Ecological Engineering 94 (2016) 516–524

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Land use influence on raw surface water quality and treatment costs for drinking supply in São Paulo State (Brazil)

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ARTICLE INFO

Article history:
Received 4 April 2016
Received in revised form 28 May 2016
Accepted 16 June 2016

Keywords:
Drinking water
Infrastructure and operating costs
Land cover
Rivers and streams
Water treatment plants

ABSTRACT

Anthropogenic disturbances to the environment can compromise valuable ecosystem services, including the provision of potable water. These disturbances decrease water quality, potentially increasing treatment costs for producing drinking water. We investigated the land use influence on raw surface water quality for drinking supply in the most populous state in Brazil (São Paulo). We analyzed a long-term dataset (2000–2014) to assess the raw water quality for drinking purposes in more than 50 river/stream sites located in predominantly industrial/urban, agriculture or forested watersheds. Temporal, seasonal and land cover influences were assessed through Analysis of Covariance (ANCOVA). Infrastructure and operating costs of the Water Treatment Plants (WTPs) were estimated. We found significantly better water quality ($p < 0.01$, ANCOVA) in forested watersheds (e.g., mean turbidity and true color: 10 NTU and 14 mgPt/L versus 48–56 NTU and 38–49 mgPt/L in industrial/urban and agriculture areas). THMPF and TOC were greater in altered basins (304 µg/L and 5.8 mg/L, respectively), especially in the rainy season. Drinking water outputs varied across WTPs located in industrial/urban (average: 525 L/s), agriculture (392 L/s) and forest (167 L/s) watersheds. Infrastructure and operating costs, shown as cost per unit of capacity, were greater for plants located in forested areas (US$ 98,208 per L/s), as compared to industrial/urban (US$ 78,230 per L/s) and agriculture (US$ 75,332 per L/s) basins. This was probably because treatment costs in these locations were greater in spite of plants starting with higher quality water. We recommend upgrades in the existing WTPs and the construction of new water utilities consider treatment costs related to predicted changes in land use in supply watersheds. Watershed conservation practices, water demand management, remediation and adaptation actions could also lower treatment costs and alter optimal treatment technology choice.

1. Introduction

Provision of drinking water is a vital ecosystem service that is compromised worldwide by human activities (Dodds et al., 2013). Drinking water is essential for human survival and development of social and economic activities (Shannon et al., 2008; Chenoweth, 2008). Water quality governance should integrate political, economic, social and environmental dimensions to assure technology improvement and compliance with drinking water guidelines. Water companies need to interact with consumers and different stakeholders to promote efficient communication and share relevant monitoring data (Dietrich et al., 2014; Kayser et al., 2015). Within this framework, however, there is only modest understanding of linkages of water treatment costs and factors that influence water quality in watersheds.

A multiple-barrier approach has been suggested to ensure clean water supply to people. Every step in the chain of water supply is considered, from supply watershed to treatment and distribution (WHO, 2004). Under such approach to provide the best water for human consumption, technology selection is also an important factor. Effectiveness of water treatment processes (e.g., coagulation, flocculation, sedimentation, flotation, filtration and disinfection) used in Water Treatment Plants (WTPs) depends on raw water quality characteristics, water demand and its temporal variations, funding availability, technical capacity and development. In Brazil and Latin America in general, true color and turbidity are important variables for guiding water treatment technology selection and

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http://dx.doi.org/10.1016/j.ecoleng.2016.06.063
0925-8574/

defining operational aspects in local WTPs (Di Bernardo and Sabogal Paz, 2008).

Urbanization and agriculture (Yu et al., 2014), wildfire and climate change (Delpla et al., 2009; Emelko et al., 2011), emerging pollutants (Simazaki et al., 2015), waterborne parasites (Burnet et al., 2014) and disinfection byproduct (DBP) formation (Goslan et al., 2009) in source and finished water can play a negative role in drinking water security, increase treatment costs and decrease human health. Ecosystem services have value to humanity (Liquete et al., 2011; Remme et al., 2015), and water source protection at the watershed level (e.g., riparian vegetation restoration) has been linked to decreasing treatment costs in WTPs (Honey-Rosés et al., 2013).

Forest cover can diminish water supply costs due to lower or no use of organic/inorganic fertilizers and pesticides, reduction in erosion processes and sediment inputs and overall benefits to water quality resulting from the filtering or buffering capacity from riparian areas and wetlands (Ernst et al., 2004; Abildtrup et al., 2013). Land use influence on drinking water intakes is relatively well known in temperate regions (e.g., Lalancette et al., 2014). However, less information is available for tropical and subtropical regions, including Brazil, where tropical climate, rapid urban and agriculture conversion rates and low levels of wastewater treatment might influence local water sources differently than in high-latitude areas. As we develop a biome-specific view of stream ecology and function (Dodds et al., 2015), a biome-specific view of human impacts on water quality and quantity could be warranted as well.

Dissolved organic matter inputs to water sources can be influenced by anthropogenic activities (Oh et al., 2013), seasonality and climate change (Ritson et al., 2014). Increased organic matter can be detrimental because DBPs can be formed after reactions between the disinfection agent and the organic matter. Recent research sought to find alternatives to remove organic matter from treated water (e.g., Phetrak et al., 2014) to avoid the potential negative consequences to human health, especially those related to carcinogenicity (Grellier et al., 2015). Trihalomethanes (THMs) are among the most important DBPs; their formation rate varies with time and on operational conditions in WTPs (e.g., pH, temperature, disinfectant dosage and contact time, Krasner et al., 2006; Ramavandi et al., 2015).

One of the challenges of the WTPs is to remove algae. Taste and odor are not desirable in drinking water and potentially toxic cyanobacteria are an additional issue (Zamyadi et al., 2012). Moreover, algal-derived organic matter might contribute to DBPs formation after chlorination (Hong et al., 2008). The increasing trophic state of freshwater sources in many countries has led to a growing interest in alternatives for algae removal from the water (Shen et al., 2011).

The São Paulo State in Brazil has more than 40 million inhabitants and they create significant anthropogenic pressure on local aquatic systems, leading to high nitrogen and phosphorus concentrations in surface waters and degradation of water quality (Cunha et al., 2011). São Paulo is responsible for about 24% of total water demand in Brazilian urban areas and around half of the municipalities in the state rely upon surface water bodies (rivers, streams, lakes and reservoirs) or combined surface water and groundwater sources for supply (ANA, 2010).

We aimed to investigate the land use influence on raw surface water quality for drinking supply in São Paulo State (Brazil), with emphasis on infrastructure/operating costs and trihalomethane potential formation (THMPF). We used a 15-year dataset on turbidity, THMPF, true color, total organic carbon (TOC) and chlorophyll-*a* to assess the water quality for drinking purposes in more than 50 river/stream sites located in predominantly industrial/urban, agriculture or forested watersheds. Seasonal influences on water quality were also addressed by comparisons among dry and rainy months. We estimated water treatment costs and compared to the respective turbidity and color values in raw water to look for correlations among such variables and land cover. Water clarification is still a challenge for many Latin America WTPs, making turbidity and color important variables for estimating costs. The relationships among THMPF, TOC, chlorophyll-*a* and the operational parameters from each treatment plant (e.g., treated discharge and technology aspects) were examined to estimate risks of disinfection byproducts formation across different land use types.

2. Material and methods

We compiled data from sampling sites monitored by the São Paulo State Environmental Agency – CETESB (Companhia Ambiental do Estado de São Paulo) from 2000 to 2014 every two or three months. The spreadsheet with the dataset is available as online supplementary material. All samples were collected and analyzed by CETESB using Standard Methods (APHA, 1998, 2005,

Table 1
Main land use, name, population density and percentage of forested areas of the Water Resources Management Units from where data from 54 sampling sites monitored by the São Paulo State Environmental Agency were analyzed. Partial and total number of sites and available data are shown for each case.

Main land use in the Water Resources Management Unit	Name of the Water Resources Management Unit	Population density (inhabitants/km^2)	Forested area (%)	Number of sites with data available
Industrial/urban (total number of data: 8693)	Paraíba do Sul	138	31	8
	Piracicaba, Capivari e Jundiaí	355	7	20
	Alto Tietê	3342	1	3
	Baixada Santista	592	74	2
	Mogi-Guaçu	97	11	2
	Sorocaba e Médio Tietê	154	12	5
	Tietê-Jacaré	128	7	1
	Mean or total[a]	*210*	*20*	*41*
Agriculture (total number of data: 1546)	Tietê-Batalha	39	6	1
	Médio-Paranapanema	41	11	2
	Baixo Tietê	49	3	3
	Aguapeí	28	5	1
	Peixe	43	4	1
	Mean or total[a]	*40*	*6*	*8*
Forest (total number of data: 684)	Litoral Norte	140	82	3
	Ribeira de Iguape	22	67	1
	Alto Paranapanema	33	14	1
	Mean or total[a]	*65*	*54*	*5*

[a] Total for the population density (inhabitants/km^2) and for the number of sites with data available; Mean for the forested area (%).

Table 2
Infrastructure costs according to USEPA (2001) updated for November 2012.

Conventional Treatment Plant
Equation if design capacity is less than or equal to 1 MGD:
$C = 1.566333333x \left[e^{\left(14.444+0.537^2/2\right)} D^{0.593} \right]$
Equation if design capacity is greater than 1 MGD:
$C = 1.566333333x \left[e^{\left(14.444+0.537^2/2\right)} D^{0.881} \right]$
C = the project cost for November 2012 (US$) for complete conventional plant with flocculation, sedimentation, filtration, waste handling and the building. The equations, with R-squared of 0.89, were outcome of the research of 144 WTPs real between small, medium and large.
D = design capacity in millions of gallons per day (MGD) and 1 MGD = 43.8126 L/s
e = the base of natural logarithms
Direct Filtration Plant
Equation if design capacity:
$C = 1.566333333x \left[e^{\left(14.472+0.575^2/2\right)} D^{0.716} \right]$
C = the project cost for November 2012 (US$) for complete direct filtration plant, including the building. Also the equation can be used for pressure filtration systems. Includes all raw water pumps, chemicals and mixing, unit processes, clearwell, waste handling and process control system. The Equations, with R-squared of 0.79, were outcome of the research of 28 WTPs real between small, medium and large.
D = design capacity in millions of gallons per day (MGD) and 1 MGD = 43.8126 L/s
e = the base of natural logarithms

2012). Sites were located in 40 rivers or streams immediately upstream the water withdrawals of 54 WTPs. The sampling sites were located within Water Resources Management Units (WRMUs) with different land use, the sites were located in industrial/urban areas (76%, Table 1) followed by agriculture (15%) and forest (9%) cover. We did not weight the data by actual areal cover, rather used all available sites with data. Industrial/urban WRMUs had higher population densities (97–3342 inhabitants/km^2) in comparison to agriculture (28–49 inhabitants/km^2) and forested ones (22–140 inhabitants/km^2). Forested WRMUs had significant percentages of native or second-growth vegetation (maximum of 82%, Table 1). The total number of samples analyzed for each land use category was 8693 (industrial/urban), 1546 (agriculture) and 684 (forest).

We analyzed differences in turbidity (NTU), THMPF (μg/L), true color (mgPt/L), TOC (mg/L) and chlorophyll-*a* (μg/L) values within studied rivers/streams. Data were divided in two sub-datasets for each land use category, separating months of higher and lower precipitation, typically from April-September (dry season) and October-March (rainy) for most of the cities in the São Paulo State. We ran Analysis of Covariance (ANCOVA) with date as a continuous variable and dry or rainy season and land use as categorical predictors to test for temporal, seasonal and land cover effects. We considered a 95% level to indicate statistical significance ($p < 0.05$). Statistica (v10.0, Statsoft Tula OK) was used to run these analyses.

For each of the WTPs receiving the water from the 54 studied sites, we sought information on drinking water production (L/s) and water treatment technologies employed (e.g., conventional treatment or direct filtration). Such information was obtained from the websites of the Water Supply and Sanitation Companies and city halls. The infrastructure costs were calculated according to USEPA (2001) equations (Table 2) for conventional treatment and direct filtration. The costs for flotofiltration plants were estimated in consonance with the prices for Itanhaém WTP, São Paulo, Brazil (cost = US$127,080 per each L/s in November 2012), according to the local water company. The infrastructure costs were updated using the Construction Cost Index (CCI) published in the Engineering News-Record (ENR). Non-Brazilian equations and indexes above were used to provide the estimated costs of WTPs because there is limited information available about the subject in Brazil.

The annual operating costs (chemicals and electric power consumption) were extracted from Sabogal Paz (2010) (Table 3), and updated for November 2012 using the General Market Price Index published in the Central Bank of Brazil. The present value of infrastructure and operating costs was calculated for each WTP and afterward it was expressed in US$ per each L/s for assessment purposes. The rate applied was 12% per year with 20 years of project period.

3. Results

Mean turbidity in forested areas (10 NTU) was approximately 5–6 times lower as compared to rivers/streams located in predominantly industrial/urban (48 NTU) and agriculture (56 NTU) watersheds (Fig. 1) (Table 4). The 75th percentile was the same for these two latter cases, 82 NTU (versus 18 NTU for forest). Mean true color also varied across the different sampling sites (38, 49 and 14 mgPt/L in industrial/urban, agriculture and forest areas, respectively). Land use and season significantly affected turbidity and true color ($p < 0.01$) (Table 5), with higher values observed during the rainy months.

The THMPF was affected by land use and season (Table 5). The highest values were observed in industrial/urban and agriculture sites (average of 304 μg/L for both) when compared to forest areas (231 μg/L) (Fig. 2), reflecting a risk of formation of such compounds ~24% lower if the raw water came from rivers or streams located in areas with more native remnant or recovered vegetation. The THMPF mean values were about 78%, 84% and 47% greater in the rainy months than in the dry ones, respectively for each land use. Mean TOC concentrations were 5.1 mg/L (industrial/urban), 5.8 mg/L (agriculture) and 2.4 mg/L (forest). Besides land use (TOC was twofold lower in forested basins) and seasonal influence (TOC was greater in the wet season), the ANCOVA also suggested a temporal trend for this variable ($p < 0.01$, Table 5).

Chlorophyll-*a* concentrations in the water column were relatively low (mean concentrations below 6 μg/L for all land uses) (Fig. 3) with no seasonal variation ($p = 0.630$, Table 5). The lowest chlorophyll-*a* concentrations were found in the forest basins (minimum: < 0.1 μg/L; maximum: 3.2 μg/L). The pigment concentrations were higher in the agricultural areas, but most cases were <10 μg/L. For all studied variables, land use played an important role in the water quality variation, with agriculture and industrial/urban basins with higher mean turbidity, THMPF, true color, TOC and chlorophyll-*a*.

Most of the analyzed WTPs used conventional technologies for water treatment (coagulation, flocculation, sedimentation or flotation, filtration and disinfection), followed by flotofiltration and direct filtration (coagulation, flocculation-optional, filtration and disinfection) alternatives (Table 6). Drinking water production varied across WTPs located in industrial/urban (average: 525 L/s; range: 40–3500 L/s), agriculture (392 L/s; 70–1075 L/s) and forest (167 L/s; 37–600 L/s) watersheds. Infrastructure and operating costs, shown as cost per unit of water production capacity, were greater for plants located in forested areas (US$ 98,208 per L/s), as compared to plants within industrial/urban and agriculture basins, which had similar costs (US$ 78,230 and US$ 75,332 per L/s, respectively) (Table 6). Notwithstanding, the present value in forested areas was lower (US$ 3.6 to US$ 84.4 mi). Regardless the land use, when compared to the operating costs per year (US$ 0.1 to US$ 6.3 mi), the infrastructure costs were higher because the treatment units, waste handling and building were considered (US$ 3.1 to US$

Table 3
Annual operating costs for Water Treatment Plants as reported by Sabogal Paz (2010) updated for November 2012.

Equation if design capacity is less than or equal to 100 L/s	
Electric power consumption (US$/year)	
Conventional Treatment Plant	C = {[-0.172D^2 + 66.262D + 1076.2]*0.58*12}/2.0312
Direct Filtration Plant	C = {[40,248D + 2004,1]*0.58*12}/2.0312
Floto-filtration Plant	C = {[-0.1402D^2 + 61.947D + 1199.2] *0.58*12}/2.0312
Sodium hypochlorite for available chlorine 12% and density 1.2 kg/L (US$/year)	
Conventional Treatment Plant	C = {[66D]*1.58*12}/2.0312
Direct Filtration Plant	C = {[0.008D^2 + 37.921D + 30.741]*1.58*12}/2.0312
Floto-filtration Plant	C = {[66D]*1.58*12}/2.0312
Aluminum sulfate for available Al_2O_3 of 7.5–8% and density of 1.3 kg/L (US$/year)	
Conventional Treatment Plant	C = {[90D]*1.86*12}/2.0312
Direct Filtration Plant	C = {[0.008D^2 + 49.921D + 30.741]*1.86*12}/2.0312
Floto-filtration Plant	C = {[90D]*1.86*12}/2.0312
Equation if design capacity is greater than 100 L/s	
Electric power consumption (US$/year)	
Conventional Treatment Plant	C = 205D
Direct Filtration Plant	C = 207D
Floto-filtration Plant	C = 205D
Sodium hypochlorite for available chlorine 12% and density 1.2 kg/L (US$/year)	
Conventional Treatment Plant	C = 616D
Direct Filtration Plant	C = 364D
Floto-filtration Plant	C = 616D
Aluminum sulfate for available Al_2O_3 of 7.5–8% and density of 1.3 kg/L (US$/year)	
Conventional Treatment Plant	C = 989D
Direct Filtration Plant	C = 561D
Floto-filtration Plant	C = 989D
C = annual cost for November 2012 (US$) D = design capacity in L/s	

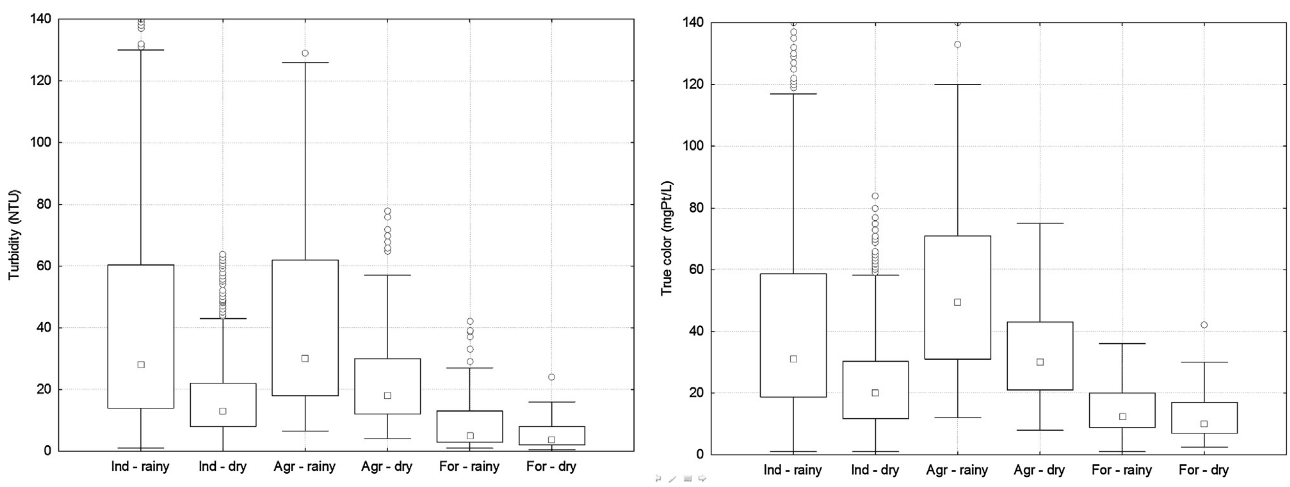

Fig. 1. Boxplots for turbidity (NTU) and true color (mgPt/L) in the studied rivers/streams according to seasonal period (rainy or dry) and their location in areas with different land use: industrial/urban (Ind), agriculture (Agr) or forest (For). Minimum, median, maximum and outlier values are shown for each case, as well as the lower (25%) and upper quartiles (75%).

203.3 mi) with the present value linked to project period and the rate applied (Table 3).

4. Discussion

4.1. Land uses and water quality

Water turbidity is associated with higher risk of pathogens (Khan et al., 2013) and can compromise disinfection efficiency in treatment plants. Intensive agriculture practices are associated with high microbial concentrations (Schreiber et al., 2015) and input of particulate materials through runoff. Agricultural catchments in Brazil, and specifically in the São Paulo State, are frequently affected by erosion processes. This has been leading to high rates of soil loss (800 million t^3/y in Brazil, Merten and Minella, 2013) and sediment inputs to nearby aquatic ecosystems. Organic farming (Barataud et al., 2014) with strict consideration of the core principles of organic agriculture (Silva et al., 2015), soil protection and agricultural "best management practices" (e.g., parallel terraces) (Strauch et al., 2013) could minimize suspended solids inputs to waters.

Urban land cover also frequently correlates with increased loads of suspended solids and overall decrease in water quality (Mouri et al., 2011). Surface runoff was strongly related with higher turbidity values during the rainy season in the studied rivers and streams, which were about 2–3 times lower in the dry period within the industrial/urban and agriculture catchments (Table 4). For forested areas, mean turbidity was similar across seasons (13 and 7 NTU). Extreme weather events (e.g., severe droughts, rainfalls and floods) can cause impacts to the water supply system and mitigation strategies are necessary to avoid water quality decrease and damages to public health (Khan et al., 2015). Our results suggested that the forest land use is probably less vulnerable to such events.

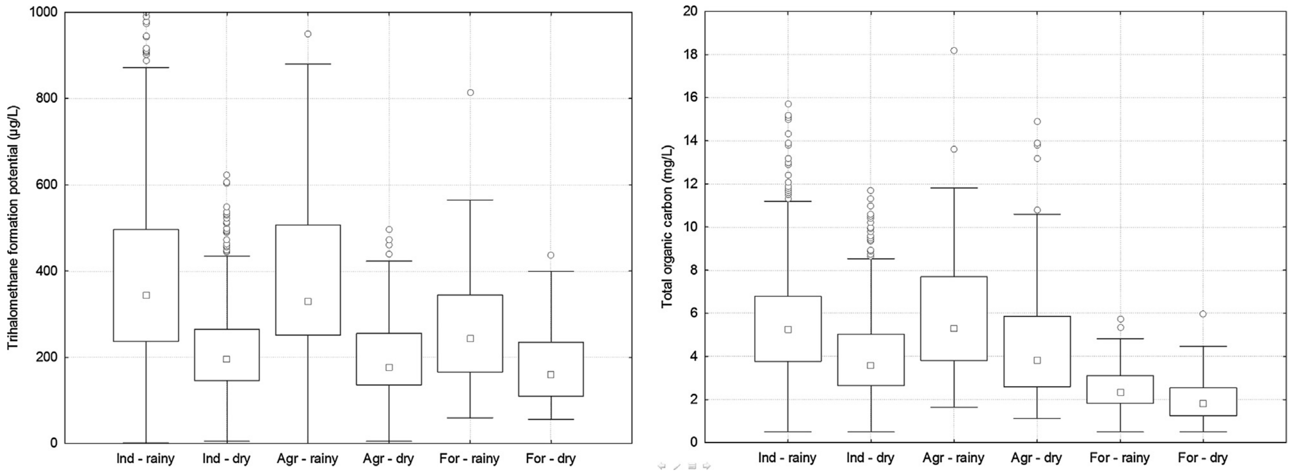

Fig. 2. Boxplots for trihalometane formation potential (μg/L) and total organic carbon (mg/L) in the studied rivers/streams according to seasonal period (rainy or dry) and their location in areas with different land use: industrial/urban (Ind), agriculture (Agr) or forest (For). Minimum, median, maximum and outlier values are shown for each case, as well as the lower (25%) and upper quartiles (75%).

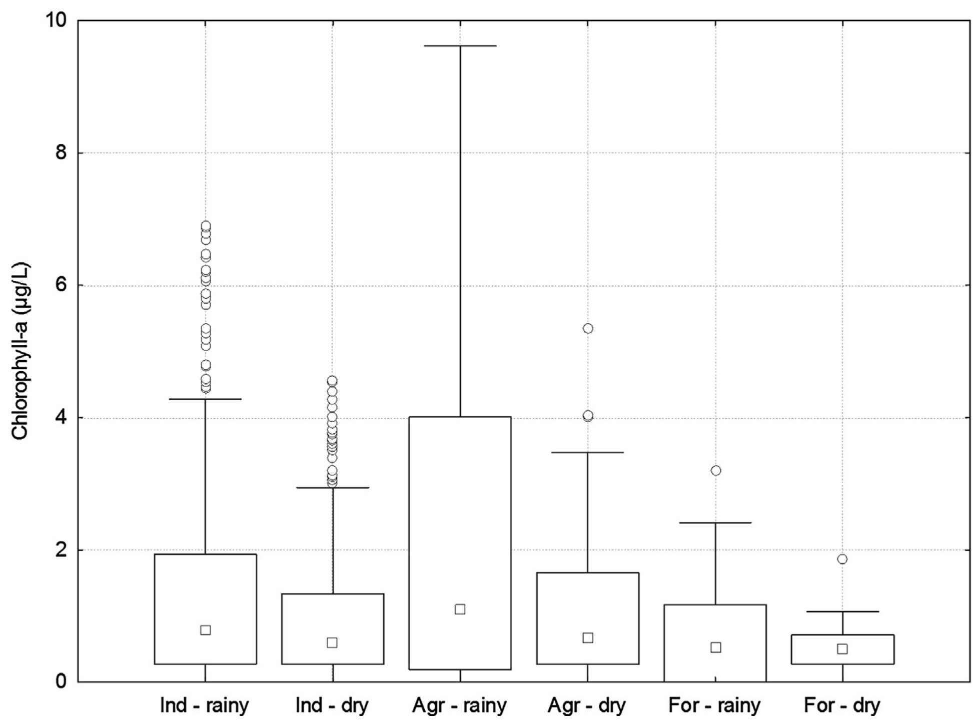

Fig. 3. Boxplots for chlorophyll-*a* (μg/L) in the studied rivers/streams according to seasonal period (rainy or dry) and their location in areas with different land use: industrial/urban (Ind), agriculture (Agr) or forest (For). Minimum, median, maximum and outlier values are shown for each case, as well as the lower (25%) and upper quartiles (75%).

Many tropical forest areas are fragmented and now have secondary vegetation mixed with native forest and other uses. Depending on the successional phase or recovery stage of its vegetation, the forest plays different roles in improving water quality. For example, old forests are usually more effective than impacted ones for decreasing water turbidity (Singh and Mishra, 2014). We did not control for age and maturity of forests, so we cannot see this effect in our data. Still, regardless of forest age, they were responsible for decreasing mean water turbidity in up to six times in comparison to other land uses (Table 4). These results reinforced that forest disturbance might decrease water transparency in tropical streams, as exhaustively described for temperate aquatic systems (e.g., Seilheimer et al., 2013).

TOC sources include natural organic matter from both autochthonous and allochthonous processes, which are affected by soil acidification, global warming and more extreme rainfall and drought events (Fabris et al., 2008; Shafiquzzaman et al., 2014). Anthropogenic inputs of organic matter are related to point and nonpoint sources (e.g., runoff) of pollution. The main human-related sources of TOC within the studied industrial/urban and agriculture watersheds were probably associated with domestic and industrial effluents discharge into local water bodies. The São Paulo State still had low percentages of sewage treatment (61%) and 10% of the wastewater was not even collected in 2014 (CETESB, 2014). The sewage treatment coverage varied across the analyzed WRMUs (Table 1), with average percentages of 63% (industrial/urban), 95% (agriculture) and 82% (forest). However, due

Table 4
Unweighted means, standard errors and number of data points available for turbidity (NTU), trihalomethane potential formation (μg/L), true color (mgPt/L), total organic carbon (mg/L) and chlorophyll-*a* (μg/L)in the studied rivers/streams considering the factors main land use (industrial/urban, agriculture or forest) and season (rainy or dry periods).

Variable	Land use	Season	Value or concentration (mean)	Value or concentration (standard error)	Data points
Turbidity (NTU)	Ind	Rainy	74	4	1439
		Dry	23	4	1482
	Agr	Rainy	78	11	230
		Dry	35	11	234
	For	Rainy	13	16	117
		Dry	7	15	123
Trihalomethane potential formation (μg/L)	Ind	Rainy	390	6	807
		Dry	219	6	811
	Agr	Rainy	393	13	174
		Dry	213	13	173
	For	Rainy	277	21	73
		Dry	187	10	77
True color (mgPt/L)	Ind	Rainy	47	2	682
		Dry	28	2	682
	Agr	Rainy	59	5	90
		Dry	38	5	90
	For	Rainy	15	5	80
		Dry	12	5	85
Total organic carbon (mg/L)	Ind	Rainy	5.7	0.1	549
		Dry	4.4	0.1	573
	Agr	Rainy	6.4	0.3	145
		Dry	5.2	0.3	139
	For	Rainy	2.6	0.5	41
		Dry	2.2	0.5	10
Chlorophyll-*a* (μg/L)	Ind	Rainy	1.9	0.3	828
		Dry	1.3	0.3	834
	Agr	Rainy	4.7	0.8	130
		Dry	7.0	0.8	141
	For	Rainy	0.9	1.8	26
		Dry	0.6	2.0	22

Ind: industrial/urban; Agr: agriculture; For: forest.

Table 5
Analysis of Covariance of the water variables (turbidity, trihalometane potential formation, true color, total organic carbon and chlorophyll-*a*) with date as continuous variable anddry or rainy season and land use as categorical variables. Correlations marked in bold are significant at a 99% confidence level ($p < 0.01$).

Variable	Factor			
	Land use	Season*	Time	Land use and season
Turbidity	**p < 0.010**	**p < 0.010**	p = 0.189	p = 0.127
Trihalomethane potential formation	**p < 0.010**	**p < 0.010**	p = 0.266	**p < 0.010**
True color	**p < 0.010**	**p < 0.010**	p = 0.055	p = 0.086
Total organic carbon	**p < 0.010**	**p < 0.010**	**p < 0.010**	p = 0.497
Chlorophyll-*a*	**p < 0.010**	p = 0.630	p = 0.582	p = 0.065

Table 6
Technology type used in the Water Treatment Plants in the São Paulo State according to the main land use in the Water Resources Management Unit (industrial/urban, agriculture or forest). Mean (minimum-maximum) values for drinking water production (L/s) are presented and costs are shown for each case, including infrastructure costs (million US$), operating costs (million US$ per year), the infrastructure and operating costs at present value (million US$ and US$ per L/s). Operating costs include chemicals and electric power consumption. For estimating the present values, a project period of 20 years and a rate of 12% per year were considered.

Main land use in the Water Resources Management Unit	Technology type	Drinking water production (L/s)	Infrastructure costs (million US$)	Operating costs (million US$ per year)	Infrastructure and operating costs (present value, million US$)	Infrastructure and operating costs (US$ per L/s)
Industrial/urban	CV (98%) FF (2%)	525 (40–3500)	31.6 (3.2–203.3)	1.0 (0.1–6.3)	37.8 (3.8–225.0)	**78,230 (59,488–140,602)**
Agriculture	CV (100%)	392 (70–1075)	22.8 (5.1–56.9)	0.7 (0.1–1.9)	28.1 (6.1–71.4)	**75,332 (66,424–86,991)**
Forest	CV (60%) FF (20%) DF (20%)	167 (37–600)	18.7 (3.1–76.2)	0.3 (0.1–1.1)	20.9 (3.6–84.4)	**98,208 (72,781–140,602)**

CV: conventional treatment (coagulation, flocculation, sedimentation or flotation, filtration and disinfection); FF: flotofiltration; DF: direct filtration (coagulation, flocculation-optional, filtration and disinfection).

to the population difference (forest basins have less inhabitants), the non-treated biochemical oxygen demand loads to the receiving water bodies reached 959, 46 and 36 t/d, respectively, in the same year (CETESB, 2014).

Hongve et al. (2004) and Murshedet al. (2014) reported increased dissolved organic matter and color in the water following rainfall events and greater river flows. The formation of disinfection byproducts is frequently greater during warmer periods (e.g., spring and summer) (Serrano et al., 2015). We also found higher true color, TOC and THMPF during the rainy (and warmer) season (15–59 mgPt/L, 2.6–6.4 mg/L and 277–393 μg/L, respectively) as compared to the dry colder season (12–38 mgPt/L, 2.2–4.4 mg/L and 187–219 μg/L) for all land use types (Table 4).

Brooks et al. (2015) studied the dissolved organic carbon and trihalomethane formation potential at two UK catchments by comparing raw and treated water. These authors found relatively high THMPF in one the source water reservoirs (maximum of ~2000 μg/L in November 2004 and ~3400 μg/L in May 2005), which was associated to a fen contributing to increase the DOC concentrations. The use of coagulation-flocculation in their local water treatment plants was related to a significant DOC abatement and reduction in THMPF in finished water. Since most of our studied WTPs encompassed coagulation-flocculation steps, we also expect organic carbon removal.

Disinfection byproducts can present very short term temporal variability (e.g., daily variability, Guilherme and Rodriguez, 2015) and there is not always a correlation between dissolved organic matter and disinfection byproduct formation potential (Li et al., 2011). However, in our case, we found a linear and positive relationship between median TOC and THMPF values across different land use sites ($r^2 = 0.63$; [median THMPF] = 40.96 x [median TOC] + 88.1), leading to an estimated potential of 129 μg/L of trihalomethanes per each mg/L of TOC. Brazilian Drinking Water Guidelines established an upper limit for total trihalomethanes (100 μg/L) (BRASIL, 2011). Therefore, the observed THMPF values are relatively high and pre-chlorination should be avoided in all local WTPs regardless the main land use in the basin. Alternative methods to chlorination and their efficiency for disinfection are available (e.g., Li et al., 2011).

If the water subject to chlorination has microcystin, there is a chance of formation of various undesirable byproducts following disinfection (Zong et al., 2015), whose toxicity is regulated by contact time with the disinfection agent. However, chlorophyll-*a* concentrations (i.e., a possible proxy for cyanobacteria) in the studied rivers/streams were relatively low most of the time as compared to reference values (Dodds et al., 1998) and did not vary significantly across seasons (Table 5). This possibly suggests a heterotrophic condition within these water bodies with a baseline chlorophyll-*a* concentration < 1 μg/L (corresponding to the forest catchments). Considering an alert level of 30 μg/L (Chorus and Bartram, 1999) for actions related to cyanobacterial blooms, the risk of microcystin production in the local water bodies was considered low.

4.2. *Costs associated with water treatment and water quality*

In general, lower turbidity in the raw water decreases the clarification costs in treatment plants. Considering water production as an important ecosystem service (e.g., Biao et al., 2010), forest cover is usually favorable for the economy of drinking water facilities (Fiquepron et al., 2013). Surprisingly, in our study, the costs we estimated for infrastructure and operation of the treatment plants (US$ per L/s) were higher for the studied WTPs located in forested areas (25–30% more expensive), even though they had more favorable water quality conditions (Table 4, Figs. 1–3); nevertheless, the present value for infrastructure and operating was lower (Table 6). According to Katko (1992), economies of scale do not necessarily apply in small water supplies systems where specific technical design and implementation can be used. In forested areas, evidently, there was no economy-of-scale. In this region, the small- and medium-sized WTPs installed since 1980 had used conventional design criteria of large systems that are not appropriate for the context. The result suggests no compliance of the technology applied in plants treating water from forest catchments with raw water characteristics, in a way that such plants did not take advantage of better water quality (e.g., lower turbidity) to decrease their overall costs.

Adjustments of the treatment processes could be more cost-effective if they consider raw water quality monitoring results (Hoyer and Schell, 1998). Our local WTPs at forested areas have been using conventional treatment technologies or flotofiltration (Table 6). Although our study analyzed only five variables of the raw water and we know there are many others within the drinking water guidelines, the results suggested that the WTPs at those catchments could consider using simpler technologies (e.g., direct filtration) in some cases. Coagulation and flocculation parameters (e.g., chemicals dosage, pH, times and velocity gradients) can also be optimized to improve treated water depending on raw water conditions (e.g., Venkatesh et al., 2015).

We recognize there is also a scale aspect for explaining the costs results. The water production in WTPs located in forested areas was the lowest (37–600 L/s, Table 6) because the population was smaller than the other land use areas. Declining costs are evident when the water utility size increases, especially when measured by the cost per volume produced (Katko, 1992; Jagals and Rietveld, 2011). However, limited economy-of-scale seemed to apply already to municipal water works serving less than 10,000 people. Moreover, the conventional design criteria used for large systems would lead to overcapacity in small systems. The results of our study corroborated the findings of Katko (1992), because robust and apparently unnecessary technologies were used in forested areas, despite the better raw water quality, increasing the average costs.

Table 6 shows the estimated costs of the WTPs evaluated; nonetheless, as reported by Montgomery (1985) the costs of water treatment plants vary widely due to variations in plant capacity, treatment process, design criteria, site and weather conditions, land costs, operations building characteristics, year and location of construction, competition among bidders and suppliers, and stability of the local and nationwide economic conditions. Therefore, the costs of Table 6 can vary dramatically depending on the context.

5. Conclusions

The access to clean water will become more and more a strategic issue worldwide in the next decades. Land use shifts and climate change can have a significant impact on drinking water quality and availability. Besides the importance of remediation initiatives and adaptation actions, our results reinforced the need for watershed conservation practices, with potential influences on human health and socioeconomic development. Demand management can also contribute for alleviating the pressure over water resources and maintaining ecosystem services. The traditional design criteria used for larger WTPs (e.g., the conventional treatment approach) may lead to overcapacity in small systems; therefore, the economy of scale can be achieved with appropriate technology selection and suitable organization for operation and maintenance.

The costs for water treatment vary with quality of water, and water quality varies with activities in the supply watershed. Here we demonstrate the influence of humans on various aspects of water quality in a subtropical region of a developing country. Our study was representative of the water treatment plants in Brazil and possibly reflects conditions in other tropical/subtropical areas. We demonstrate the importance of seasonal variation in the water chemistry across our 15-year long dataset under a wide spectrum of land use conditions. The relatively pristine forested watersheds we studied could be considered a starting point for reference conditions in future environmental studies.

Many of these water quality factors made worse by human impacts on watersheds should increase treatment costs. However, some factors were independent of land use. The trihalomethane potential formation was relatively high under all land uses and therefore pre-chlorination should be avoided in the local water treatment plants to prevent public health implications. In contrast, the industrial/urban and agriculture land uses negatively affected the water quality in relation to turbidity, true color, total organic carbon and chlorophyll-*a* which should increase treatment costs. This suggests that development of new and upgrading existing water treatment facilities should not only consider current conditions in supply watershed, but also potential future conditions. Cost of treatment is not only related to quality of incoming water. For example, the infrastructure and operating costs were up to 30% greater in the WTPs located in less impacted areas due to scale issues and compliance mismatch between technology and raw water characteristics.

We thus suggest that it could be possible to increase cost effectiveness by upgrading existing WTPs and selecting treatment technologies for new plants, in consonance with the raw water quality. This might contribute for saving financial resources, especially in developing countries like Brazil. While some of our cost figures are taken from the literature, and more research in this area is warranted, this paper indicates some of the steps planners might take as they consider options for increasing water treatment capacity in developing countries. These planners can then consider predicted trends of watershed use, management approaches such as watershed restoration and protection, against costs for various technologies for water treatment.

Acknowledgements

The authors are grateful to CETESB (Companhia Ambiental do Estado de São Paulo) for the dataset used in this paper, which is available at the website: https://servicos.cetesb.sp.gov.br/infoaguas. We thank FAPESP (Fundação de Amparo à Pesquisa do Estado de São Paulo) for the research grant to DGF. Cunha (Process Number 2014/02088-5) and for the financial support to LP. Sabogal-Paz (Process Number 2007/54608-9). We also thank CAPES (Coordenação de Aperfeiçoamento de Pessoal de Nível Superior) for the grant to WK. Dodds (Process Number 88881.068045/2014-01) as a visiting researcher in Brazil. Two anonymous reviewers provided valuable comments to improve the manuscript.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ecoleng.2016.06.063.

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