnaghi, M., & Zaramella, M. (2011). Evidence of distinct contaminant transport patterns in rivers using tracer tests and a multiple domain retention model. *Advances in Water Resources, 34*(6), 737–746.

CCME. (2013). Canadian water quality guidelines for the protection of aquatic life.Canadian Council of Ministers of the Environment. Winnipeg, Canada. http://ceqg-rcqe.ccme.ca/download/en/221 Date accessed 31-12-2013

Chapman, D., UNESCO, WHO, & UNEP. (1996). *Water quality assessments - a guide to use of biota, sediments and water in environmental monitoring* (2nd ed.). London: Chapman & Hall.

Edwards, A., Cook, Y., Smart, R., & Wade, A. (2000). Concentrations of nitrogen and phosphorus in streams draining the mixed land-use Dee Catchment, north-east Scotland. *Journal of Applied Ecology, 37*(s1), 159–170.

Eppley, R. W., & Rogers, J. N. (1970). Inorganic nitrogen assimilation of Ditylum brightwellii, a marine plankton diatom 1, 2. *Journal of Phycology, 6*(4), 344–351.

Ettema, R., Arndt, R., Roberts, P., & Wahl, T. (2000). *Hydraulic modeling*. New York: American Society of Civil Engineers.

Franke, N., Boyacioglu, H., & Hoekstra, A. (2013). *Grey water footprint accounting: tier 1 supporting guidelines*. Delft: UNESCO-IHE.

Gali, R. K., Soupir, M. L., & Helmers, M. J. (2012). Electrical conductivity as a tool to estimate chemical properties of drainage water quality in the Des Moines Lobe, Iowa. In 2012 Dallas, Texas, July 29-August 1, 2012 (p. 1). American Society of Agricultural and Biological Engineers.

Gomes, P. I. A., & Wai, O. W. H. (2014). Sampling at mesoscale physical habitats to explain headwater stream water quality variations: its comparison to equal-spaced sampling under seasonal and rainfall aided flushing states. *Journal of Hydrology, 519*, 3615–3633.

Gomes, P. I. A., Wai, O. W., Kularatne, R. K. A., Priyankara, T. D. P., Anojika, K. G. M. S., & Kumari, G. M. N. R. (2014). Relationships among anthropogenic disturbances representative riparian and non-riparian herbaceous indicators (biomass and diversity), land use, and lotic water quality: implications on rehabilitation of lotic waters. *Water, Air, & Soil Pollution, 225*(9), 2060.

Gomes, P. I. A., Weerasinghe, G., & Paulnath, R. C. M. (2016). Deriving eco-hydraulic reference conditions for physical heterogeneity for canals in Colombo, 110th Annual Sessions of Institution of Engineers Sri Lanka (IESL), Colombo, Sri Lanka, 17–18 October 2016.

Jinadasa, K. B. S. N., Wijewardena, S. K. I., Zhang, D. Q., Gersberg, R. M., Kalpage, C. S., Tan, S. K., et al. (2012). Socio-environmental impact of water pollution on the mid-canal (Meda Ela), Sri Lanka. *Journal of Water Resource and Protection, 4*(07), 451–459.

Kaye, J., Groffman, P., Grimm, N., Baker, L., & Pouyat, R. (2006). A distinct urban biogeochemistry. *Trends in Ecology & Evolution, 21*(4), 192–199.

Kiat, C., Ghani, A., Abdullah, R., & Zakaria, N. (2008). Sediment transport modeling for Kulim River – a case study. *Journal of Hydro-Environment Research, 2*(1), 47–59.

Kim, H., & Jang, C. (2019). A review on ancient urban stream management for flood mitigation in the capital of the Joseon Dynasty, Korea. *Journal of Hydro-Environment Research, 22*, 14–18.

Kumar, A., & Reddy, M. (2008). Assessment of seasonal effects of municipal sewage pollution on the water quality of an urban canal—a case study of the Buckingham canal at Kalpakkam (India): NO3, PO4, SO4, BOD, COD and DO. *Environmental Monitoring and Assessment, 157*(1–4), 223–234.

Lenart-Boroń, A., Wolanin, A., Jelonkiewicz, Ł., Chmielewska-Błotnicka, D., & Żelazny, M. (2015). Spatiotemporal variability in microbiological water quality of the Białka River and its relation to the selected physicochemical parameters of water. *Water, Air, & Soil Pollution, 227*(1), 1–12.

Martin, F. D., Robert M., & Colpitts P. G. (1996). *Standard handbook of petroleum and natural gas engineering.* (6th ed). 2(38).

McCorquodale, J. A., Imam, E. H., Bewtra, J. K., Hamdy, Y. S., & Kinkead, J. K. (1983). Transport of pollutants in natural streams. *Canadian Journal of Civil Engineering, 10*(1), 9–17.

Ministry of Megapolis and Western Development, Western Region Megapolis Planning Project. (2018) The Megapolis; Western Region Master Plan-2030 Sri Lanka, Ministry of Megapolis and Western Development. https://drive.google.com/file/d/1G-GtIjc-sOZ_tO6p-2-lEGKQsQlF9zpB/view. Accessed 8 May 2018.

Nakamura, K., Tockner, K., & Amano, K. (2006). River and wetland restoration: lessons from Japan. *BioScience, 56*(5), 419–429.

Statistics.gov.lk. (2016). Population and population density by sectors and D.S. Division-2015. http://www.statistics.gov.lk/statistical%20Hbook/2016/Colombo/Table2.4.pdf. Accessed 22 Mar 2019.

Urbaniak, M., Kiedrzyńska, E., Kiedrzyński, M., Mendra, M., & Grochowalski, A. (2013). The impact of point sources of pollution on the transport of micro pollutants along the river continuum. *Hydrology Research, 45*(3), 391–410.

Veuger, B., Middelburg, J., Boschker, H., Nieuwenhuize, J., van Rijswijk, P., Rochelle-Newall, E., et al. (2004). Microbial uptake of dissolved organic and inorganic nitrogen in Randers Fjord. Estuarine. *Coastal and Shelf Science, 61*(3), 507–515.

Vidal, M., & Melgar, M. (2000). Spatiotemporal characterization of groundwater contamination as a result of urban effects. *Water, Air, and Soil Pollution, 121*(1/4), 367–377.

Walker, W. J., McNutt, R. P., & Maslanka, C. K. (1999). The potential contribution of urban runoff to surface sediments of the Passaic River: sources and chemical characteristics. *Chemosphere, 38*(2), 363–377.

Wang, A., Tang, L., & Yang, D. (2015). Spatial and temporal variability of nitrogen load from catchment and retention along a river network: a case study in the upper Xin'anjiang catchment of China. *Hydrology Research, 47*(4), 869–887.

Yuan, S., Tang, H., Xiao, Y., Chen, X., Xia, Y., & Jiang, Z. (2018). Spatial variability of phosphorus adsorption in surface sediment at channel confluences: field and laboratory experimental evidence. *Journal of Hydro-Environment Research, 18*, 25–36.

Zhang, J., & Millero, F. (1993). Kinetics of oxidation of hydrogen sulfide in natural waters. *ACS Symposium Series* (pp. 393-409).