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Thi Ngoc Tuyet Nguyen

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THÈSE

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préparée au sein du **Institut des Géosciences de l'Environnement (IGE)** dans l'**École Doctorale TERRE ET UNIVERS, ENVIRONNEMENT**

Risk of eutrophication in the Saigon River: impacts of the Ho Chi Minh City Megacity (Vietnam)

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ABSTRACT

The Saigon River, a part of Saigon – Dongnai River basin located in Southern Vietnam, is about 250 km long with a catchment area of 4,717 km². The Dau Tieng reservoir in the upstream of the Saigon River has been constructed for irrigation, flood protection purposes and the control of the intrusion of saline water. When flowing through Ho Chi Minh City (HCMC), Saigon River is connected with canals and then joins Dongnai River to become the Nha Be River flowing through Can Gio Mangrove to the East Sea (also called South China Sea). Draining a low elevation coastal zone (LECZ), below 10 meters above sea level, Saigon River is affected by the asymmetric semi-diurnal tides.

The general goal of this thesis is to (i) assess trophic status in the Saigon – Dongnai River, (ii) assess the processes that drive nutrient behavior in the salinity gradient to estimate retention capacity with respect to the nutrients fluxes and (iii) calculate nutrient budgets.

The first step of this thesis consisted in a survey of nutrient concentrations (N, P, Si) at four sampling sites within the Saigon – Dongnai River system, which was carried out bi-monthly from July 2015 to December 2017, allowing to quantify the levels of nutrient concentrations and of indicators of eutrophication. This thesis pointed out an excess of nutrients in HCMC with concentrations of NH₄⁺ and PO₄³⁻ averaging to 0.7 mgN L⁻¹ and 0.07 mgP L⁻¹, respectively. We observed that untreated domestic discharges lead to the degradation of Saigon River's water quality with extreme value of algal biomass (up 150 µChl-a L⁻¹) and hypoxia conditions occurring episodically (Dissolved Oxygen < 2 mg L⁻¹) during dry season. The eutrophic issue in the city center has no clear effect downstream because eutrophic water mass from Saigon River is efficiently mixed with Dongnai and sea water masses during the semi-diurnal tidal cycles, leading to efficient metabolism of nutrients within the estuarine part.

Based on field and laboratory surveys, we assessed the eutrophication of the river, and investigated phosphorus (P) adsorption-desorption capacity onto suspended sediment (SS) within the salinity gradient. Field surveys showed a clear impact of the Ho Chi Minh megacity, total P increasing three fold in HCMC center, as compared with upstream values. Downstream, in the tidally mixed estuarine area, the total P lowered to less than 0.5 gP kg⁻¹. Laboratory experiments were carried out to characterize the influence of SS concentrations, salinity and turbulence on sorption/desorption processes. Among these observed variables, SS concentration was shown to

be the main driver for adsorption capacity of P onto SS in salinity gradient. This underlines the role of cohesive sediment dynamics, as an important driver of nutrient dynamics in this estuarine river system.

In this thesis, we analyzed the nutrients, suspended sediments and water discharges database from the Vietnamese Center of Monitoring of the Department of Natural Resources and Environment (period 2012 to 2016). Nine sites along the Saigon River and one site in the Dongnai River were used to identify the reference water status upstream of HCMC and the impact of the megacity in situ and downstream. The calculated fluxes allow drawing a first sediment and nutrients budgets at the scale of the Saigon – Dongnai Rivers and discussing the contribution of each sub basins to the total fluxes to the estuarine and coastal zones.

Keywords. Tropical river, Saigon – Dongnai River, eutrophication, nutrient budget, transformation, nitrogen, phosphorus, silica, suspended sediments.

RÉSUMÉ

La rivière Saigon, qui constitue un sous bassin du fleuve Nha Be, est située dans le sud du Vietnam. Cette rivière a une longueur d'environ 250 km et le bassin versant total à une surface de 4717 km². Le réservoir de Dau Tieng, situé en amont de la rivière Saigon, a été construit pour mieux gérer l'irrigation, lutter contre les inondations et contrôler l'intrusion d'eau salée. En traversant Ho Chi Minh Ville (HCMV), la rivière Saigon est connectée à plusieurs canaux puis rejoint la rivière Dongnai pour devenir le fleuve Nha Be qui traverse la zone de mangrove de Can Gio avant de se jeter dans la mer de l'Est. La rivière Saigon et son bassin versant sont situés dans la plaine côtière et sont soumis au balancement quotidien des marées.

L'objectif général de cette thèse est (i) d'évaluer l'état trophique de la rivière Saigon - Dongnai, (ii) d'étudier les processus à l'origine de la dynamique des nutriments, en particulier le rôle des sédiments en suspension, du mélange turbulent et du gradient de salinité, afin d'estimer la capacité de rétention par rapport aux flux d'éléments nutritifs; (ii) d'utiliser ces mesures de terrain et de laboratoire pour dresser des bilans et discuter les contributions de chaque source.

La première étape de cette thèse a consisté en une étude des concentrations en nutriments (N, P, Si) sur quatre sites d'échantillonnage dans le système de la rivière Saigon – Dongnai. Les mesures ont été réalisées deux fois par mois de juillet 2015 à décembre 2017 pour évaluer les niveaux de concentration en éléments nutritifs et l'état d'eutrophisation. Cette thèse a mis en évidence un excès de nutriments dans HCMV avec des concentrations de NH₄⁺ et de PO₄³⁻ atteignant en moyenne 0.7 mgN L⁻¹ et 0.07 mgP L⁻¹, respectivement. Nous avons observé que des rejets domestiques non traités entraînaient une dégradation de la qualité de l'eau de la rivière Saigon, avec une valeur extrême de biomasse algale (150 µChl-a L⁻¹) et des périodes d'hypoxie (Oxygène dissous < 2 mg L⁻¹), principalement en saison sèche. L'eutrophisation de la rivière dans la zone urbaine dense n'a pas d'effet clair en aval, car la masse d'eau eutrophe de la rivière Saigon est efficacement mélangée avec les masses d'eau de mer et de la rivière Dongnai pendant chaque cycle de marée. Ce qui met en évidence le rôle tampon (capacité de métabolisation) de l'estuaire entre les apports amont et la mer.

Sur la base d'enquêtes de terrain et d'expérience en laboratoire, nous avons évalué les causes de l'eutrophisation de la rivière et étudié la capacité d'adsorption-désorption du phosphore sur les sédiments en suspension dans le gradient de salinité. Les enquêtes sur le terrain ont montré un

impact clair de la mégapole, le P total augmentant de trois fois dans le centre de HCMV, par rapport aux valeurs en amont. En aval, dans la zone estuarienne mélangée par des marées, le P total descend à moins de 0.5 gP kg^{-1} . Les expériences en laboratoire ont montré le rôle clé de la concentration en sédiments cohésifs en suspension sur la capacité d'adsorption de P sur les particules dans le gradient de salinité.

Dans cette thèse, nous avons également analysé une base de données des nutriments, des sédiments en suspension et des débits de 2012 à 2016 (données du service de surveillance de la qualité de l'eau vietnamien de la province d'HCMV). Neuf sites le long de la rivière Saigon et un site dans la rivière Dongnai ont permis de déterminer l'état de référence à l'amont et à l'aval d'HCMV. Les flux calculés permettent de dresser un premier bilan sédimentaire et nutritif à l'échelle des fleuves Saigon – Dongnai, de discuter de la contribution de chaque sous bassin aux flux totaux et de dresser des hypothèses sur l'état environnemental actuel et futur des zones estuariennes et côtières.

Mots clés. Rivière tropicale, rivière Saigon - Dongnai, eutrophisation, bilan de nutriment, transformation, azote, phosphore, silice, sédiments en suspension.

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ABBREVIATION

ASEAN	Association of Southeast Asian Nations
HCMC	Ho Chi Minh City
DIC	Dissolved inorganic carbon
DIN	Dissolved inorganic nitrogen
DN	Dongnai
DOC	Dissolved organic carbon
ICEP	Indicator of coastal eutrophication potential
ITCZ	Inter Tropical Convergent Zone
N	Nitrogen
NBR	Nha Be River
NO ₂ ⁻	Nitrite
NO ₃ ⁻	Nitrate
NH ₄ ⁺	Ammonium
P	Phosphorus
PO ₄ ³⁻	Ortho-phosphate
PIC	Particulate inorganic carbon
PIN	Particulate inorganic nitrogen
PIP	Particulate inorganic phosphorus
PIP	Particulate inorganic phosphorus

POC	Particulate organic carbon
POP	Particulate organic phosphorus
SG	Saigon
Si	Silica
TOC	Total organic carbon
Total N	Total nitrogen
Total P	Total phosphorus
TPP	Total particulate phosphorus

INTRODUCTION

1. Background

In recent decades, many developing countries have been facing a significant challenges in link with the ever-growing appetite for fresh water and in the context of global warming. Industrialization, urbanization, agricultural intensification, and the construction of artificial reservoirs have modified, and frequently threatened the water supply, especially from surface waters which are sensitive to eutrophication. Whilst Asian countries are experiencing faster growth over the last decades, rapid industrialization and urbanization have led to quantitative shortage and qualitative deterioration of water resources. Water is a vital and vulnerable natural resource, on which the future development of almost all countries will depend. It is therefore essential to identify the major factors of the change in water quality, and to adapt in consequence, the management of water resources. Such modifications could successfully combine water management practices and processes (Vo, 2007).

Vietnam is an agricultural country with a high density of rivers network; of which 2360 rivers have a length of more than 10 km (Huynh et al., 2014). The total annual water discharge is 835 billion m³, of which 313 billion m³ are available for human use (Tran Ngoc et al., 2016). Water is plentiful in the wet season (from June to November) but serious shortages occur during the dry season (from December to May). Despite the abundant water resource, the dependency on upstream countries also makes water quality management more difficult in Vietnam (for instance on the large watersheds of Mekong River and Red River). There is an increasing contamination of Vietnam's surface, ground, and coastal waters. Although upstream river water quality remains good, the downstream part of major rivers exhibits poor water quality and most of the lakes and canals in municipal areas are fast becoming sewage sinks. Groundwater shows local pollution and saline intrusion. All these facts are existing in the Saigon River, in southern Vietnam.

Saigon River is the second largest river supplying water to Ho Chi Minh City (HCMC) after Dongnai River. Because of the urban sprawl in HCMC and neighboring provinces nearby Saigon River basin, water quality of this river is now at risk. In spite of the vital function, Saigon River water quality has received poor attention. Most of the national water quality monitoring programs

have been conducted only in the downstream basin, especially in HCMC (Cornelis et al., 2015; Downes and Rujner, 2010; Luong et al., 2013).

Nowadays, the effects of industrialization and urbanization on water quality are the main issues not only in Vietnam but also worldwide. Human activities have significantly changed Nitrogen (N), Phosphorus (P), Carbon (C) and Silica (Si) cycles; in which, three first elements control many functions of aquatic ecosystems. Many factors are affecting the nutrient balance into river-water column, such as hydrology, nutrient inputs from soil system and biogeochemical transformation processes of nutrients between sediment and the water column, from upstream to downstream. Additionally, there are a difference in N, P and Si inputs from diffuse and point sources into the drainage network. Their concentrations and fluxes can fluctuate significantly from upstream to downstream according to the variations of the hydrological conditions, land use, industrial and agricultural activities. These changes can alter ecosystem structures and functions (Billen et al., 2007).

One of the key, which is particularly useful for assessing coastal eutrophication status, is the concentration of nutrients which are carried to the coastal zone by rivers. Billen and Garnier (1997) showed that coastal eutrophication is the consequence of unbalanced riverine nutrients inputs, with an excess of nitrogen and phosphorus with respect to silica, compared to the requirements for diatoms growth. Five decades ago, Redfield et al. (1963) proposed the Redfield stoichiometric ratio of carbon, nitrogen and phosphorus (C:N:P) as a tool to evaluate eutrophic status of water body; stated to be C:Si:N:P = 106:15:16:1 (Brzezinski, 1985). It is very likely that non-siliceous algae would developed by the imbalance between these nutrients inputs. This type of algae has a serious impact on coastal eutrophication. Furthermore, sediments from watershed to estuary have played an essential role in the biogeochemical process of elements such as nitrogen, phosphorus, carbon, and silica. Some decades ago, Martin and Meybeck (1979) predicted that over 90% of the global riverine transfer of some major biogenic elements such as phosphorus was transported with suspended matter. And about 45% of total organic carbon discharged from rivers into the ocean was in particulate form annually.

Many researches have been carried out on the water quality of rivers around the world. However, few scientists focused their studies on emerging countries, especially in the very dynamic region of South East of Asia. This is actually the case for the water body of Saigon River in HCMC,

where the respective impacts of natural and anthropogenic pressures remain poorly characterized. With aforementioned gap, this study has been conducted to evaluate the concentration of nutrients and assess the status and driving factors of water quality of Saigon River and its main tributary, Dongnai River; and to contribute to a better management of the waterbodies for an efficient and sustainable development?

2. Scientific objectives

In order to understand the pollution levels of the hydrosystem, its dynamics and the effects on the ecosystem, a group of French and Vietnamese scientists has been working together during the last years, within the framework of a cooperation project CMIRA Saigon River Région Rhône Alpes (2016-2018) and EC2CO (2016-2017). As part of the RESCIF international network (www.rescif.net), the close collaboration between the Institut des Géosciences de l'Environnement (IGE, Grenoble-INP, IRD, Université Grenoble Alpes, CNRS, Grenoble France) and Ho Chi Minh University of Technology (HCMUT, Vietnam), led to the creation of the “Centre Asiatique de Recherche sur l'Eau (CARE)”, which focuses on the study of water quality and eutrophication risk assessment.

The overall objective of this thesis is to outline the water quality status of Saigon River and to provide the scientific understanding of natural and anthropogenic factors affecting Saigon River water quality. The specific objectives are:

- 1) Quantifying the different forms (dissolved or particulate inorganic and organic nutrients) and ratio of C:N:P:Si in the Saigon – Dongnai River basin to assess the trophic status and the risk of eutrophication.
- 2) Studying exchangeability of phosphorus between sediments and water body; and assessing phosphorus behaviors in suspended sediments within the salinity gradient in order to evaluate the estuarine retention capacity, with respect to the phosphorus fluxes delivered from the upstream watershed.
- 3) Preparing a comprehensive database of sources of pollution (wastewater from domestic, and industrial activities) for establishing nutrient and sediment budgets in the Saigon River.

3. Content of the manuscript

This manuscript comprises 5 chapters; three of them are under the form of scientific papers already published, submitted or in preparation.

The chapter 1 gathers the presentation of the study area providing the necessary background information about the Ho Chi Minh City, the Saigon – Dongnai River system and all the biogeochemical characteristics related to eutrophication risk.

Chapter 2 presents the study sites, the material and methods, the monitoring strategies, the experimental set-up and data treatment that were adopted for the study.

Chapter 3 is under the form of a scientific paper published in Science of the Total Environment. In this chapter, the results of water quality are presented based on bi-monthly monitoring program during period of two year and a half (from July 2015 to December 2017). The construction of a database on nutrient (N, P and Si), organic carbon and phytoplankton dynamics of the Saigon – Dongnai River system required for assessing eutrophic status is also introduced and discussed.

Chapter 4 reports the results of Phosphorus contamination observations in the Saigon River and Phosphorus transformation under different conditions of TSS concentrations, turbulence and salinity. These results, based on field measurement and laboratory experiments are used to discuss the effect of TSS, turbulence and salinity on the adsorption/desorption of Phosphorus onto suspended sediments, and the transformation of phosphorus within the estuary. This chapter is currently submitted for a publication in Estuarine, Coastal and Shelf Science.

Chapter 5 presents the budgets of water discharge, suspended sediment, and phosphate and P fluxes in six sampling sites of the Saigon – Dongnai River system, the input fluxes of atmospheric deposition, urban zone and industrial zone. It is based on the analysis of the database provided by the Center of Environmental Monitoring – DONRE (department of Natural Resources and Environment), as well as our own measurements database within the Saigon – Dongnai Rivers system. This chapter is designed to be possibly submitted the journal River Research and Applications.

The conclusions and perspectives section summarizes the main results obtained in this study and discusses some perspectives that could be addressed in the future for a better understanding of the

fate of nutrients in tropical estuaries, and for an improvement of local water management to restore water quality.

CHAPTER 1. LITERATURE REVIEW

1. Water quality and the impacts of megacities on aquatic ecosystems

1.1. Water quality in South East Asian Countries and in Vietnam

It took hundreds of thousands of years for the world population to grow to 1 billion – then in just another 200 years or so, it grew sevenfold. In 2011, the global population reached 7 billion, and today, it stands at about 7.6 billion (data from www.unfpa.org). 2007 was the first year in which more people lived in urban areas than in rural areas, and by 2050 about 66 % of the world population will be living in cities. This urbanization unfold in Africa and Asia (Figure 1-1), bringing huge social, economic and environmental transformations. Among all consequences of urbanization (health care, education, housing, sanitation, food, energy and water), the present work focuses on the water-related issues, and more especially on the deterioration of water quality and its impact on aquatic ecosystems.



Figure 1-1. Population distribution in the biggest cities in the world (source: <https://news.cnrs.fr/articles/ever-larger-cities>).

The study is conducted in a megacity of South East Asia (Ho Chi Minh City, Vietnam), of the ASEAN region (Association of Southeast Asian Nations, Figure 1-2). This region comprises ten

Southeast Asian countries that share common water related challenges and promote cooperation and facilitates economic, environmental, educational actions to fight against water pollution.

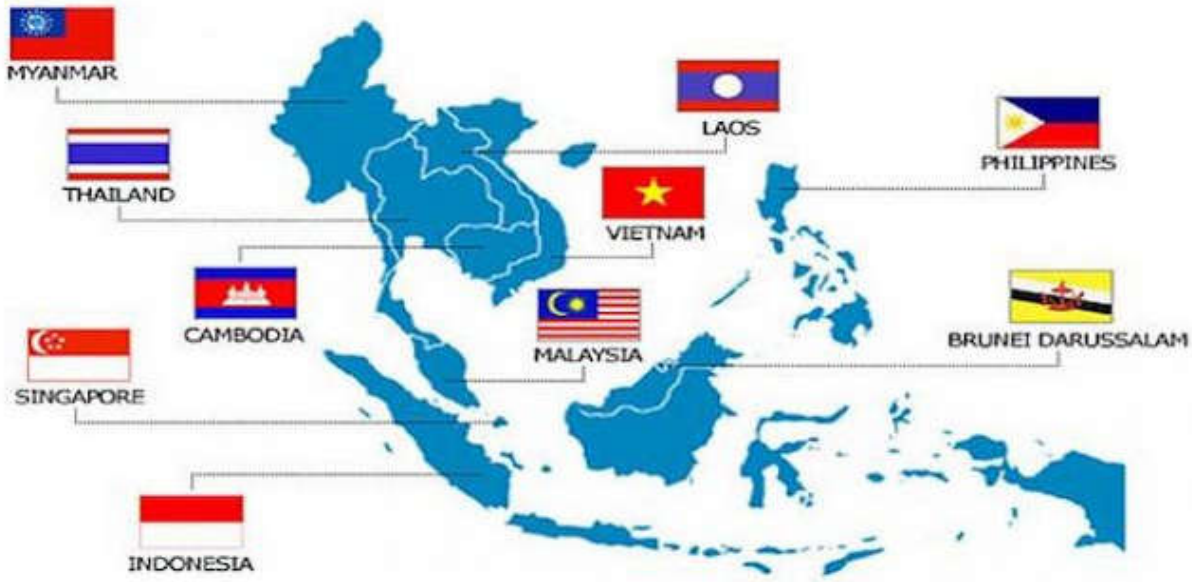


Figure 1-2. Map of the assembled countries considered (ASEAN countries: Indonesia, Malaysia, Philippines, Thailand, Vietnam, Lao, Cambodia, Singapore, Brunei and Myanmar; source: worldatlas.com).

Among the most serious problems of water quality in ASEAN figure: lack of wastewater treatment system and particles released through soil leaching; domestic wastes discharged directly to river body; water pollution from small-scale industries and agricultural activities or even groundwater contaminated with coliforms (sources: Water Environment Partnership in Asia – www.wepa-db.net, Alexandra et al., 2012; Stephen and Liz, 2009). The megacity of Ho Chi Minh city (Vietnam) gathers all these water-related issues and thus provides an interesting study case to evaluate the impact of megacities on the water quality and aquatic ecosystems, to anticipate scenario of evolution in the megacities of the ASEAN and to propose ways of mitigation. Some emblematic water quality issues in ASEAN are synthesised in Table 1-1.

Table 1-1. Water quality in ASEAN (source: Water Environment Partnership in Asia - wepa-db.net, Alexandra et al., 2012; Stephen and Liz, 2009).

Country	Water quality problems
Cambodia	Garbage and sewage; wastewater treatment; pesticides and chemical fertilizers (discharging highly amount of toxic waste into water column, carrying toxic out shrimp field); particles released during soil leaching and proper dumping places; illegal fishing.
Indonesia	56 % of household disposes their domestic waste directly to river body; sources of water pollution also come from small-scale industries, agriculture, textile, pulp and paper, petrochemical, mining, and oil and gas; level of mercury (Hg) concentration and BOD reach $2.78 \mu\text{g L}^{-1}$ and 100mg L^{-1} , respectively.
Laos	Waste and polluted water in urban areas, water pollution from agricultural and industrial activities, degradation of natural water and water catchments from sedimentation, erosion from land.
Malaysia	The water pollution from point sources (sewage treatment plants, manufacturing and agro-based industries and animal farms) and non-point sources (agricultural activities and surface water discharges).
Myanmar	The low percentage of people covers by safe drinking water and sanitation facilities.
Thailand	High loading of pollutants from human activities, pollutants discharged from industrial, agricultural or populated areas.
Philippines	58 % of groundwater was contaminated with coliforms and many cities were experiencing a shortage of water supply during the dry season. Nearly 2.2 million metric tons of organic pollution are produced annually by domestic, agricultural, and industrial sectors. Water pollution was dominated by domestic and industrial sources.

Vietnam	Poor water quality; lakes and canals becoming sewage sinks; pockets of contamination and some salinity intrusion in groundwater; rapid urbanization and industrialization and transport development, expansion, and an increase in the number of oil spills contributing to the deterioration of water quality.
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1.2. The megacities and their impacts on the environment in Vietnam, with a focus on HCMC

1.2.1. Characteristics of megacities

Csomos (2014) classified megacities based on two parameters: (i) the specific geographic regions (population distribution) and (ii) the economic performance (income per capita). The population growth is due to the positive birth/death ratio, the rural-to-urban and urban-to-suburb migration. Population growth and high population densities are driving the municipal sprawl. In ASEAN, radical spatial, demographic, social and political structural changes in urban areas took place over the last decades, associated with the economic rise of the whole region (Kraas, 2007). Megacities will get bigger in the coming years, as the population continues to grow and people continue to abandon rural areas. Some Southeast Asian cities are already some of the most populated metropolises in the world (Figure 1-1), with millions of inhabitants like Manila (Philippines, 11.9 million inhab.), Jakarta (Indonesia, 30.2 million inhab.), Bangkok (Thailand, 14.5 million inhab.), Kuala Lumpur (Malaysia, 6.6 million inhab.), Singapore (5.8 million inhab.), Yangon (Myanmar, 4.3 million inhab.) and Ho Chi Minh City (Vietnam, 8.4 million inhab.).

In Vietnam, annually, 100 000 hectares (or more) of agricultural land are recovered for construction of industrial parks, services, and transportations (Nguyen et al., 2017; Vo, 2007). In some rural areas, typically as in the Red River Delta and in the Mekong River, the population has migrated to big cities such as Hanoi and Ho Chi Minh City. This rural migration flow leads to a city expansion that witnessed very heavy overload on population, technical and social infrastructures (Decker et al., 2002). After reaching an urban proportion of the population of 10 % during the 1950's, the speed of urbanization increased, and was around 21.5 % in the 1970's. After the country was reunified, there was a decline in the urban proportion of the population throughout the country until 1982, when it had fallen to 18.4 %. Since then, the level of urbanization has increased gradually to overpass 35% in 2015 (Figure 1-3).

Aside from population size, megacities are very high consumers of materials and energy. Power generation and the energy consumption in megacities have enormous impacts on the environment that is unequally considered by policy makers. Developed and wealthy cities, such as Los Angeles, propose mitigation strategies and offer central water treatment plants and distribution facilities. They can support 100% of their population with fresh and clean water (Decker et al., 2002). This is generally not the case in ASEAN megacities.

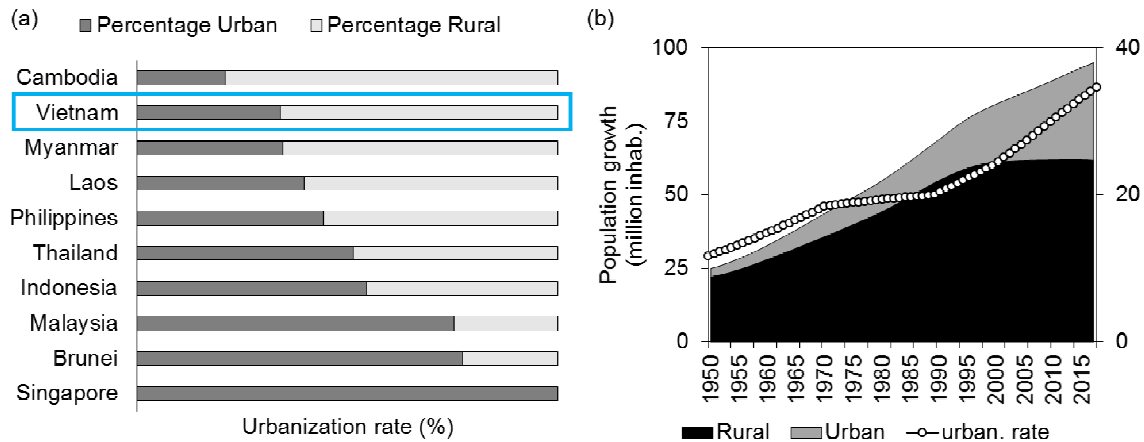


Figure 1-3. (a) Urbanization rate of ASEAN in 2016 and (b) Population growth in rural and urban area of Vietnam from 1950 to 2017 (source: FAOSTAT)

1.2.2. The effects of megacities on aquatic ecosystems

Urban development patterns are closely linked to an increase in the consumption of freshwater and, therefore, in an increase in the water demand for many activities. Among them, the food consumption and different nutrition patterns in megacities are higher than in rural populations. The urban development leads an increase in the food demand and a decrease in the availability of arable lands (Phdungsilp, 2006; Hawkes et al., 2017; Popkin, 2001).

Urban expansion or municipal sprawl with the construction of urban, tourism and transportation facilities is closely linked to the decrease in arable lands, degradation of biodiversity, air and water quality (Penning de Vries et al., 2003; Zhao et al., 2006). Untreated domestic wastewaters, which are released directly into aquatic ecosystems, remain one of the most common sources of pollution. In the case of Vietnamese megacities, canals and small water bodies are generally becoming a sink for domestic and industrial wastewaters, especially in HCMC and Hanoi. According to the Viet Nam Environment and Sustainable Development Institute, the megacities account for (only ~30

%) of the national population, but they generate more than 6 million tons of waste each year (nearly 50 % of the total amount of waste of the country) (Nguyen et al., 2017).

2. River systems in tropical area

Only 19 % of the land surface locate in tropical climate region; in which, 60 % in the South American, 31 % in Africa, 16.3 % in Asia, 8.3 % in Australia and 5.9 % in North America (Figure 1-4) (Peel et al., 2007). The most important control of the tropical climate relates to the position of the Inter Tropical Convergent Zone (ITCZ) (Syvitski et al., 2014). In tropical areas, there is a climate without strong winter since these regions receive a large amount of solar energy. The temperature is neither shift from day to night, nor from summer to winter. The tropical zones are characterized by the intense convective rainfall (Syvitski et al., 2014).

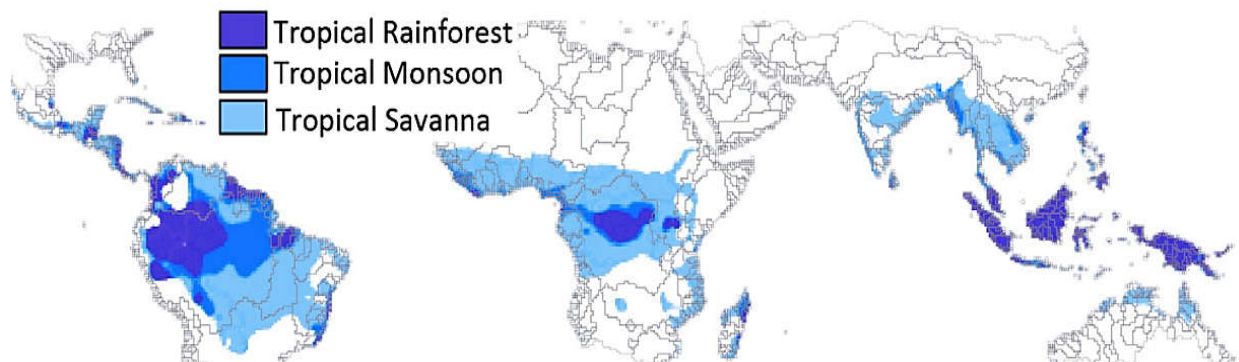


Figure 1-4. Tropical regions of the world superimposed on drainage basins (from Peel et al., 2007, reported in Syvitski et al., 2014).

In tropical regions, such as in Vietnam, the main factor, which specifies the seasons, is rainfall (Figure 1-5). Therefore, the amount and temporal distribution of the precipitations are the important elements for disparity in climatic zones; for instance, wet (> 1800 mm), wet-dry (700-1800 mm), and dry (< 700 mm) (Latrubesse et al., 2005). All rivers in Vietnam flows through wet-dry climatic regions, which is classified as a monsoonal regime. The alternation of ocean tropical and continental atmosphere masses dominates the seasons. As warm, moisture-laden air flows from the Indian Ocean in summer, a wet season develops. In winter, a high pressure system develops over the Asian continent and becomes the source of dry air masses.

The climate in the South of Vietnam, is controlled by “trade wind” monsoon in winter and South West monsoon in summer. The wet season lasts from May to October and the dry season is from

December to April. The wet season cumulates 85 – 90 % of the total annual rainfall (Figure 1-5). The climate of the South of Vietnam, confers the typical hydrologic regime characterized by large runoff during wet season and low runoff during dry season.

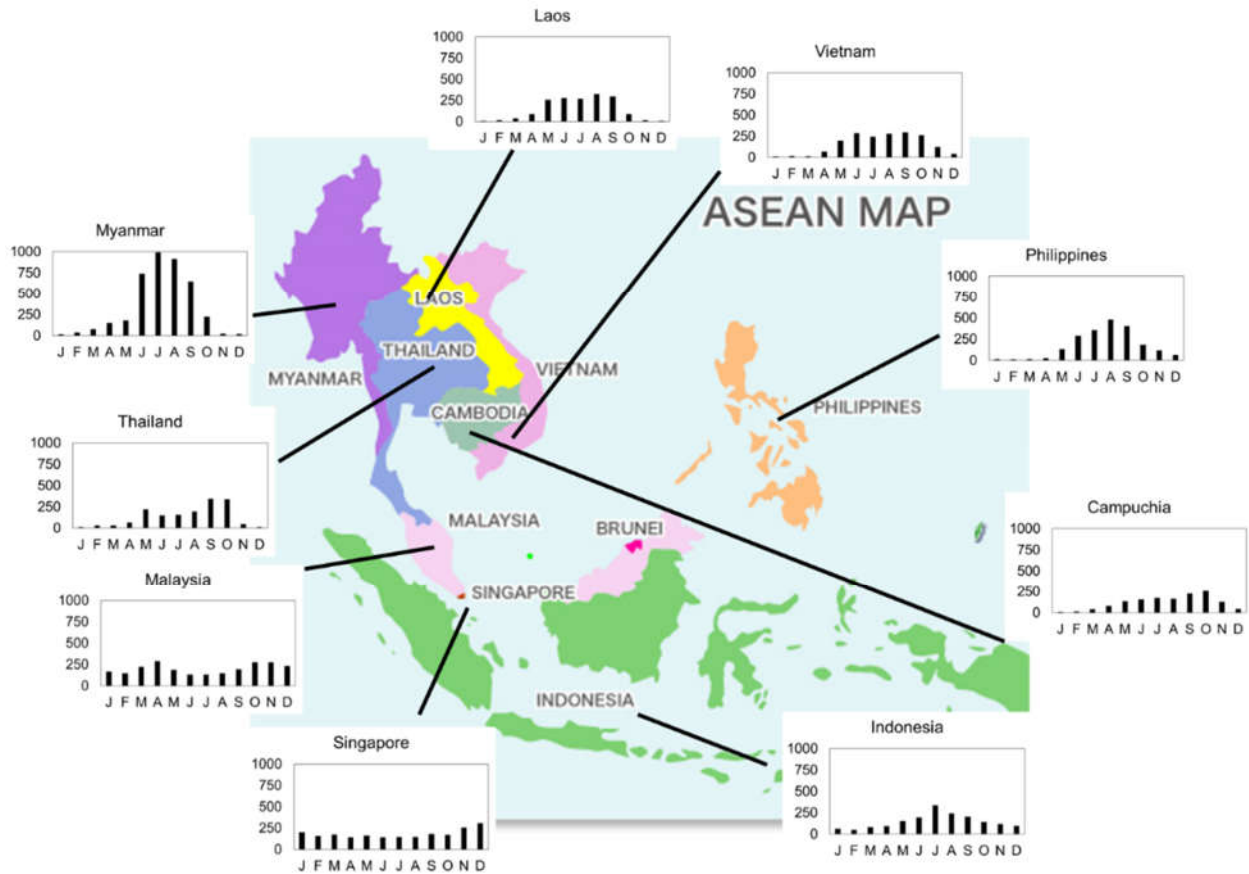


Figure 1-5. The distribution of rainfall (mm) in ASEAN region (source: climatemps.com).

3. Nutrients in tropical estuaries

Nutrients are key substances for organisms and life development. In our study, we focus on three key nutrients for the development of aquatic life in estuaries, namely nitrogen (N), phosphorus (P) and silica (S).

3.1. Nitrogen

Nitrogen (N) is essential for crop growth. Nitrogen which comes from synthetic fertilizers, has become the main anthropogenic inputs changing the N cycle, in comparison to N discharged from

urban wastewater treatment plants or nitrogen oxide emitted through motor vehicle traffic and the thermal electricity generation (Billen et al., 2013).

3.1.1. Nitrogen cycle

There is a numerous variety of nitrogen form in the environment such as organic nitrogen, ammonium (NH_4^+), nitrite (NO_2^-), nitrate (NO_3^-), nitrous oxide (N_2O), nitric oxide (NO) or inorganic nitrogen gas (N_2). Many processes, in which microbes can transform nitrogen from one form to another, both to get energy or to accumulate nitrogen in a form needed for their growth. The diagram below shows how these processes fit together to form the nitrogen cycle (Figure 1-6).

In biogeochemistry, there are three processes of nitrogen transformation.

- First, *fixation/assimilation*: Nitrogen in the atmosphere is fixed by plants. However, most fixations are carried out by symbiotic bacteria known as diazotrophs. Most biological nitrogen fixation takes place by the nitrogenase enzyme, found in a wide variety of bacteria (Moir, 2011).
- Secondly, *ammonification/anaerobic ammonia oxidation*: When organisms (plant or animal) die, bacteria or fungi transform organic nitrogen form back into NH_4^+ , a process called ammonification or mineralization. This process contributes to a main proportion of nitrogen conversion into aquatic ecosystems.
- At last, *nitrification/denitrification*: The nitrification process converts NH_4^+ to NO_3^- , which is easy to dissolve in water and is not retained in soils (Vitousek et al., 1997). In addition, the denitrification process is the reduction of NO_3^- back into the inert N_2 gas, completing the nitrogen cycle. This process is conducted by bacterial species such as *Pseudomonas* and *Clostridium* (Smil, 2000). Oxygen is required for nitrification but denitrification is anaerobic strict process.

In aquatic ecosystems, N enters to water column by many pathways, e.g. precipitation, soil runoff and leaching, domestic and industrial wastewater discharge. N cannot be used directly by phytoplankton under the form of N_2 gas, except for cyanobacteria (Charles and Wheeler, 2012) (Figure 1-6). Other phytoplankton species use N as NH_4^+ or NO_3^- in water environment to synthesis organic matter.

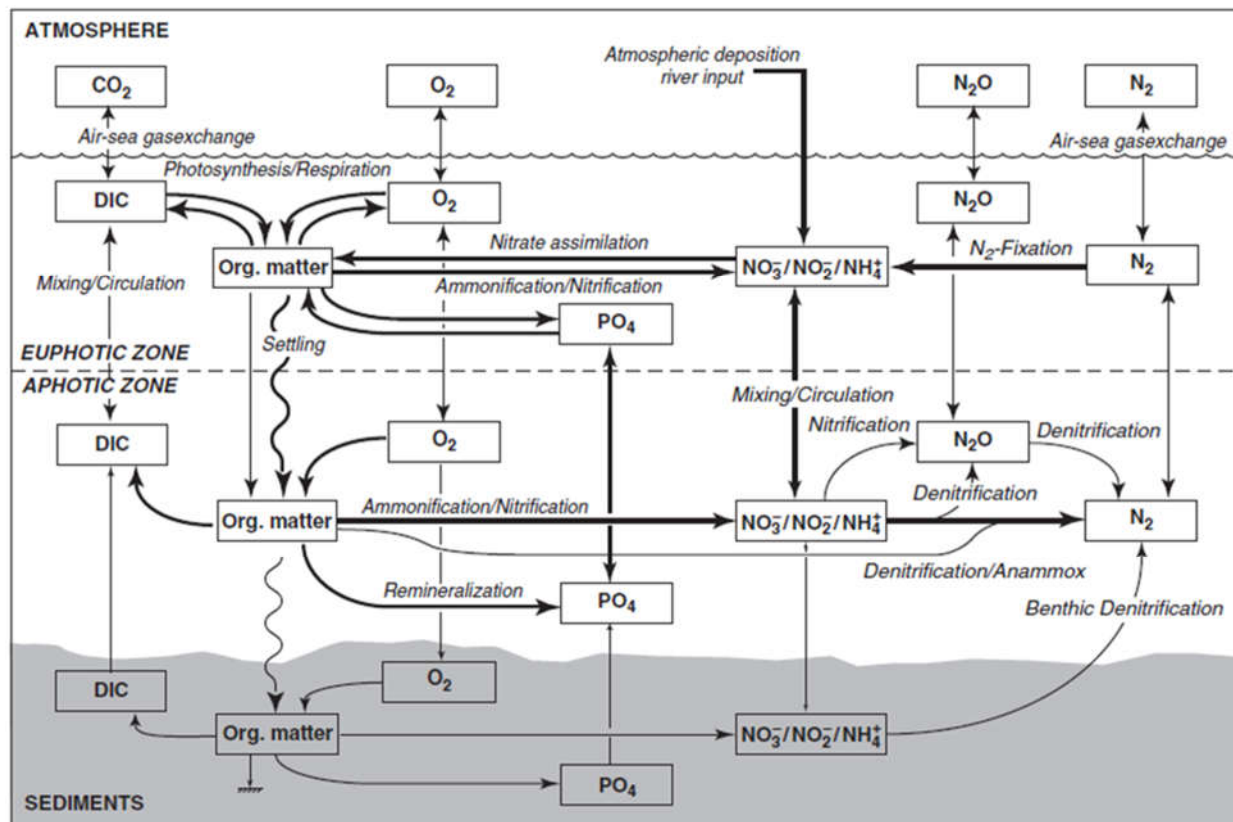


Figure 1-6. Nitrogen cycle (Gruber, 2008).

In sediments, organic nitrogen can be transformed into NH₄⁺ by the ammonification or mineralization processes by bacteria, actinomycetes and fungi in oxic conditions (Herbert, 1999). After that, in oxic conditions, nitrifying bacteria living in sediment can transform NH₄⁺ to NO₂⁻ (e.g. *Nitrosomonas* species) and then to NO₃⁻ (e.g. *Nitrobacteria* species); this process is called nitrification or ammonia oxidation (Smil, 2000; Tuominen et al., 1998). Besides these processes, the denitrification process can reduce nitrate into N₂ gas under anoxic conditions and release it to the atmosphere.

3.1.2. Nitrogen sources

The global demand for food has been responsible for an enormous increase in the anthropogenic N input to the environment with a negative impact on human and ecosystem health. Generally, about 2.7 Tg of N yr⁻¹ are emitted by food waste during consumption (Grizzetti et al., 2013); which took place in North America and Australia with over 1 kgN capita⁻¹ yr⁻¹ (Figure 1-7). Grizzetti et

al. (2013) showed that 35% of nitrogen emissions are in the atmosphere and 65% in the water system.

Domestic and industrial activities have significantly changed N cycle, especially N fixation in land and N input to hydrosystems (Camargo and Alonso, 2006). The expansion of cultivation of legumes – especially soy, alfalfa which consume lot of chemical fertilizers (Vitousek et al., 1997). Agricultural fertilization, biomass burning, cattle and feedlots, and industrial sources are the facilitator of increase of N₂O concentration in the atmosphere (Chapin et al., 2011).

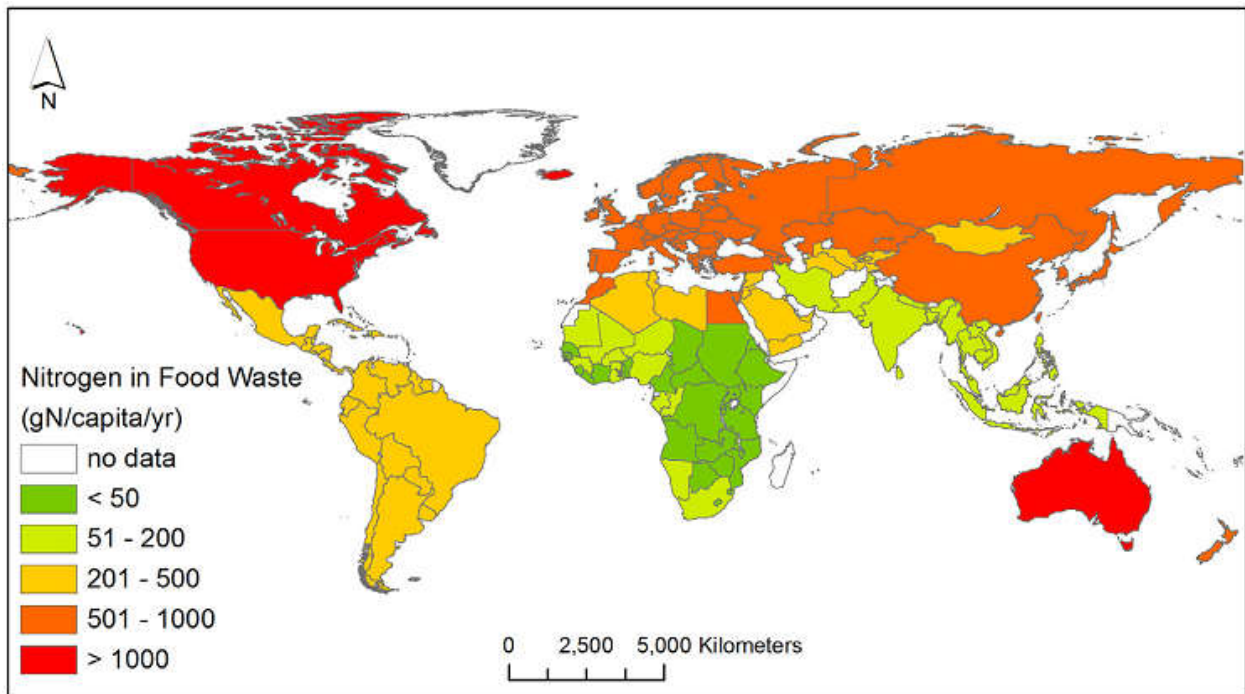


Figure 1-7. Nitrogen loose by food waste in consumption (Grizzetti et al., 2013).

3.1.3. Nitrification-denitrification processes

There are many factors which influence nitrogen transformations in estuaries, among which, saline water is the most important. At oxidized surface layers of sediment, mineralize organic nitrogen processes are affected by salted water. Most of mineralized nitrogen is released as NH₄⁺ form in estuaries. Conversely, in freshwater, nitrogen is a gas. This difference is explained by the physical characteristics and the competition of nitrification and denitrification process in both brackish and freshwater systems (Gardner et al., 1991). The movement of NH₄⁺ in oxic zones can be restricted by cation exchange. This process occurs more easily in freshwater than in brackish water. This can

lead to long retention time of NH_4^+ in freshwater and river-water column potentially contains high concentration of NH_4^+ . In contrast, in brackish (or salt) water, NH_4^+ existed in ion pair formation with anions, e.g. Cl^- , Br^- ... It leads to prevent cation exchange process between sediment and seawater; and NH_4^+ can be diffused to salted water before nitrification process happened (Magalhães et al., 2005) (Figure 1-8).

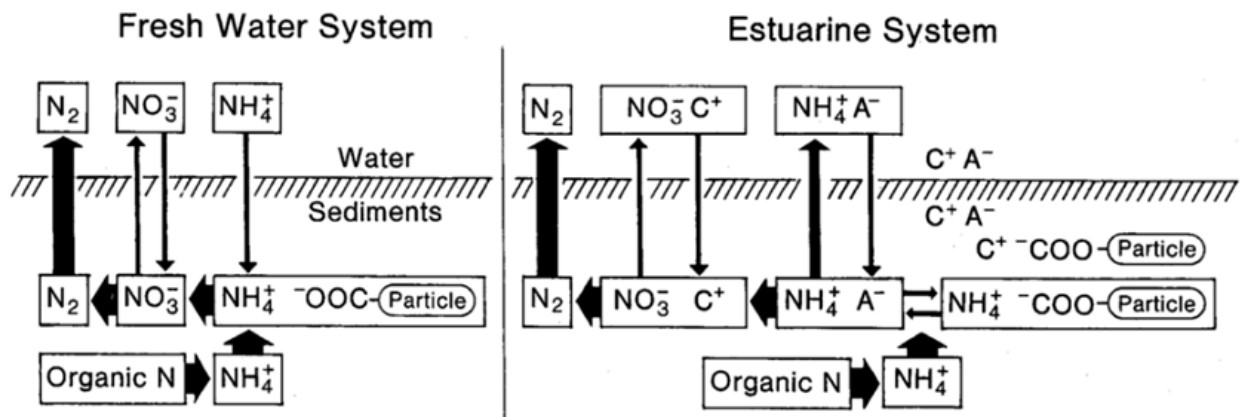


Figure 1-8. Model of differential ammonium release between freshwater and brackish estuarine water (and coastal marine) sediments. Ion pairing is indicated by the coupling of anions (A^-) and cation (C^+) in the diagram (see in Gardner et al., 1991).

3.2. Phosphorus

3.2.1. Phosphorus cycle

Phosphorus in ecosystems is released from rocks during weathering process to create a soluble form which can be uptake by plant to form organic compounds. Soil erosion and groundwater leaching then transfer phosphorus into the rivers, down to the ocean (Figure 1-9).

In aquatic systems, phosphorus exists in various forms such as particulate organic or inorganic phosphorus and dissolved organic or inorganic phosphorus. Whilst particulate organic phosphorus (POP) contributes to the organic matter, dissolved organic phosphorus (DOP) is produced during the mineralization process. Ortho-phosphate (PO_4^{3-}) is dominant source for the living organism growth. In rivers or estuaries, PO_4^{3-} concentration is controlled by biogeochemical processes. Algal and microorganism in rivers will uptake PO_4^{3-} for growth (Cotner and Wetzel, 1992). However, adsorption/desorption process will regenerate PO_4^{3-} in the river-water column (Froelich, 1988;

Houser and Richardson, 2010; James and Barko, 2004). When adsorption or precipitation processes on sediment surface occur, PO_4^{3-} becomes particulate inorganic phosphorus (PIP) (Avilés et al., 2006; House, 2003).

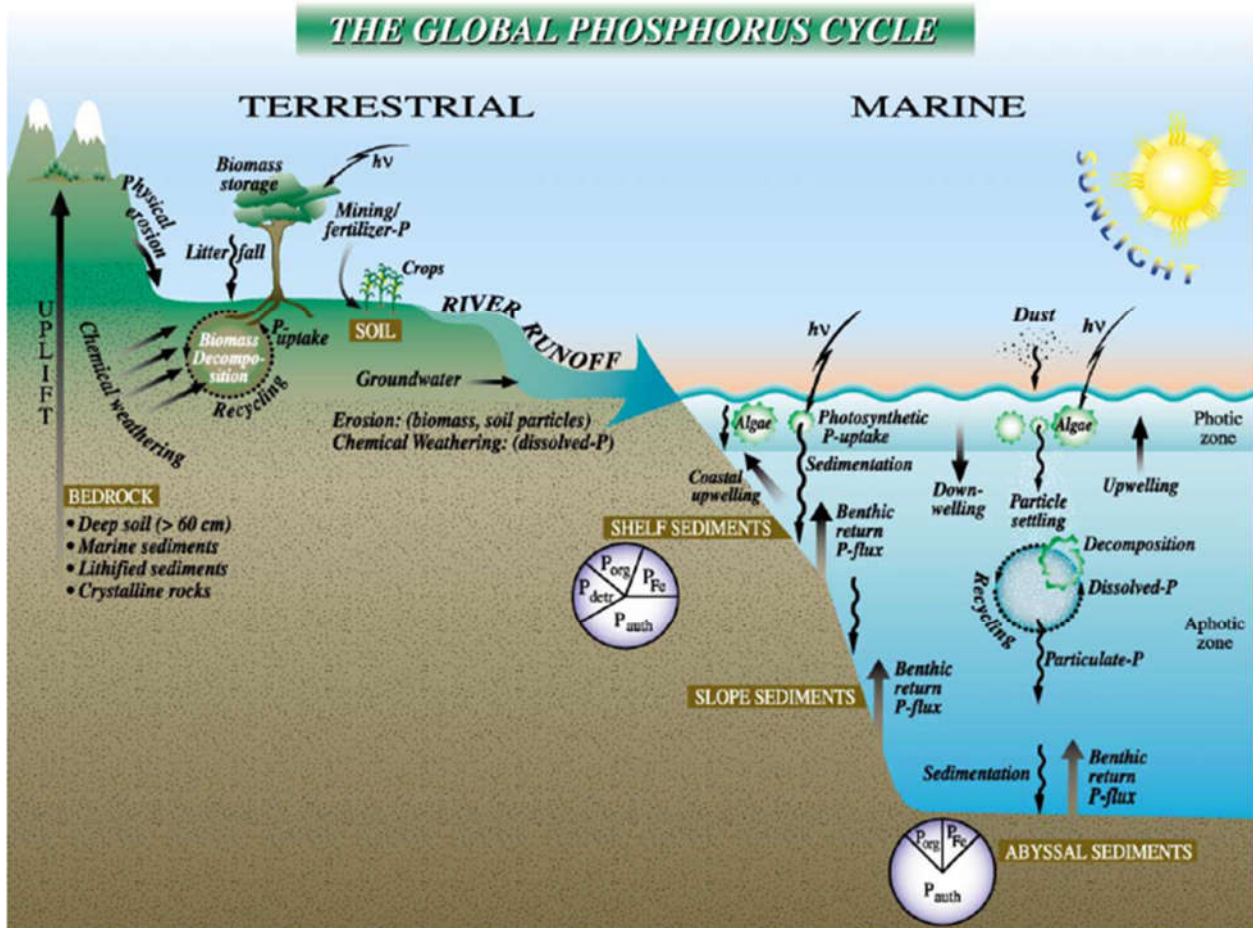


Figure 1-9. Phosphorus cycle.

3.2.2. Phosphorus sources

Phosphorus is generally used to produce fertilizers and detergents, which caused contaminations in lakes, rivers and oceans. Point sources have been by far the main source of anthropogenic dissolved inorganic phosphorus (DIP). Human activities contribute to 19 % of total dissolved organic phosphorus (DOP) export, with diffuse sources. DIP and DOP transfers to coastal zone (1.09 and 0.67 Tg yr⁻¹, respectively) are significantly lower than transfer of particulate phosphorus – PP (9.03 Tg yr⁻¹) (Seitzinger et al., 2005 and references therein).

By combining with nitrogen, phosphorus has been by far the most important nutrient limiting phytoplankton development in coastal zone. The increase of nitrogen input leads to high value of dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) ratio, with some impacts on the phytoplankton growth and risks of algal blooms (Jickells, 1998). Phosphorus in marine waters depends on the balance between PO_4^{3-} input from rivers, sediments and the recycling in the ocean system (Baker et al., 2011). By contrast to nitrogen cycle, phosphorus input from atmosphere are not important (except for some oligotrophic ocean zones, Mills et al., 2004). Most of the time, estuaries are very important catalyzer, in the transformation of P between continental and oceanic areas (Statham, 2012).

3.2.3. The factors influencing the adsorption-desorption of phosphorus onto sediments

The transformation of phosphorus and its movement between sediment and water column is controlled not only by physico-chemical processes, such as adsorption/desorption, precipitation/dissolution and diffusion, but also by biological processes such as mineralization of organic phosphorus. When the water body contained high phosphorus level, PO_4^{3-} will be transferred from solution to sediment; this is adsorption process. At the opposite, if algal uptake phosphorus, it leads to a decline of phosphorus level. Phosphate ion will be released from particles into the water. This is desorption process. These two processes can control PO_4^{3-} concentrations in the river-water column via sorption process and exchanges between sediment bed and the water column (Garban et al., 1995; House et al., 1995; Jarvie et al., 2005).

3.3. Silica

3.3.1. Silica cycle

Dissolved silica (DSi – H_4SiO_4) plays an important role for the growth of organisms (diatoms, radiolarian, and siliceous sponges) (DeMaster, 1981, 2002). These organisms use DSi to establish their bodies (frustules or skeleton), and they deposit Si as a solid amorphous Si (Conley, 2002; Sferratore et al., 2006). DSi originates from diffuse sources and essentially depends on basin lithology (Sferratore et al., 2005). Plants get silica from soils, and then transform the soluble silica into opal, which is occupied in the cell wall, cell laminas, and intercellular spaces. DSi can be transferred to the ocean by many paths including rivers, submarine volcanism and associated

hydrothermal emanations, glacial weathering, low temperature submarine weathering, release from silica-enriched pore water of pelagic sediments on the seafloor (DeMaster, 1981, 2002).

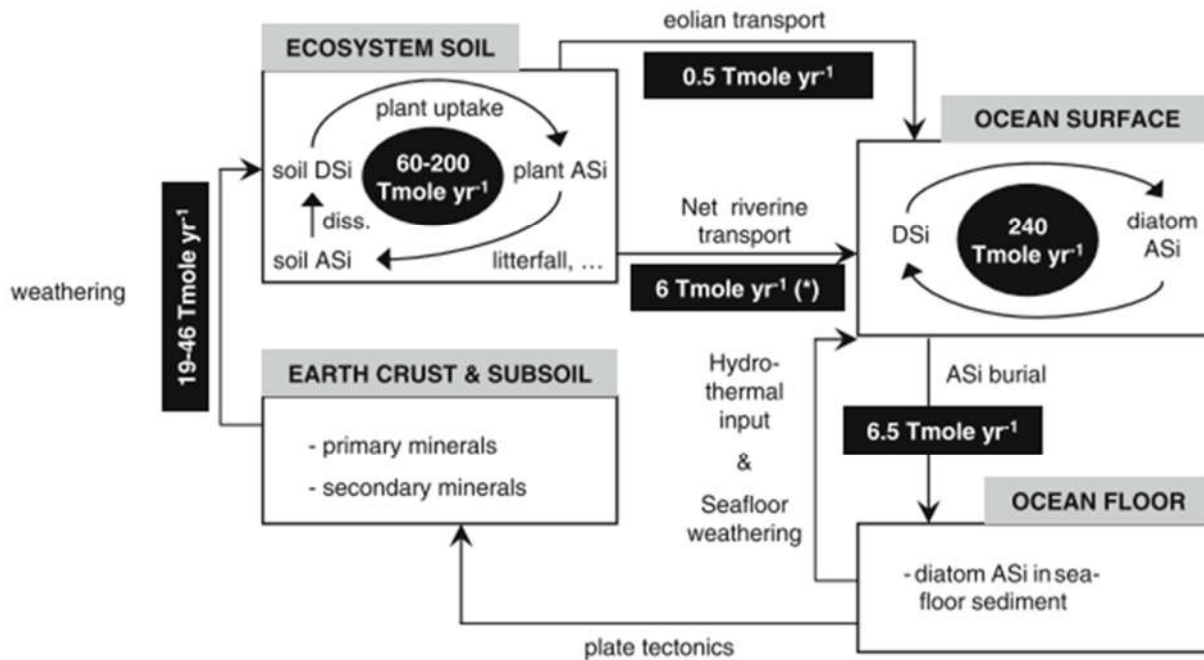


Figure 1-10. Silica cycle (Struyf et al., 2009).

It is also linked with some biologic processes that take place in deep-water layer (Figure 1-10). Firstly, part of silica from dead organism is dissolved into water column, enriching DSi in aquatic environment. Secondly, diatoms stand at 43% of the ocean main producer, and dominate silica input in marine water (Boggs, 2006). The production is then radiolarians which exist under form of suspended amorphous silica.

3.3.2. Silica sources

The estimated global river export of DSi is about 380 Tg yr⁻¹ (340 – 427 Tg yr⁻¹). Globally, 18 – 19 % of DSi amount is retained in reservoirs and river basins (Beusen et al., 2009). In fresh water bodies, silica concentration typically ranges from 5 to 25 mgSiO₂ L⁻¹ and the global average silica level is about 13 mg L⁻¹ in river water, normally seawater contained about 6.4 mgSi L⁻¹.

DSi is carried to aquatic ecosystem through diffuse sources by weathering. Point sources are not likely to be significant, even though it is reported that DSi also enter to water column from domestic and industrial wastewater. Moreover, Amorphous silica (ASi), produced from terrestrial

ecosystem through erosion, is able to enter and can also become an important source of DSi (Graiver et al., 2003; Van Dokkum et al., 2004).

Many studies illustrated that silica level can decrease in estuary, through adsorption process. Suspended sediment can sorb DSi (Aston, 1980). The uptake of DSi by diatoms and opal mineral is an important biogeochemical process in estuaries. Bloom of diatoms is frequently related to a significant uptake of silica. This process occurs if the resident time of water in the estuary is longer than the time of development rate of diatoms; otherwise, the growth of diatoms occurs downstream, in the coastal zone.

3.4. Nutrients fluxes

Many studies have linked riverine nutrient exports with anthropogenic driven changes on freshwater, coastal and marine ecosystem at global scale (Meybeck, 1982; Turner et al., 2003). Because of their impacts on aquatic ecosystems, quantifying nutrient fluxes is necessary for developing management strategies to control limiting factor and eutrophication. As discussed before, nutrients are retained or eliminated through various processes during their carriage to the ocean. The difference between total inputs of nutrients and total outputs represents the retention processes (sedimentation and storage, adsorption/desorption of P onto sediment, DSi uptake of phytoplankton) or elimination processes (mineralization and nitrification/denitrification processes).

4. Carbon

Carbon is an element, which has played a major role for life growth on the Earth, since it is presented in most of the biological compounds. Carbon is also a main element of many minerals and also exists in different forms in the atmosphere (Falkowski et al., 2000). In the last centuries, anthropogenic activities have drastically changed the global carbon cycle, by releasing large quantity of carbon, with some consequences on the global climate and ecosystems (Falkowski et al., 2000).

4.1. Carbon cycle

The carbon cycle is represented by the exchange of carbon among the biosphere, geosphere, hydrosphere and atmosphere (Figure 1-11). The chemical, physical, geological and biological processes have stemmed of the exchanges of carbon between formations (Falkowski et al., 2000). The natural cycle of carbon in the atmosphere, ocean, terrestrial ecosystems and sediments is in equilibrium, so that human would not be affected because of stable carbon concentration in all compartments (Griggs and Noguera, 2002).

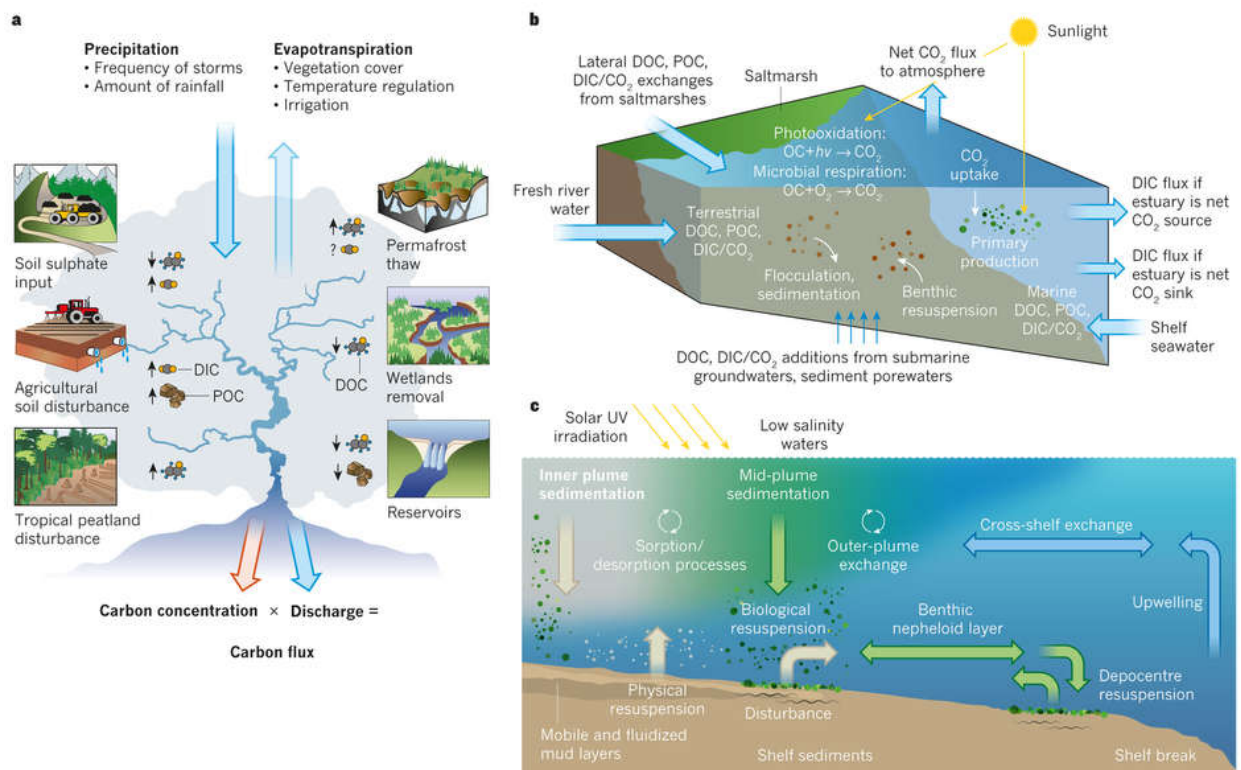


Figure 1-11. Carbon cycle (Bauer et al., 2013)

In the atmosphere, by photosynthesis processes, carbon dioxide goes out of the atmosphere and enters into the terrestrial and oceanic biosphere. In water, carbon dioxide reacts with water to form carbonic acid. After that, weathering process leads to the the adsorption of carbonic acid by rocks (Falkowski et al., 2000). Besides, carbon forms as organic matter and inorganic carbon in terrestrial biosphere (Lal, 2008). Compared to approximately 500 gigaTons of carbon in plants and organisms, there is about 1 500 gigaTons of carbon in soil (Griggs and Noguera, 2002; Lowe and Zealand, 2007). Organic carbon is by far the major component among all organisms. Autotrophic

organisms convert carbon dioxide to organic carbon, whilst heterotrophic organisms consume carbon of other organisms. Carbon leaves terrestrial biosphere by the combustions or respiration of organic carbon. After that, carbon is carried out to the sea by rivers or retained in soil (Bond-Lamberty and Thomson, 2010).

In aquatic system, oceans is the largest space for storing carbon in the world (Falkowski et al., 2000). Besides that, dissolved inorganic carbon (DIC) is around 15% in deep layer higher than in the surface layer (Jorge L. Sarmiento and Nicolas Gruber, 2006). In the deep layer, DIC can be restored for much longer periods of time (Falkowski et al., 2000). Carbon is transferred into the water column (e.g. rivers, lakes, reservoirs) from the atmosphere as dissolved CO₂ (aq.) or dissolved organic carbon (DOC). Organisms convert carbon into the form of organic carbon through photosynthesis process and it contributes to the food web process (Griggs and Noguera, 2002; Lowe and Zealand, 2007).

The riverine and estuarine carbon cycles are directly linked to global carbon cycle. The organic carbon input is carried out by rivers to estuaries and to coastal zones under the forms of POC and DOC (Cadee, 1984). Additionally, particulate or dissolved carbon concentrations are changing according to river-water discharge, seasonal variations (Meybeck, 2006). POC and DOC forms are produced by weathering/erosion from soil or soil leaching during wet season and by aquatic production during dry season (Cauwet and Mackenzie, 1993; Schlesinger and Melack, 1981), before being transported to riverine and estuarine systems. In the recent decades, POC proportion in suspended sediment was ranged from 1.7 to 4.0%; usually this value declined with an increase in suspended sediment concentration (Milliman et al., 1984).

4.2. Anthropogenic influences

Recently, land use change has led to the loss of biodiversity, which decrease ecosystems' regeneration and their ability to remove carbon from the atmosphere. For agricultural purposes, woodlands were replaced with agricultural or urban areas. These replacements of land cover lead to a lower concentration of carbon retained in soil; and as consequences, a longer resident time in the atmosphere. Another reason for the shift of carbon contents in ecosystems is human induced air pollution. Agricultural activities can also lead to a serious increase of atmospheric carbon through the leaching and erosion of soils, which alter plant productivity.

The carbon cycle in oceanic biosphere is also affected by human activities. Aquatic environment at urban regions and craft villages are contaminated by wastewater and solid waste. In some megacities, hundreds of industrial companies are polluting water environment due to release of untreated wastewaters directly to drainage/canals or rivers. Water pollution from industrial activities are extremely heavy, especially organic matter pollution. For instance, in textile, paper and pulp industries, wastewater has an average pH from 9-11 and high values of BOD, COD, CN^- , H_2S , NH_3 (Falkowski et al., 2000). On the other hand, many companies without treated wastewater plants – major hospitals, health care facilities – are the main contributor of organic pollution sources.

5. Eutrophication: causes, consequences and assessment

5.1. Causes of eutrophication

For classification of water quality, three principles are used as standards, e.g. trophic state, acidification and toxic pollutions. Waters classified as “eutrophic waters” have a high content of nutrients, which favors the rapid growth of plants and algae. Long growing and slow living flora species cannot resist to high species competition, so that maximum biodiversity generally develops in oligotrophic or mesotrophic waters. Figure 1-12 describes schematically the causes and consequences of eutrophication (Volterra et al., 2002).

Natural eutrophication is a long term process by which lakes, rivers and coastal waters become progressively more (too) productive. It can take millennia to happen. In contrast, human eutrophication is extremely fast (years to decades). It is caused by an artificial and excessive input of nutrients in water bodies, which accelerates the rate of plants and algal growth. When it occurs near water surface, it leads to high biomass production; when it occurs near bottom, it generally leads to oxygen depletion, with some consequences on local flora and fauna and their biodiversity. Surface and subsurface flow and erosion from enriched-phosphorus soil is the main facilitator of the eutrophication in continental freshwaters. A complex interaction between types of phosphorus, type of soils and hydrological conditions are the major factors controlling the process of phosphorus release to surface and subsurface water compartments (Branom and Sarkar, 2004; Schelde, 2006).

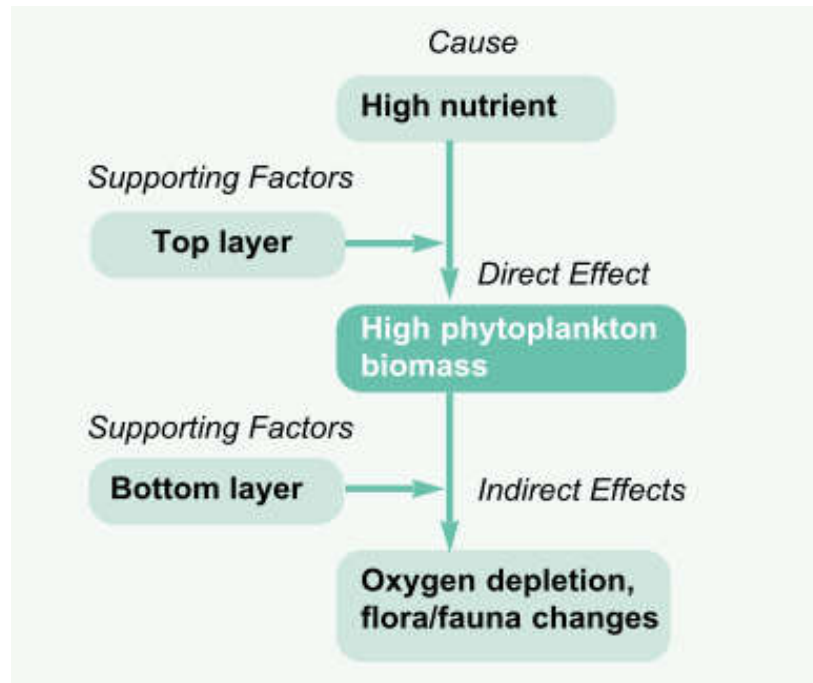


Figure 1-12. The process of eutrophication (Volterra et al., 2002).

During eutrophication process, the amount of nutrients in water change significantly. In some situations, one of the three nutrients may be fully consumed and will no more be available for further growth of algae. This nutrient is then called the limiting factor (Volterra et al., 2002). The N:P ratio in aquatic system is an important indicator, which is used to assess which element will become the limiting factor and has to be controlled in priority to efficiently reduce a bloom. P is likely to be the limiting factor for phytoplankton in freshwaters. The medium areas such as river are often phosphorus limited during spring, but might turn to silica or nitrogen limited systems during summer.

Among the main causes of anthropogenic eutrophication figure the effluents from wastewaters and aquaculture (and other farm wastewaters) (Volterra et al., 2002). Its wide development in many countries leads to a strong and uncontrolled increase of nutrients, through the release of unused animal food and excreta of fishes into the water bodies. Other recent human activities, such as the mining of P, the use of P as a fertilizer have also exacerbated eutrophication all around the world.

5.2. Physical factors supporting the development of eutrophication

The first factor contributing to possible eutrophication is the excess of nutrients. However, the aquatic system can accentuate risks of eutrophication from purely physical reasons. The stability

of the water column, such as frequently observed in lakes or even in some large rivers, can mechanically “trapped” some water masses in such a way that nutrients accumulate and promote the risk of eutrophication (Steinberg and Hartmann, 1988). The physical process can also be dynamic; the shape of the shores, the bathymetry, and local hydrodynamic conditions driven by streams, turbulent mixing and tidal movements can generate some gyres where nutrients accumulate. However, we may notice that in most coastal and estuarine regions, the mixing of water by tidal currents contributes to the dilution of nutrients in excess and limits the risk of eutrophication.

5.3. Consequences and effects of eutrophication

The most important effect of the eutrophication is the depletion of oxygen in water bodies (Billen and Garnier, 1997; Volterra et al., 2002), which can lead to the death of some species (e.g. zooplanktons and fishes). Symptoms of eutrophication are the death of some animals, high density of algal mass, the increase of biomass and change in species of algae and phytoplankton, decrease in the clarity of water.

For human beings, there are two main risks when eutrophic water is used as drinking water. First, water that contains too much organic matter is not suitable for consumption. The presence of specific cyanobacteria in fresh water is the second risk for human consumption (Arash et al., 2012). The cost to reduce organic matter and cyanobacteria from eutrophic water is high and techniques can be complex (EPA, 1975). In water treatment plants, the main issues caused by eutrophication are the disruption of flocculation and chlorination processes, the clogging of filters and the increase of pH (Smith et al., 1998).

5.4. Assessment of eutrophication

To assess eutrophication risks in aquatic system, many trophic parameters exist, including total nitrogen, total phosphorus, carbon, silica, chlorophyll *a*, etc. The quantity of nutrient (N, P, Si, and C) carried out to the coastal regions by large river systems as well as the balance between these elements, are the key determinants of coastal marine eutrophication.

The increase of N input in many rivers lead to a higher DIN:DIP ratio. When the DIN:DIP ratio is far higher than Redfield ratio (N:P = 16:1), biogeochemical cycles are strongly impacted with risks

of eutrophication (Slomp and Van Cappellen, 2004). Five decades ago, Redfield et al. (1963) proposed the Redfield stoichiometric ratio of carbon, nitrogen and phosphorus (C:N:P) as a tool to evaluate eutrophic state of water body. More recently, Billen and Garnier (2007) improved the concept to address the question of coastal waters that eutrophication is the consequence of unbalanced riverine nutrient inputs, with excess nitrogen and phosphorus with respect to silica (C:Si:N:P = 106:15:16:1). The enrichment of riverine waters in N and P, together with decreasing suspended solids and Si, often result in eutrophication of coastal areas (Billen and Garnier, 1997; Conley et al., 1989), characterized by non-diatoms harmful algal blooms. The new production of non-siliceous algae can be facilitated by the imbalance between nutrient inputs. In addition, the changes in amounts and forms (dissolved and particulate, inorganic and organic) and the ratios of nutrients (N and P) have received more attention as a predictor of unnatural phytoplankton community composition and production rate (Redfield et al., 1963; Turner et al., 1998). Redfield et al. (1963) suggested a nutrient ratio (C:N:P:Si = 106:16:1:16) as an optimum for the development of phytoplankton, while other scientists proposed a similar ratio based on some different forms of nutrient; e.g. DIN:DIP:DSi = 14:1:83 (Turner et al., 2003), DIN:DIP = 8–45 (Klausmeier et al., 2004), C:N:P = 88:14:1 (Seitzinger et al., 2005) and DIN: DIP:DSi = 30:1:114 (Seitzinger et al., 2010).

In our work, we focus on the relative percentage of nutrients in freshwater inflows, and particularly to the content of Si in relation to N and P. Unlike N and P, Si concentration in the water is due to natural processes of rock weathering and erosion, and is poorly affected by anthropogenic activities, as demonstrated previously by (Sferratore et al., 2006) from investigations conducted in the Seine River.

6. Sediments in estuary

Nutrient dynamics in estuaries is complex. As reported in Figure 1-13, there are many parameters and processes that can influence nutrients speciation over time and space (Statham, 2012). Anthropogenic nutrient inputs are an important source affecting nutrients fluxes from the land to the ocean (Meybeck, 1982). The second main physical driver is undoubtedly the mixing of river and coastal water by tides, which acts as a biogeochemical catalyzer between nutrients and sediments.

Tides are driven by the conjunction between gravitational forces (mainly lunar and solar) and the geomorphology and bathymetry of coastal and estuarine zones. Water level fluctuations between low tides and high tides is an active process of waterbodies advection and of exchange between sediments and water in the low lying area. This cause currents contribute actively to the turbulent mixing of fluvial and coastal waters.

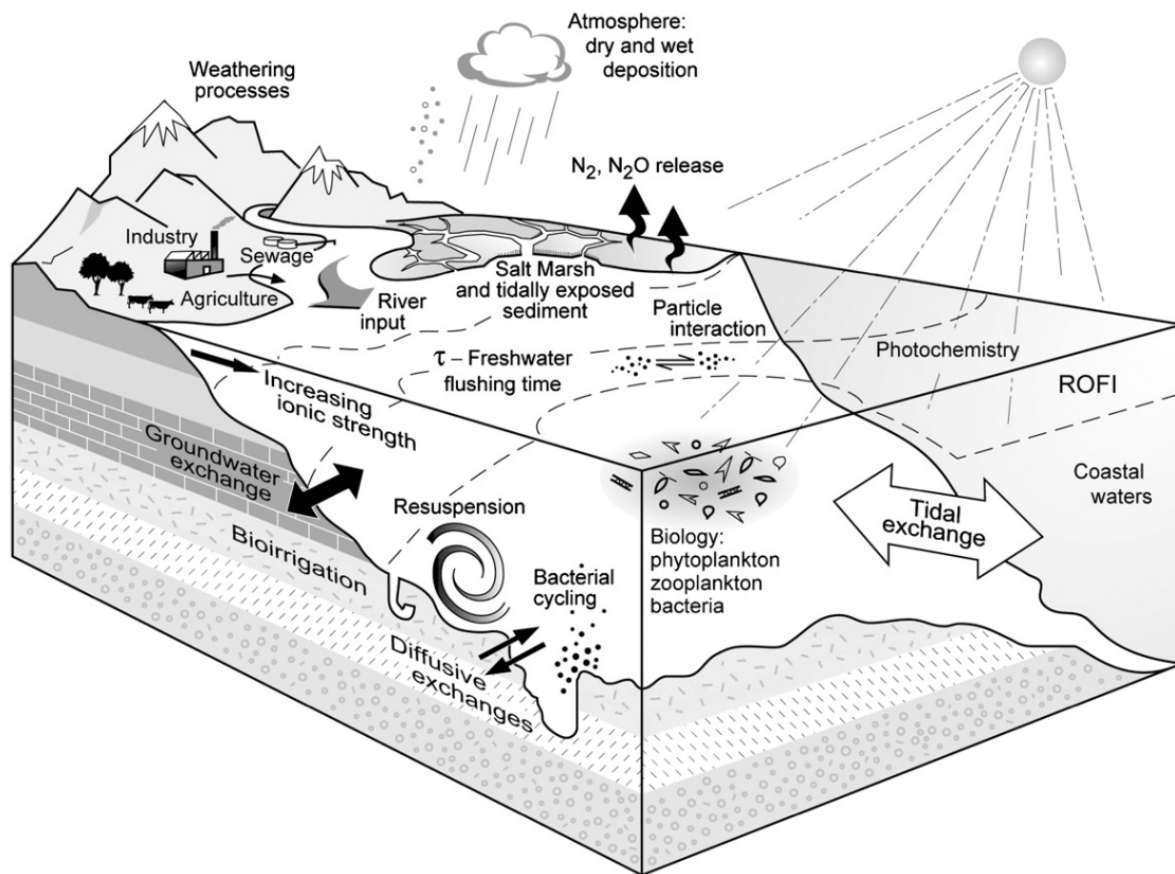


Figure 1-13. Estuarine processes influencing nutrients dynamics (Statham, 2012).

Hydrodynamics and biogeochemical interactions are the keys to understand the nutrients behavior in estuaries. There are two alternate phase during tidal cycles, which have contrasted effects on the hydrodynamics. Ebb tidal period corresponds to the freshwater discharge from river to the estuary; and flood period is the period during which there is an injection of salted water from sea to river. The complete tidal cycle (ebb/flood) governs the vertical stratification and mixing of water masses in the estuary (Shi, 2010). In addition, the nutrient biogeochemical processes can be affected by some processes taking place in the intertidal mudflats. Mudflats are active environments where sediments can fix metals, nutrient, organic and inorganic compounds (Tamtam et al., 2008). These

compounds can further be uptake by organisms (Urban et al., 2009). Sediment stored in the water column and in the mudflats are generally actively remobilized by hydrodynamics forcings, so that the understanding of biogeochemical budgets of estuarine environments depends on many interconnected processes and requires a holistic approach.

CHAPTER 2. MATERIALS AND METHODOLOGY

Chapter 2 describes the study site, the field monitoring campaigns and the laboratory analysis that have been realized to evaluate the water quality and eutrophic status of the Saigon – Dongnai River system. The water quality data supported by the Center of Monitoring (CEM) of the Department of Natural Resources and Environment (DONRE) have also been processed and are used for fluxes calculation.

1. Field study

1.1. The Saigon – Dongnai Rivers hydrosystem

The Saigon River, located in Southern Vietnam, is about 250 km long with a catchment area of 4717 km² (Figure 2-1) and stretches between 10°40'N to 12°00'N and 106°10'E to 106°40'E. It originates from Phum Daung in south-eastern Cambodia, flows to the Dau Tieng Reservoir – the fourth biggest reservoir in Vietnam (120-270 km²; 470-1680 million m³). The regulation of its water discharges aims at controlling the intrusion of saline water to prevent damages on industries; especially on the drinking water treatment plant at Hoa Phu located upstream of HCMC (Figure 2-1). Downstream the reservoir, the Saigon River flows through the HCMC where the river is connected to urban canals and confluences with the Dongnai River to form the Soai Rap River which flows through the Can Gio mangrove ecosystem and discharges into the East Sea, 20 km north of the Mekong Delta. The Saigon River and Dongnai River are the main sources of water supply for HCMC (Tran Ngoc et al., 2016). The Saigon River is connected with the Vam Co Dong River by Vam Thuat canal. Vam Thuat canal is the main canal that collects most of the drainage waters from the actual acid sulfate soils in the Saigon River basin and in Vam Co Dong River. The main tributary of the Saigon River in the upstream of the megacity (Thi Tinh River) drains wastewater from industrial zones. The urban canals (700 km length) were built in the nineteenth century to ensure the connection of the city with the Mekong Delta, to supply the city with freshwater, and to control seasonal monsoon-related flood. They have been deteriorated since the last four decades due to natural and socioeconomic factors, including flood sediment deposition and a poor maintenance.

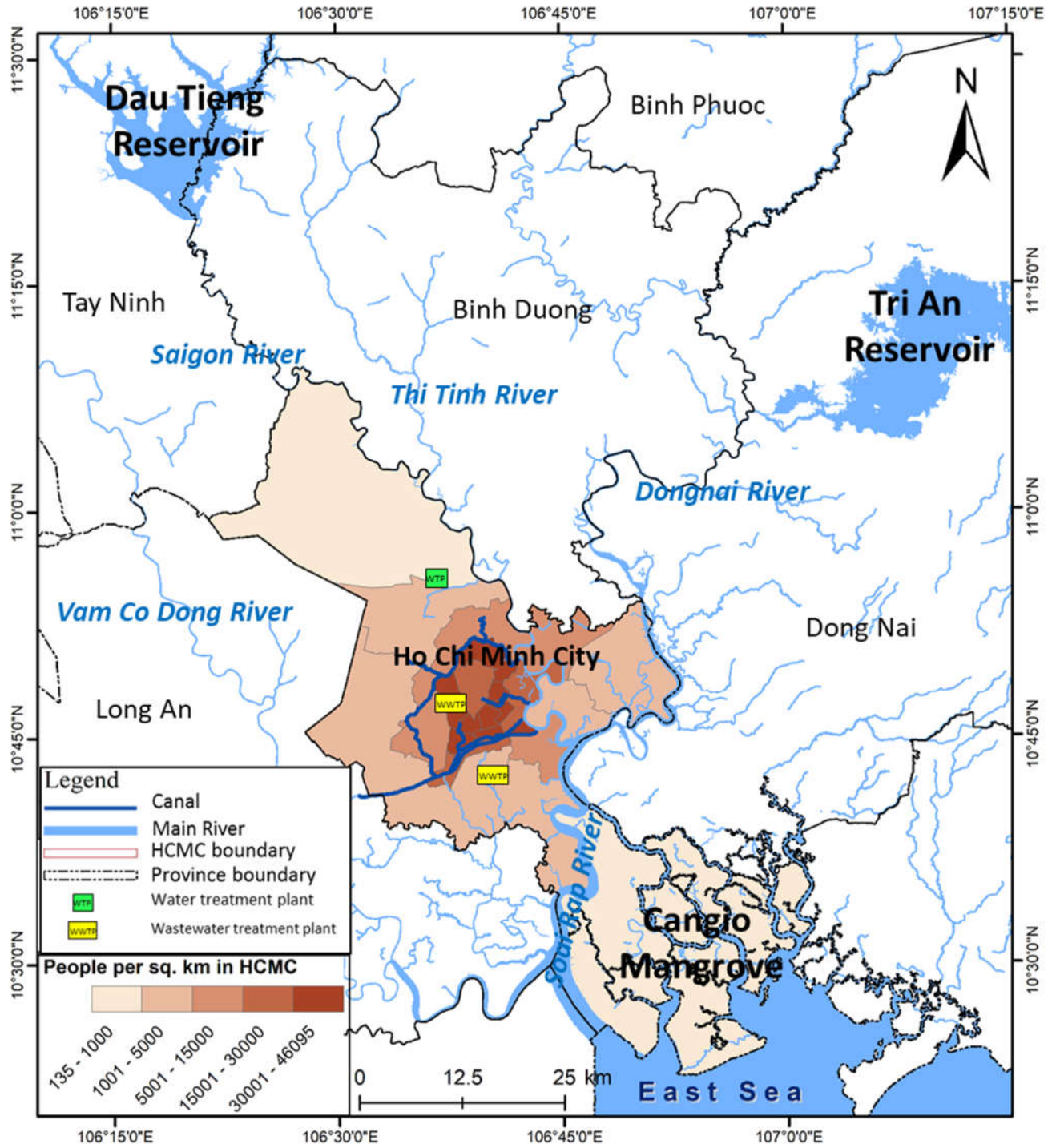


Figure 2-1. Geography and hydrography of Ho Chi Minh City.

The Saigon – Dongnai River system falls in a tropical climate, with an average humidity of 78-82 %. The year is divided into two distinct seasons. The rainy (wet) season, with an average rainfall of about 1800 mm annually (about 150 rainy days year⁻¹), usually begins in May and ends in

November. The dry season lasts from December to April (Tran Ngoc et al., 2016) (Figure 2-2). The Saigon River is affected by asymmetric semi-diurnal tides, which induces alternating river flow direction up to 20 km downstream of the Dau Tieng Reservoir and salty water intrusion upward HCMC, especially during dry season. In the Saigon River, the distribution of the water discharges is mainly controlled by the Dau Tieng reservoir during the dry season to prevent the salt water intrusion (Trieu et al., 2014). The wet season shows the increase of the water discharge from June to October following by a recession at the end of November in both the Saigon and Dongnai Rivers, which is controlled by the precipitation. According to Trieu et al. (2014), the operating and rule regulation of water release from the reservoir can be up to $30 \text{ m}^3 \text{ s}^{-1}$ during the dry season.

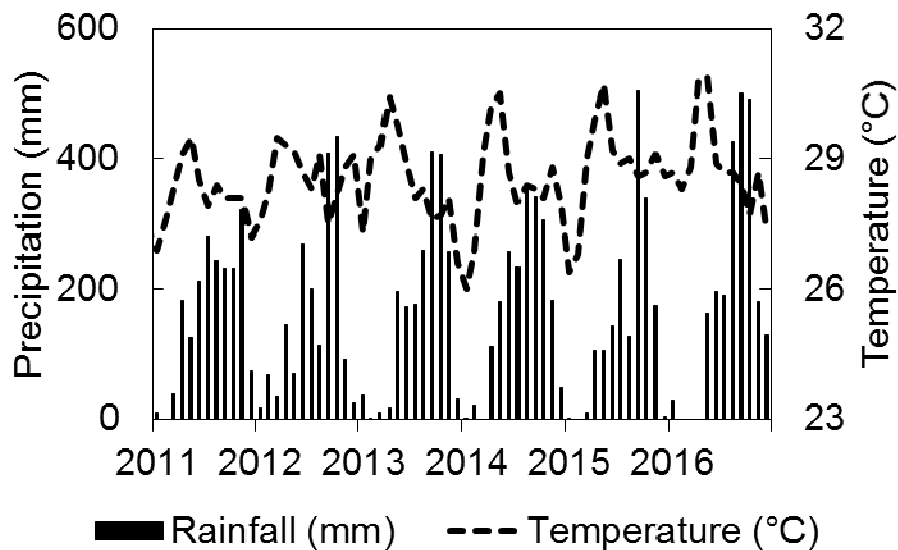


Figure 2-2. Precipitation and temperature in Ho Chi Minh City (source: HCMC Statistical Year Book 2016).

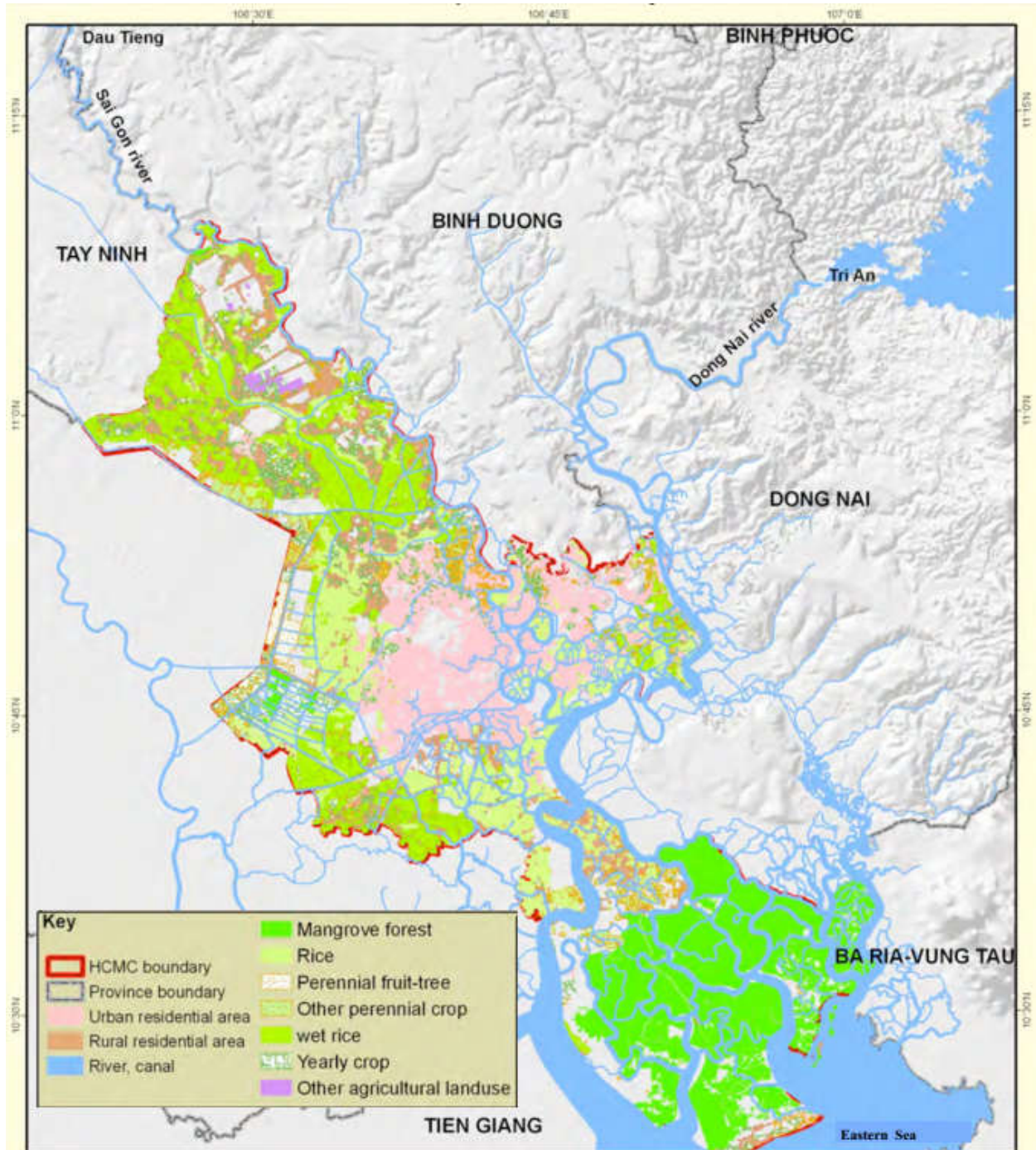
1.2. The Ho Chi Minh City

Ho Chi Minh City (HCMC) is located in a low-elevation coastal zone (LECZ) in the South-eastern region of Vietnam, about 1760 km South of Hanoi capital (Vachaud et al., 2018). The average elevation is 19 m above the sea level. It borders Tay Ninh and Binh Duong province to the North, Dongnai and Ba Ria – Vung Tau province to the East, Long An province to the West and the East Sea to the South with a coast 15 km long. Ho Chi Minh province covered an area of 2096 km^2 in 2014 (Chien, 2014; Tran Ngoc et al., 2016), extending up to Cu Chi district (19 km from the

Cambodia border) and down to Can Gio mangrove ecosystem on the East Sea. The distance from the Northernmost point (Phu My Hung commune, Cu Chi district) to the Southernmost point (Long Hoa commune, Can Gio district) is 102 km, and from the Eastern most point (Long Binh ward, district nine) to the Western most one (Binh Chanh commune, Binh Chanh district) is 47 km.

The population of HCMC was 8.4 million inhabitants in 2016, about 8.9% of total population of Vietnam, making it the highest population-concentrated city in the country. Because of the largest economic and financial hub of Vietnam, HCMC has attracted more and more people from other provinces, resulting in the rapid population growth. Since 2001, the population of HCMC has increased by 196,852 people per year (data from HCMC Statistical Yearbook 2016). During the last 15 years, after reaching an urban population proportion of 88% in 2004, the speed of urbanization in HCMC decreased slightly, thus in 2016, 81% of population lived in urban areas (data from HCMC Statistical Year Book, 2016) (Figure 2-1). The population has been grown rapidly in the boundary district of HCMC. The urban districts tend to stabilize or slightly decrease whereas the new districts and the rural districts increase. Nevertheless, there is a considerable difference in term of density between districts. The highest population density can reach 40,000-45,000 inhabitants km⁻² in some urban and peri-urban sectors.

In HCMC, the land use is dominated by agricultural activities in the north, urban settlement in the center and Can Gio Mangrove forest in the south – 40km southeast of HCMC, which has been recognized as biosphere reserve by UNESCO (Figure 2-3). Agricultural production land was about 32% in 2016, decrease 2% in comparison to 2012 (HCMC Statistical Yearbook 2016).



Source : HCMC Adaptation to Climate Change Study Report

Figure 2-3. Land use in Ho Chi Minh City (source: ATLAS 2010).

Industrial production in HCMC is forecast to see stable growth in 2016, with the index of industrial production expanding by 8-8.5% annually. There are about 30 000 factories in the industrial sector of HCMC, including many large enterprises, high-technology, electronic, processing, light

industries, construction, building materials and agro-products companies. Currently, the city has 22 industrial parks (Figure 2-4) and export-processing zones (source: le PADDI – Centre de prospective et d'études urbaines).

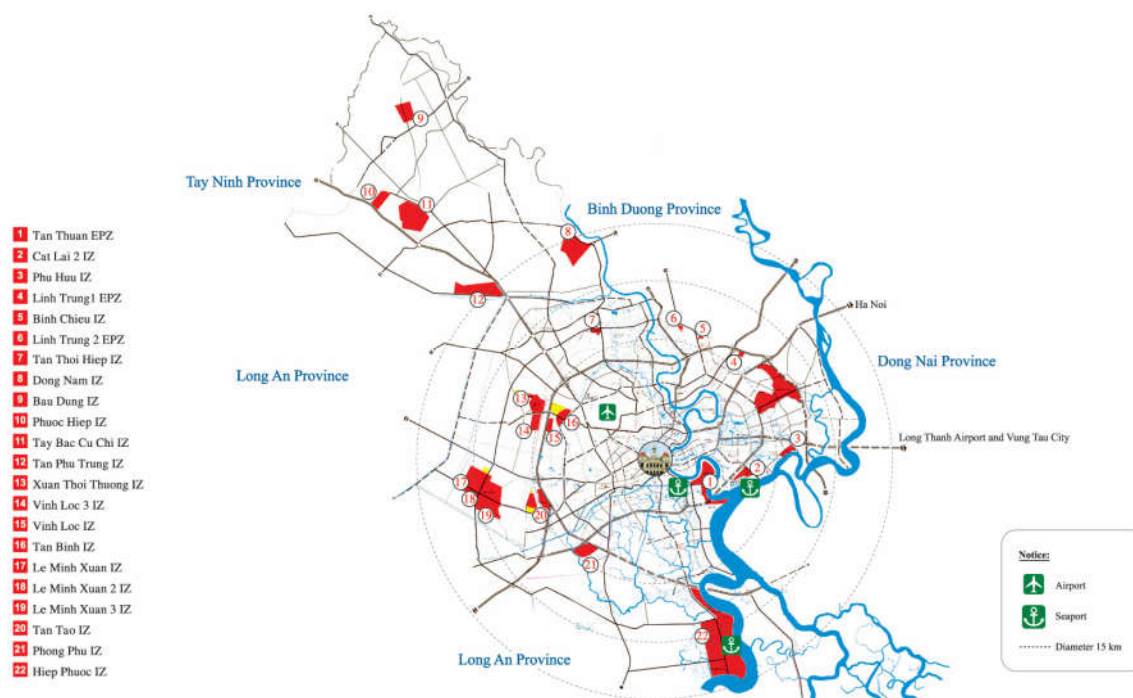


Figure 2-4. Industrial zone distribution in Ho Chi Minh City (source: le PADDI – Centre de prospective et d'études urbaines).

At present, four water resources are used for water supply in HCMC. They are (i) Dongnai River, (ii) Saigon River, (iii) groundwater and (iv) rain water (Le et al., 2003). The current and projected total daily water use demand for domestic and industrial activities in HCMC were 1.75 million m³ in 2005 and 3.6 million m³ in 2020, respectively (Nga, 2006). The key water users in HCMC are residents, industries and services. Water resources used for agriculture in HCMC are (i) raw water taken from the irrigation canals network of the Saigon and Dongnai Rivers and (ii) storm water during the wet season. Groundwater has not been used for agriculture in HCMC because of (i) availability of fresh raw water of the irrigation canals located in the west and southeast of HCMC and (ii) unsuitability of groundwater quality for irrigation, such as high iron content and low pH. The Saigon Water Supply Company (SAWACO) is responsible for exploitation, purification and distribution of water in HCMC. The capacity of piped water in 2006 under SAWACO's

management was 1 236 000 m³ day⁻¹, which include 1 150 000 m³ day⁻¹ produced by three surface water treatment plants taking raw water from the Saigon and Dongnai Rivers (SAWACO, 2004) and 86 000 m³ day⁻¹ produced from groundwater treatment plants. According to an estimation realized a decade ago, there is only less than 10 % of domestic wastewater collected and treated before discharged directly to urban canals or rivers (Marcotullio, 2007). Taking into account population growth, we may hypothesise that treated waters are even less nowadays (~5%). The sewage treatment plant (Binh Hung WWTP) is located in Binh Chanh District, south of the city center. Its treatment capacity is 141 000 m³ day⁻¹ (426 000 inhabitants, i.e. ~5% of the total inhabitants). In 2017, one extension was under construction to reach a total capacity of 469 000 m³ day⁻¹ (1 390 000 inhabitants) and ten new WWTP are planned to be constructed in the next decade (Tran Ngoc et al., 2016). To our knowledge, the corresponding financing plan is not yet consolidated.

2. Methodology

2.1. Monitoring and sampling strategy

2.1.1. Bi-monthly monitoring

Field campaigns were undertaken to investigate the spatial and temporal evolution of water quality of the Saigon River from July 2015 to December 2017. To quantify the major nutrients fluxes and the impact of HCMC, four monitoring sites were chosen (Figure 2-5). Two upstream stations on the Saigon River (SG-up) and the Dongnai River (DN-up) represent the reference water status before the densely urbanized area of HCMC. One monitoring station in the city center (SG-city) highlights the water quality of the Saigon River within the heart of the city. The last point downstream the confluence between the two rivers (NBR) allows evaluating the global impact of HCMC to the estuarine and near shore waters. The major purpose of this monitoring strategy was to characterize both seasonal (dry and wet season) and year to year variations in water quality of the Saigon River.

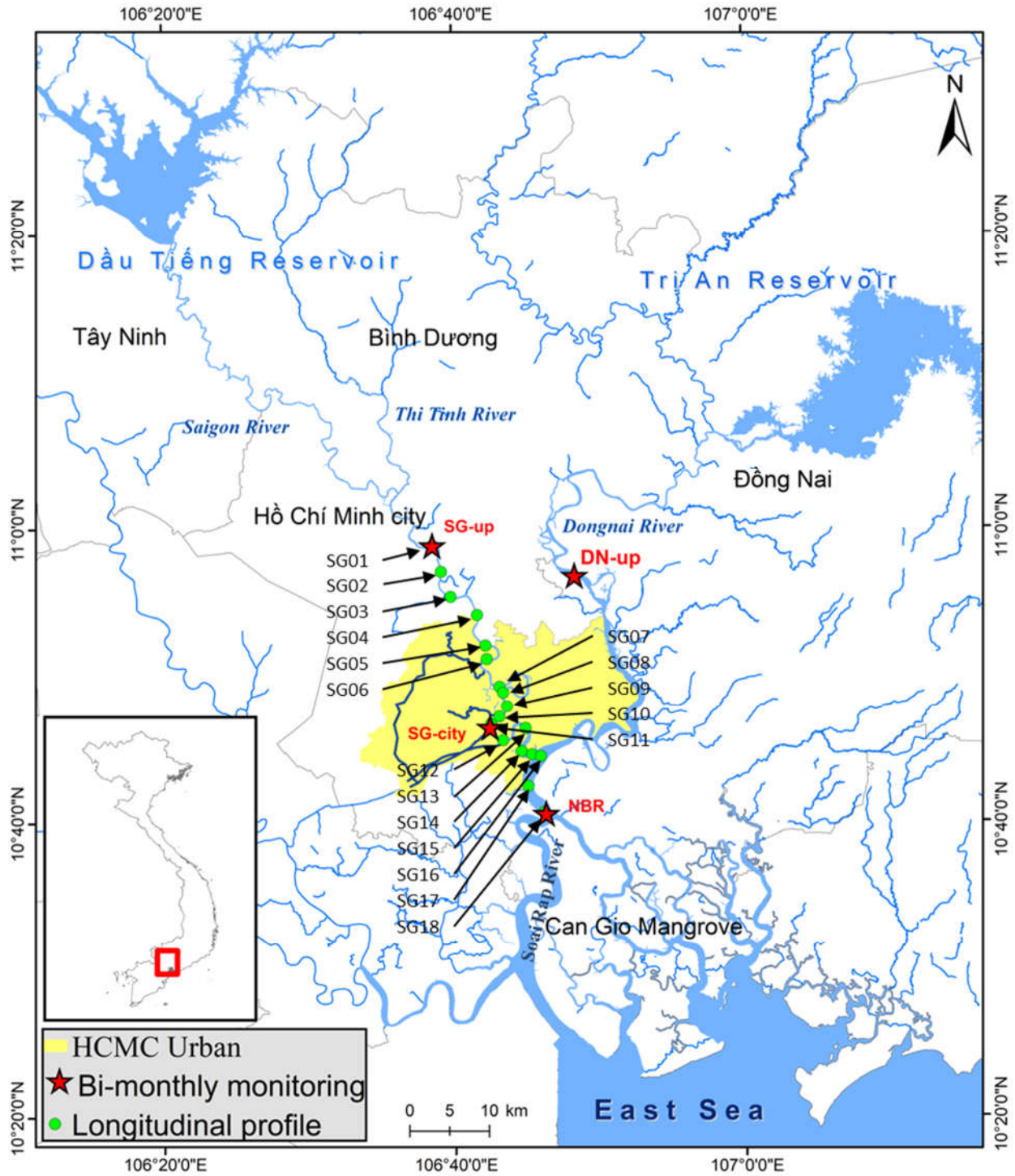


Figure 2-5. Sampling sites of bi-monthly monitoring, longitudinal and salinity profile campaigns.

For each bi-monthly sampling campaigns, surface water (0-30cm below surface) was taken by using a 2.5 L Niskin bottle in the middle of the river either from a bridge or from a boat to immediately measure physico-chemical parameters using a multi-parameter probe (WTW 3420[®]),

e.g. temperature, pH, conductivity, salinity, and dissolved oxygen concentration and percentage of oxygen saturation. Each water sample was collected in a 5-L polypropylene recipient for Total Suspended Sediment (TSS), C, N, Si, and P measurements and Chlorophyll-*a* analysis (see Figure 2-6). Additionally, from October 2016, 2 liters of water sample were taken to identify and to count phytoplankton species (complementary study, see Appendix 1).



Figure 2-6. Field campaigns photo: bi-monthly campaigns in the middle of bridge and boat (a, b), diurnal campaign (c, d e), longitudinal campaign ((f, g, h)

Among the variables followed bimonthly, some are known to vary greatly both on a smaller time and space scales. The hydrological and tidal forcings at high frequency may complicate the assessment of seasonal fluctuations and pluri-annual trends. Additional field trip over 24hours was conducted independently from the pre-cited schedule in order to investigate the possible temporal heterogeneities during tide cycle and limit bias from sub sampling as much as possible.

2.1.2. Diurnal sampling surveys

To measure nutrients concentrations levels and estimate annual fluxes, tide cycle has to be taken into account for the monitoring strategy. In this low land hydrosystem, the tidal cycle actually implies the reversal of flow directions and water body oscillations from upstream to downstream. Based on four surveys during the wet season (September 2016) and the dry season (March 2017) covering a complete tide cycle during 24 hours on the two major tributaries (the Saigon and Dongnai Rivers), we analyzed the hydrodynamic effect on physico-chemical parameters and nutrients concentrations fluctuation within the tide cycle. In particular, these experiments were used to estimate net fluxes of nutrients and propose an adapted monitoring strategy for river managers (complementary study, see Appendix 2).

Two diurnal sampling surveys during the wet and dry seasons were carried at DN-up (from 1:00 pm 9th September 2016 to 1:00 pm 10th September 2016 and from 11:00 am 16th March 2017 to 11:00 am 17th March 2017) and SG-up (from 12:20 pm 20th September 2016 to 1:00 pm 21st September 2016 and from 11:00 am 21st March 2017 to 11:00 22nd March 2017). During these surveys, surface water (0-30cm below surface) was taken every one hour by using a 2.5 L Niskin bottle from a boat, to immediately measure physico-chemical parameters using a multi-parameter probe (WTW 3420[®]) and an Hydrolab MS5 probe. The measured parameters are temperature, pH, conductivity, salinity, turbidity, chlorophyll-a and dissolved oxygen concentration and percentage of oxygen saturation. Every two hours, each water sample was collected in a 5-L polypropylene recipient for Total Suspended Sediment (TSS), C, N, Si, and P measurements and Chlorophyll-*a* analysis (see Figure 2-6). Besides, during the dry season field trip, we took 2 liters of water sample to identify phytoplankton species. Another boat was equipped with an acoustic Doppler current profiler (ADCP) to monitor the flow velocity and calculate the net water discharge at the river section (triplicate every hour during 24h).

2.1.3. Longitudinal and salinity profile surveys

Longitudinal profiles. Two campaigns during the wet and dry seasons from upstream to downstream of HCMC were carried out to understand the spatial fluctuation of water quality in the Saigon River. Total length of the profile is 50 km, with 18 sampling points distributed along the profile (from SG01 to SG18, see Figure 2-5 and Table 2-1 for sampling site locations). The

longitudinal profiles were performed on-boat on 19th April 2017 from 8:00 am to 4:30 pm (dry season) and 20th October 2017 from 9:30 am to 6:00 pm, starting at Saigon upstream – Phu Cuong bridge (SG-up – SG01) and ending at Saigon downstream – Binh Khanh ferry (NBR – SG18).

Salinity profiles. Two surveys were conducted during the wet season and the dry season from the City Center to the downstream of HCMC to understand the variability of nutrient in the salinity gradient. The longitudinal profile was 30 km in length, with 12 sampling points distributed along the profile (from SG07 to SG18, see Figure 2-5 and Table 2-1 for sampling site locations). These surveys were carried out on-boat on 21st and 25th April 2017 (two campaigns during the dry season) and 23rd and 25th October 2017 (two campaigns during the wet season), starting at Thanh Da canal (SG07) to Binh Khanh ferry (SG18).

For all longitudinal profiles and salinity profiles, surface water (0-30cm below surface) was taken by using a 2.5 L Niskin bottle in the middle of the river either to measure physico-chemical parameters using a multi-parameter probe (WTW 3420[®]). Each water sample was collected in a 5-L polypropylene recipient for Total Suspended Sediment (TSS), C, N, Si, and P measurements and Chlorophyll-*a* analysis; and 2 liters of water were used to identify and count phytoplankton species (see Figure 2-6). At each sampling points, the Hydrolab multi-parameter probe was also used to measure the vertical profiles of temperature, pH, conductivity, salinity, Total Dissolved Solid (TDS), turbidity, DO and Chl-*a* from surface to bottom river. It shows that the water column is homogeneous vertically. Besides, deposit sediment samples (1st layer (about 1cm) of fresh sediment) were taken by using sediment grab at the river bank and during the longitudinal profile campaigns. During the 1st campaign of salinity profile of each season, 30 liters of surface water were taken to analyzed particulate C, N and P in the suspended sediments at 12 sampling sites (from SG07 to SG18). During 2nd campaign of salinity profile of each season, 30 liters of water were taken at 0-30 cm below surface and at the near-bottom river by using auto sample equipment to analyzed particulate C, N and P in the surface and bottom suspended sediments at six sampling sites (from SG13 to SG18).

A GPS has been used for georeferencing and for calculating the corresponding kilometric point (pK), in which pK = 0 was defined as the confluence between the Saigon and Dongnai Rivers, negative values upward and positive downward.

Table 2-1. Sampling site locations

Sampling site name	Coordinates	
SG01	10°58'50"N	106°38'33"E
SG02	10°57'04"N	106°39'08"E
SG03	10°55'21"N	106°39'47"E
SG04	10°54'06"N	106°41'35"E
SG05	10°52'00"N	106°42'10"E
SG06	10°51'07"N	106°42'16"E
SG07	10°49'16"N	106°43'03"E
SG08	10°48'47"N	106°43'23"E
SG09	10°47'53"N	106°43'37"E
SG10	10°47'10"N	106°43'03"E
SG11	10°46'28"N	106°42'33"E
SG12	10°45'35"N	106°43'30"E
SG13	10°46'16"N	106°44'54"E
SG14	10°44'51"N	106°44'44"E
SG15	10°44'32"N	106°45'23"E
SG16	10°44'13"N	106°45'42"E
SG17	10°42'32"N	106°45'20"E
SG18	10°40'46"N	106°46'02"E

2.1.4. Atmospheric deposition

To evaluate the nutrient atmospheric deposition and calculate nutrient flux deposition during the rainy season of the year 2017, rain water was collected at four sampling sites: R1 – CARE laboratory roof in HCMC Center, R2, R3 and R4 – resident houses in the south, north and east of HCMC Center, respectively (Figure 2-7). This monitoring started from May 2017 and ended in December 2017; the volume of rain water was collected after each rain event as well as some basic physico-chemical parameters (pH and conductivity). The samples from Monday to Sunday were combined together to have one sample per week. These samples were filtered to analyzed TSS and nutrients (PO_4^{3-} , Total P, NH_4^+ , NO_3^- , Total N and dissolved Silica) (complementary study, see Appendix 3).

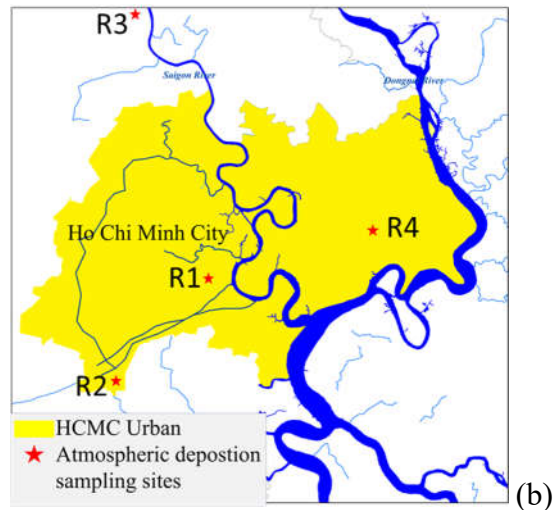


Figure 2-7. Sampling sites of atmospheric deposition monitoring.

2.2. Water quality and hydrological database of DONRE

Water discharges and water quality parameters data were provided by the Center of Monitoring (CEM) of the Department Of Natural Resources and Environment of HCMC (DONRE). This national monitoring program provides monthly discharges and bi-monthly water quality dataseries at different locations within Saigon – Dongnai River basins and within the urban canal network (Figure 2-8). Since the hydrodynamic effect of the tide propagates up to the dams, discharges are measured every hour during 24 hours cycle. The residual discharge is then calculated as the difference between positive and negative discharges over the tide cycle. The residual discharge

corresponds to the net positive flow of the river from land to the sea. Many parameters, e.g. temperature, pH, TSS, salinity, turbidity, NH_4^+ , PO_4^{3-} , COD, BOD_5 , DO, Pb, Cu, Cd, Hd, Mn, oil, coliforms and *E. Coli* were measured. Since 2005, CEM is in charge of a bimonthly monitoring of 26 sites for water quality every two weeks and a monthly monitoring of water discharge at 15 sites distributed in the Saigon – Dongnai River basin (HCM province) and 16 sites in urban canals. The selected period 2012-2016 allows getting homogenous and suitable database of water discharge, TSS, NH_4^+ and PO_4^{3-} in order to characterize the mean monthly distribution of the discharge and flux calculation.

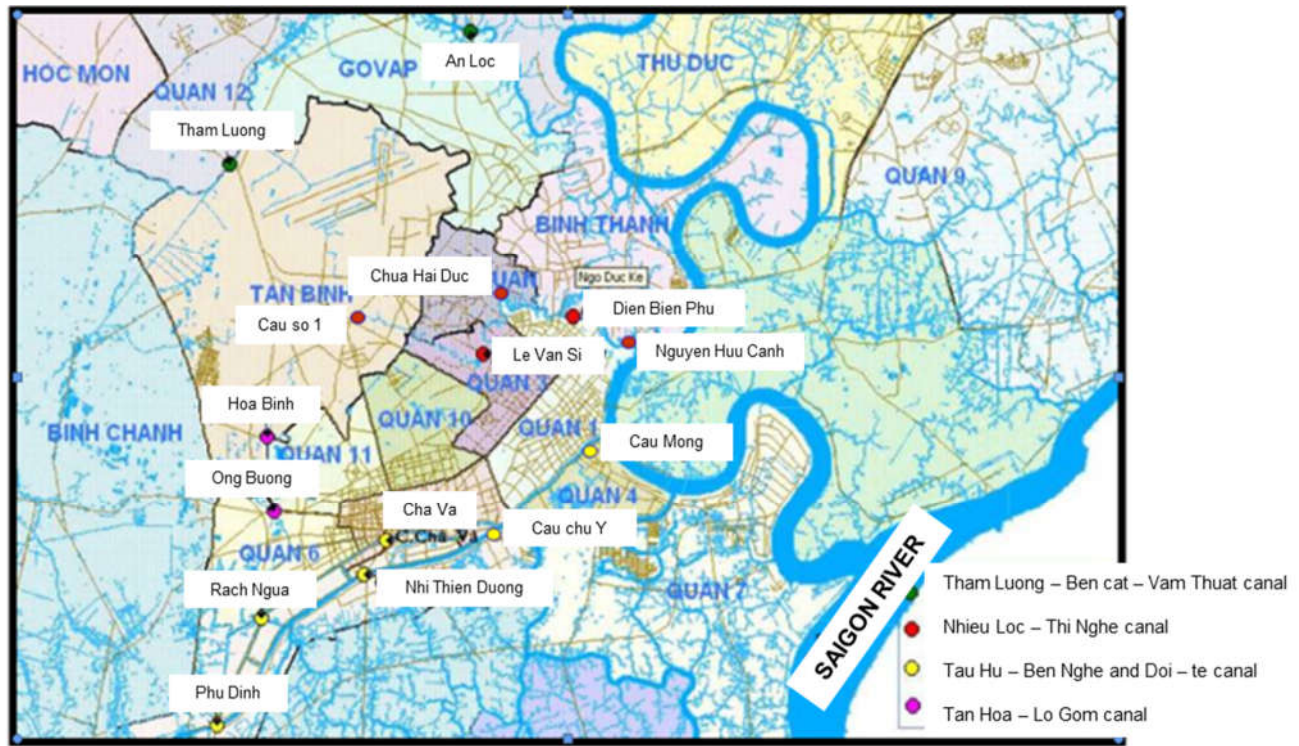


Figure 2-8. Sampling sites in urban canals network and the Saigon – Dongnai River basin (source: CEM-DONRE).

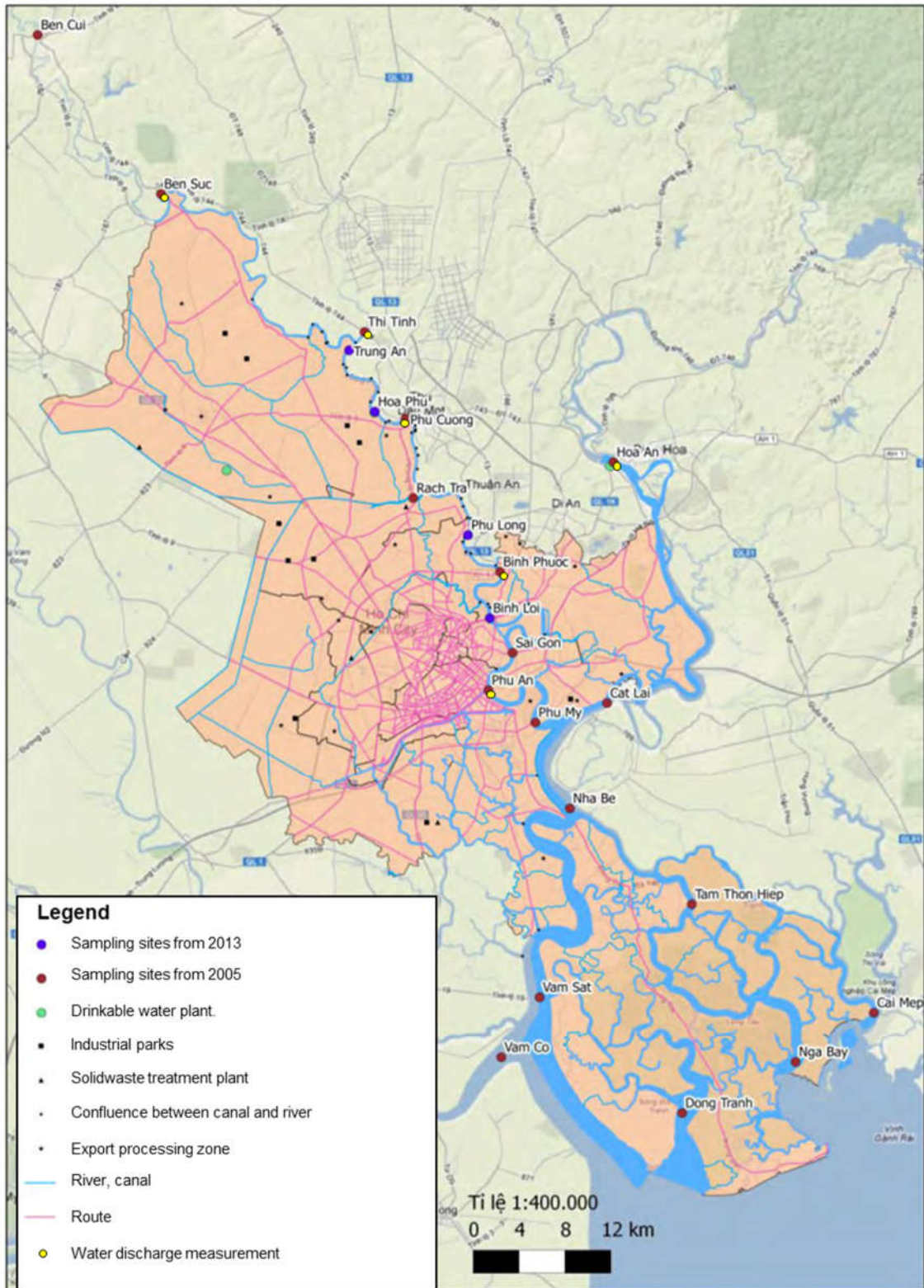


Figure 2-8. Sampling sites in Saigon – Dongnai River basin (source: CEM-DONRE).

3. Measurement and analytical methods

3.1. In-situ measurements of physico-chemical parameters

During field trip, physico-chemical parameters such as temperature (°C), pH, conductivity ($\mu\text{S cm}^{-1}$), salinity (‰), turbidity (NTU), redox potential (mV), TDS (total dissolved solids – mg L^{-1}) and dissolved oxygen ($\text{DO} - \text{mgO}_2 \text{ L}^{-1}$) and oxygen saturation were measured in-situ using a multi-parameter probe (WTW 3420[®]) and Hydrolab MS5.

3.2. Filtration and preservation of samples in laboratory

During sampling campaigns, about 5 liters of surface water samples were collected in the middle of river at each station. Back to the laboratory, samples were filtered immediately to ensure a good storage. Samples were filtered through:

- Whatman GF/F paper filter (glass microfiber filters $0.7 \mu\text{m}$) previously burned at $500 \text{ }^\circ\text{C}$ in order to eliminate any trace organic for dissolved nutrient analyzes as nitrogen (nitrite, nitrate, and ammonia), phosphorus (ortho-phosphate), dissolved organic carbon and dissolved silica (DSi).
- For suspended sediment (SS) and particulate organic carbon (POC) determination on the filter, GF/F filter papers were pre-weighted. Unfiltered samples were also stored to analyze total phosphorus and total nitrogen. After filtration, filters were stored in the fridge at $4 \text{ }^\circ\text{C}$ or $-20 \text{ }^\circ\text{C}$, until the analysis of DOC and nutrients, respectively.
- Whatman GF/F paper filters (glass microfiber filters $0.7 \mu\text{m}$) without burning at $500 \text{ }^\circ\text{C}$ were used for chlorophyll a determination.

3.3. Sediment preparation

Suspended sediments collected from longitudinal and salinity profiles were settled in 30-liters bottles for 24 hours before the water was siphoned off. Wet freshly deposited sediment was then dried in oven or by freeze drier. After that, dry sediment samples were crushed and sieved through $200 \mu\text{m}$ sieves to remove coarse particles.

3.4. Analytical methods

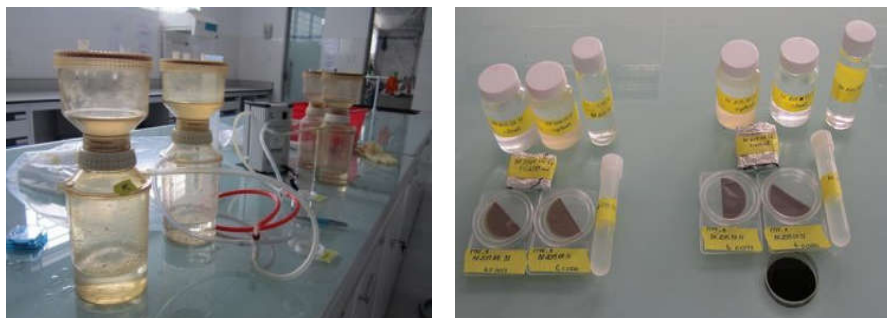


Figure 2-9. Sample preparation in CARE Laboratory

3.4.1. Total suspended solid (TSS)

Suspended solids were analyzed on a pre-weighed standard glass-fiber filter (GF/F) through which a well-mixed sample were filtered. The residue retained on the filter was dried for about 24 hour at 50 °C. Taking into account the filtered volume, the different weight of the filter represented the total suspended solids per unit volume (mg L^{-1}). This protocol was realized in Centre Asiatique de Recherche sur l'Eau (CARE) laboratory in the Ho Chi Minh University of Technology (Figure 2-9).

3.4.2. Chlorophyll a and phaeopigments

The method considered is based on an acetone extraction followed by a spectrophotometry measurement. The method measured chlorophyll a and phaeopigments concentrations in river, estuarine and marine water sample. Chlorophyll a and phaeopigments were extracted in 9mL of 90 % acetone solution. The optical density of sample was measured (using 1cm cell optical path) at 750 and 665 nm, before and after acidification (Aminot and K erouel, 2004).

3.4.3. Dissolved nutrients

The portable spectrophotometer (HACH Lange DR1900) was used for all nutrient samples. Measurements were conducted at CARE laboratory. This instrument is a microprocessor-controlled, single-beam instrument for colorimetric testing with wavelength range of 340-800 nm and silicon photodiode detector. It can be used in the laboratory and in the field. Depending on

their concentration, the dissolved nutrients give a peak in the spectrum, with a specific wavelength signature for each nutrient.

3.4.3.1. Nitrogen

Total nitrogen (TN) was analyzed by persulfate digestion method (American Public Health Association: APHA, 1995). Inorganically and organically bonded nitrogen is oxidized into nitrate (NO_3^-) by digestion with peroxy-disulphate. The nitrate ions react with 2,6-dimethylphenol in a solution of sulphuric and phosphoric acid to form a nitrophenol. The measurement wavelength is 345 nm.

Ammonium (NH_4^+) was analyzed by Nessler method (APHA, 1995). The Polyvinyl Alcohol Dispersing Agent helps the color formation in the reaction of Nessler Reagent with ammonia and certain other amines. A yellow color is formed that is proportional to the ammonia concentration. The measurement wavelength is 425 nm.

Nitrate (NO_3^-) was analyzed by Cadmium reduction method (APHA, 1995). Cadmium metal reduces nitrate in the sample to nitrite (NO_2^-). The nitrite ion reacts in an acidic medium with sulfanilic acid to form an intermediate diazonium salt. The salt couples with gentisic acid to form an amber colored solution. The measurement wavelength is 500 nm for spectrophotometers or 520 nm for colorimeters.

3.4.3.2. Total phosphorus (TP) and Orthophosphates (PO_4^{3-})

Total phosphorus (TP) was analyzed by acid ascorbic method (APHA, 1995). Phosphates in organic and condensed inorganic forms (meta-, pyro- or other polyphosphates) are first converted to reactive orthophosphate in the total phosphorus procedure. Treatment of the sample with acid and heat provides the conditions for hydrolysis of the condensed inorganic forms. Organic phosphates are also converted to orthophosphates in the total phosphorus procedure by heating with acid and persulfate. The reactive phosphorus procedure measures only the reactive phosphorus that is in the sample. The reactive or orthophosphate ions react with molybdate and antimony ions in an acidic solution to form an antimonyl phosphomolybdate complex, which is reduced by ascorbic acid to phosphomolybdenum blue. The measurement wavelength is 714 nm.

Ortho phosphate (PO_4^{3-}) was analyzed by acid ascorbic method (APHA, 1995). Orthophosphate reacts with molybdate in an acid medium to produce a mixed phosphate/molybdate complex. Ascorbic acid then reduces the complex, which gives an intense molybdenum blue color. The measurement wavelength is 710 nm for spectrophotometers.

3.4.3.3. Silica

Dissolved silica (DSi) was analyzed by silicomolybdate method (APHA, 1995). Silica and phosphate in the sample react with molybdate ion under acidic conditions to form yellow silicomolybdic acid complexes and phosphomolybdic acid complexes. Addition of citric acid destroys the phosphate complexes. Silica is then determined by measuring the remaining yellow color. The measurement wavelength is 452 nm for spectrophotometers.

3.4.4. Organic matter

Total organic carbon analyzer apparatus (TOC-V Shimadzu) was used to determine dissolved organic carbon (DOC) in water samples by using NPOC measurement (TOC by acidification/sparging method) (Sugimura and Suzuki, 1988). The key point about a TOC analyzer is whether it can efficiently oxidize hard-to-decompose insoluble and macromolecular organic compounds, not just the easily decomposed, low molecular weight, organic compounds. The 680 °C combustion catalytic oxidation method, that was developed by Shimadzu and subsequently spread around the world, can efficiently analyze all organic compounds. A small amount of hydrochloric acid is added to acidify the sample and the sample is sparged with gas. This converts all inorganic carbon (IC) in the sample to carbon dioxide and drives the CO_2 out of the sample solution. The TOC concentration is determined by measuring the total carbon (TC) of the sample after the IC is eliminated. NPOC measurement is equivalent to TOC measurement using acidification and sparging (IC elimination) defined for official test methods (EPA, ASTM, EN, etc.). As any purgeable organic compounds may be lost from the sample during sparging, the TOC measured by this method can be called NPOC (Non-Purgeable Organic Carbon).

Particulate organic carbon (POC) was performed on suspended matter retained on a GF/F filter (ignited at 550 °C). POC was measured by using combustion in a LECO CS 125 analyzer (Etcheber et al., 2007). POC is measured by determining mass lost upon combustion of a sample. In aqueous

samples, this can be done by measuring the dry mass of a filter that had a known amount of water passed through it before and after it is subjected to combustion via heating the filter to 550 °C. This method requires that the filter is purged of extraneous POC before filtration (by combusting it at 550 °C for 4 hours), and that the filter and sample are dried at 50 °C during 24h. The method also requires that the sample has a measurable amount of organic carbon present. POC in sediment samples can also be measured by mass loss by measuring the dry weight of a given volume of sample before and after combustion. These methods assume that the mass loss is attributable solely to carbon, rather than any other sample component. These experiments were realized in Laboratoire d'Environnements et Paléoenvironnements Océaniques et Continentaux (Laboratory EPOC, Université de Bordeaux, France).

3.4.5. Particulate nutrients in deposit and suspended sediment

Phosphorus. Total particulate phosphorus (TPP) concentration in sediments was measured using a high temperature/HCl extraction technique (Némery and Garnier, 2007) prior to phosphate measurement by colorimetric method (Murphy and Riley, 1962). Deposit or suspended sediment was weighted (about 1.5 g) and put into aluminum pan; then mineralized in oven at 500 °C during 12h. After that, about 1.0 g of mineralized sediment was added into plastic tube with 30 mL HCl 1N and shake slightly in 12h. This solution was centrifuged at 7000 rpm in 10 minutes; and 1 mL of solution was diluted with 50 mL miliQ water and measurement PO_4^{3-} concentration.

The different weight of sediment before and after mineralization was used to calculate percentage of organic matter (%OM). To calculate particulate inorganic phosphorus (PIP), the measurement was similar to procedure for analysis TPP, except the step of high temperature organic phosphorus mineralization. Particulate organic phosphorus (POP) was determined by subtracting particulate inorganic phosphorus concentration from total particulate phosphorus concentration ($[\text{POP}] = [\text{TPP}] - [\text{PIP}]$).

Carbon and nitrogen. 25 mg of sediment were weighted and used to measure particulate carbon and nitrogen by using Thermal Scientific FlashEA Nitrogen and carbon analyzer for soils, sediments and filters at LECA laboratory, Grenoble, France. The FlashEA 1112 series is based on the well-known Flash Dynamic Combustion method, which produces complete combustion of the sample within a high temperature reactor, followed by an accurate and precise determination of

the elemental gases produced. The method is rapid, less than 5 minutes for a complete sample characterization, and easily automated with solid or liquid auto-samplers. Suitable for organic or inorganic samples, it takes advantage of the generation (for a few seconds) of a very high temperature into the oxidation reactor (about 1800°C), and allows the complete conversion of all samples, even those containing refractory material, to elemental gases. A dedicated chromatography column connected to a highly sensitive thermal conductivity detector ensures a wide detection range for both Carbon and Nitrogen determinations. The GC separation column is an open window on the analytical system; in fact the chromatogram (peak shape, peak separation, peak retention time etc.) demonstrates the real status of the instrument in every condition.

3.4.6. Phytoplankton identification and counting

Prior to phytoplankton counting, 1 liter of water sample was settled at least 48 h in the laboratory (Sournia, 1978). The supernatant (top water) was removed and the settling material including phytoplankton was transferred into a measuring beaker and concentrated to 10 – 30 mL depending on the amount of settling material. Phytoplankton were observed at 100 – 1000 x magnification on a microscope and identified to species or genus levels based on morphology following the system of Komárek and Anagnostidis (1989, 1999, 2005) for cyanobacteria, Krammer and Lange-Bertalot (2004) for diatoms, and other taxonomy books for green, golden algae, dinoflagellates and euglenoids. Phytoplankton were enumerated with a Sedgewick Rafter counting cell (volume of 1 mL; PYSER-SGI, England) with total number phytoplankton from 400 individuals or more for each sample (Sournia, 1978). An individual of phytoplankton is defined as a single cell, a trichome or a colony of phytoplankton as commonly occurred in nature.

3.4.7. Physical characteristic of sediment

In estuaries, Total suspended solid (TSS) play an important role in both adsorption/desorption mechanism and mineralization. To assess the impact of sediment on these two mechanisms, we conducted laboratory experiments under controlled conditions of turbulence, salinity, turbidity and TSS. A portable granulometer of LISST XR (Sequoia ©) system was used to characterize Particle Size Distribution; and a new device, System for the Characterization of Aggregates and (SCAF®, Gratiot et al., 2015) (Figure 2-10) was developed in our laboratory and used to assess settling velocity and flocculation of particles.

All experiments were conducted in a rectangular tank filled with 2 liters of samples. The fluid sediment mixture was immediately mixed by the rotation of a propeller at a specific speed (100 rpm) during 30 min to ensure that the particles were in equilibrium with the hydrodynamic conditions (Gratiot et al., 2017), i.e., that flocculation/deflocculation processes were nearly at equilibrium. For each sample, we measured the aggregated particle size distribution (aPSD) with LISST-XR system and the dispersed particle size distribution (dPSD) after two minutes of ultrasonic bath. Sedimentation was also characterized with SCAF instrument. From the analysis of sedimentation, we deduced settling velocity and flocculation index. After 24h of deposition, the sample was mixed again and measured a second time with SCAF to assess the effect of deposition-re-suspension cycle on the characteristics of suspended particles. And the third sample was characterized sedimentation after seven days of deposition.

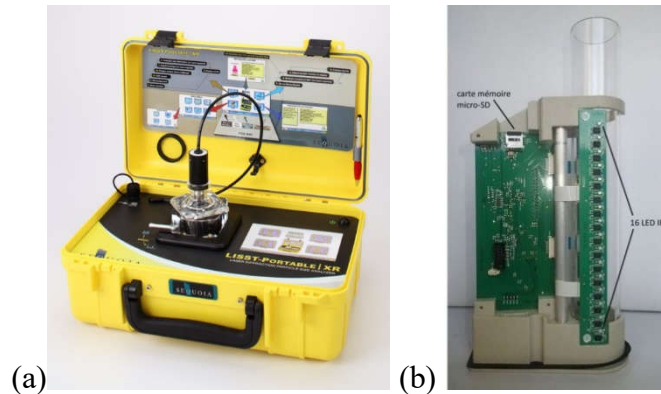


Figure 2-10. (a) The portable granulometer of LISST XR system and (b) system for characterization of aggregates and flocs (SCAF).

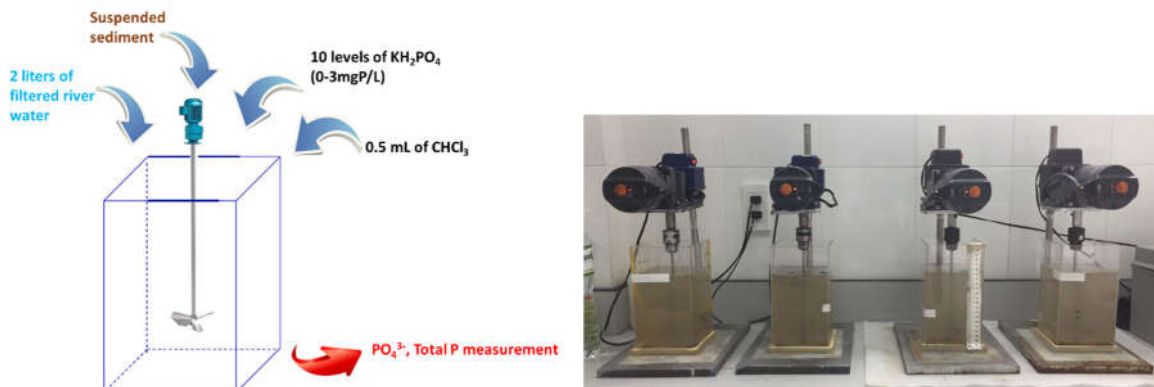


Figure 2-11. Sampling sites in Saigon – Dongnai River basin and urban canal network (source: CEM-DONRE).

4. Assessment of phosphorus mobility from laboratory experiments

4.1. Experiment design

Table 2-2. Phosphorus sorption experimental conditions

Conditions	TSS (g L ⁻¹)	Turbulence (s ⁻¹)	Salinity	PO ₄ ³⁻ standard solution (mgP L ⁻¹)
C1	0.5	44	2.7	0; 0.05; 0.1; 0.2; 0.5; 0.75; 1.0; 1.5; 2.0 and 3.0 mgP/L
C2	0.5	22	2.7	
C3	0.25	44	2.6	
C4	0.25	22	2.6	
C5	0.5	44	9.3	
C6	0.5	22	9.3	
C7	0.9	44	2.7	
C8	0.9	22	2.7	

To characterize the adsorption/desorption processes of phosphorus under contrasted conditions of salinity, turbulence and TSS concentration, we undertook some Jar test experiments. These experiments could, in a way, reproduce the biogeochemical processes in the estuary. To simulate the real conditions, water sample in Saigon River was collected and settled for 24 hours to remove suspended solid by filtered through GF/F filter (porosity 0.7 μm), and then stored in cool room until using for experiments. Then different levels of phosphate standard solutions (KH₂PO₄ – from 0 to 3.0 mgP L⁻¹, see Table 2-2) were added into 2 liters of Saigon River water sample (Figure 2-11). The suspended sediment obtained from salinity profile surveys was added to reach the concentration of 0.25 to 0.9 g L⁻¹. Pure chloroform (0.5 mL) was added to all the samples to stop any possible biological activity. These duplicate Jar tests were stirred during four hours at room temperature. Equilibrium was generally reached after four hours. After this, the solutions were

filtered through a 0.45 μm porosity/25mm diameter GF/F syringe filter to determine PO_4^{3-} concentration and unfiltered sample was analyzed Total P. All experimental samples were also submitted to further hydro-sedimentary measurements to determine the characteristics of sediment (flocculation index and particle size distribution) with SCAF and LISST systems. In total 80 experiments were realized, each of them being duplicated.

4.2. Sorption isotherm

Langmuir isotherm was used to evaluate the adsorption and desorption of P onto sediment (Rossi et al., 2012; Vilmin et al., 2015; Aissa Grouz et al., 2016; Wang and Li, 2010); the adsorbed P onto suspended sediment is characterized by an equilibrium law:

$$\frac{PIP}{[SS]} = P_{ac} \cdot \frac{[SRP]}{[SRP] + K_{ps}} \quad (1)$$

In which, PIP is particulate inorganic phosphorus (mgP L^{-1}); SS is suspended sediment concentration (mg L^{-1}); SRP is soluble reactive phosphorus concentration or PO_4^{3-} concentration (mgP L^{-1}); P_{ac} is the maximal absorption of P onto suspended sediment (mgP gSS^{-1}); K_{ps} is the half-saturation concentration of PO_4^{3-} (mgP L^{-1}). The concentration of PIP can be estimated by subtracting the content of total phosphorus (TIP) to SRP in the water ($PIP = TIP - SRP$). Replace the TIP value to the equation (1) provides a new formulation of SRP concentration:

$$SRP = \frac{TIP - Pac \cdot SS - Kps + \sqrt{(-TIP + Pac \cdot SS + Kps)^2 + 4 \cdot TIP \cdot Kps}}{2} \quad (2)$$

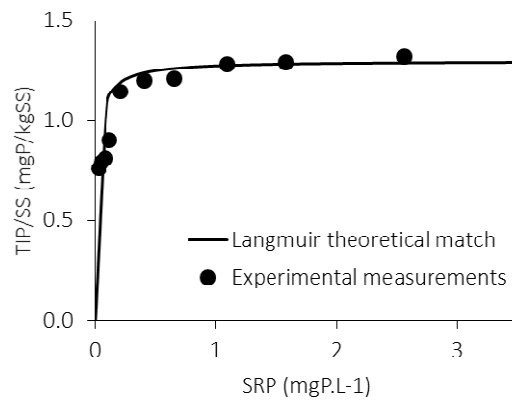


Figure 2-12. Fitting curve to obtain adsorption characteristics.

Phosphorus sorption isotherms are used to identify the characteristic of sediment samples by their P sorption and also to compare the sorption capacity at different conditions of SS concentration, turbulence and salinity in this study. Two parameters (P_{ac} and K_{ps}) of the Langmuir isotherm were determine by fitting relation (2) to the measured SRP vs TIP, using the criterion of least square deviation (Aissa-Grouz et al., 2016). An example of the result is shown in Figure 2-12.

CHAPTER 3. NUTRIENT DYNAMICS AND EUTROPHICATION ASSESSMENT IN THE TROPICAL RIVER SYSTEM OF SAIGON – DONGNAI (SOUTHERN VIETNAM)

This section is copied from the paper that was published in Science of the Total Environment:

Tuyet T.N. Nguyen ^{a,b,*}, Julien Némery ^{a,b}, Nicolas Gratiot ^{a,b}, Emilie Strady ^{a,b}, Viet Quoc Tran ^b, An Truong Nguyen ^b, Joanne Aimé ^c, Alice Peyne ^b (2019). Nutrient dynamics and eutrophication assessment in the tropical river system of Saigon – Dongnai (Southern Vietnam), Science of the Total Environment (653) 370-383.

Results of the bi-monthly monitoring from July 2015 to December 2017 and the longitudinal profile campaigns during dry season (April 2017) and wet season (October 2017) were used to characterize nutrients dynamics in the Saigon – Dongnai River system, evaluate the eutrophic status of water bodies and determine the controlling factors of eutrophication and the global impact of HCMC on water quality.

NUTRIENT DYNAMICS AND EUTROPHICATION ASSESSMENT IN THE TROPICAL RIVER SYSTEM OF SAIGON – DONGNAI (SOUTHERN VIETNAM)

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Abstract

Saigon-Dongnai Rivers in Southern Vietnam is a complex lowland hydrological network of tributaries that is strongly influenced by tidal cycles. The increasing economic, industrial and domestic developments in and around Ho Chi Minh City (HCMC) have led to serious impacts on water quality due to lack of appropriate wastewater treatment. Drinking water production is impacted and the large aquaculture production areas may also be affected. We analyzed spatial and seasonal variability of nutrient concentrations (Phosphorus, Nitrogen and Silica) and eutrophication indicators (Organic Carbon, Chlorophyll-a and Dissolved Oxygen) based on bi-monthly monitoring during two hydrological cycles (July 2015 – December 2017). Four monitoring sites were selected to assess the impact of HCMC: two upstream stations on the Saigon River and Dongnai River branches to provide the reference water quality status before reaching the urbanized area of HCMC; one monitoring station in the city center to highlight Saigon River water quality within the heart of the megacity, the fourth station downstream of the confluence to evaluate the impact of HCMC on estuarine waters. This study points to excess nutrients in HCMC's water body with concentrations of NH_4^+ and PO_4^{3-} averaging to $0.7 \pm 0.6 \text{ mgN L}^{-1}$ and $0.07 \pm 0.06 \text{ mgP L}^{-1}$, respectively, in mean over the monitored period and rising up to 3 mgN L^{-1} and 0.2 mgP L^{-1} , in extreme conditions. During the dry season, we evidenced that untreated

domestic discharge leads to degradation of the Saigon River's water quality with extreme values of algal biomass (up $150 \mu\text{Chl-a L}^{-1}$) and hypoxic conditions occurring episodically ($\text{DO} < 2 \text{ mg L}^{-1}$) in the heart of the megacity. Until now, eutrophication in the urban center has had no clear effect downstream because eutrophic water mass from the Saigon River is efficiently mixed with the Dongnai River and sea water masses during the semi-diurnal tidal cycle.

Keywords. Water quality, nutrients, megacity, eutrophication, tidal river.

1. Introduction

In the past few decades, eutrophication has become a significant issue for surface water bodies' worldwide (Hilton et al., 2006; Romero et al., 2013; Smith and Schindler, 2009; Smith et al., 1999). Eutrophication is an accelerated growth of algae caused by the enrichment of nutrients in an aquatic system, especially nitrogen (N) and phosphorus (P); which results in oxygen depletion in the water body through intense bacterial mineralization (Billen and Garnier, 1997; Nixon, 2009; Volterra et al., 2002). Five decades ago, Redfield et al. (1963) proposed the Redfield stoichiometric ratio of carbon, nitrogen and phosphorus (C:N:P) as a generic tool to assess which is the limiting factor for primary production. The N:P ratio in an aquatic system is an important key which is used to verify which element will become the limiting factor to be controlled to reduce algal blooms. More recently, Billen and Garnier (2007) improved the concept to address the question of coastal waters. They showed that eutrophication is the consequence of unbalanced riverine nutrient inputs, with excess nitrogen and phosphorus with respect to silica (Si), when compared to the requirements for algal growth; stated to be C:Si:N:P = 106:15:16:1 (Brzezinski, 1985). The enrichment of riverine waters in N and P, together with decreasing suspended solids and Si, often result in eutrophication of coastal areas (Billen and Garnier, 1997; Conley et al., 1989; Howarth et al., 2011), characterized by non-diatom harmful algal blooms. The new production of non-siliceous algae can be facilitated by the imbalance between nutrient inputs.

Nutrients (N, P) generally originate from over fertilized agricultural soils through leaching and erosion processes (Cloern et al., 2014; Lau and Lane, 2002; Singkran, 2017; Trinh et al., 2015) or from untreated domestic and industrial wastewaters (Billen et al., 2007; Cloern et al., 2014; Trinh et al., 2007). In industrialized European and North American industrialized countries, wastewater treatment started in the 1970s in order to reduce the point sources of N and P towards the hydrosystem (Van Drecht et al., 2009). While the reduction of point sources pollution was shown to be efficient, diffuse inputs remained a worrying source of nutrients for watersheds in these countries (Grizzetti et al., 2013). The situation described for industrialized countries described above is very different to that of emerging countries, especially for those located in the tropical regions, such as Vietnam (Le et al., 2015). With the onset of the 21st century, the Southeast Asian countries faced significant environmental challenges, such as quantitative shortages and qualitative deterioration of water resources due to rapid industrialization, urbanization, and

agricultural intensification (Le, 2007; Vo, 2007). Indeed, sanitation network development in large cities does not provide sufficient support to cover population growth (Trinh et al., 2012). As a result, domestic wastewaters are released without appropriate treatment into the receiving aquatic environments, causing high levels of organic and nutrient pollution. In parallel with urban growth, changes in agricultural practices have involved an increasing supply of fertilizers (Garnier et al., 2015). The change in land-use have led to an increase of nutrient inputs to surface water and of the flows to the outlets of major watersheds (e.g., the Mekong River). Moreover, high rainfall intensity in the tropics leads to significant erosion rates and associated nutrient flows into the rivers (Syvitski et al., 2014). The seasonality of precipitation, with marked alternation between the dry and wet seasons, influences the hydrological regime and promotes eutrophication during the dry season (Boulton et al., 2008).

The Saigon River is the second largest river after the Dongnai River in supplying water to Ho Chi Minh City (HCMC), the economic capital of Vietnam. Due to increasing urban sprawl in HCMC and neighboring provinces situated near the Saigon River basin, the water quality has significantly deteriorated (Nguyen et al., 2011; Strady et al., 2017; Tran Ngoc et al., 2016; Vo, 2007). Water supply demand from the Saigon River was estimated as 64 % (1.6 million m³ day⁻¹) for households in 2009 and will increase to 72 % (3.4 million m³ day⁻¹) according to future projections for 2025 (Nguyen et al., 2011). In addition, water from the Saigon River water is also used for irrigation, aquaculture and navigation.

The objectives of this study were to (i) characterize nutrients dynamics in the Saigon – Dongnai River system to evaluate the eutrophic status of water bodies and (ii) determine the controlling factors of eutrophication and the impact of HCMC on water quality downstream. We implemented a bi-monthly monitoring sampling strategy from July 2015 to December 2017 to determine and analyze physico-chemical parameters and nutrients (N, P and Si) in the dissolved and particulate phase.

2. Materials and methods

2.1. Study area

The Saigon River is a part of the Saigon – Dongnai River basin and located in Southern Vietnam. It is about 250 km long with a catchment area of 4,717 km² down to the confluence with the Dongnai River (Figure 3-1a). Upstream of the Saigon River, the Dau Tieng reservoir (surface area of 270 km² and storage capacity of 1,580×10⁶ m³) was constructed in 1985 for irrigation, flood protection purposes and control of saline water intrusion (Trieu et al., 2014). The Saigon River is connected to urban canals when flowing through HCMC and then joins the Dongnai River to become the Nha Be River. It then continues its flow through Can Gio Mangrove to the East Sea. The flow direction of the Saigon River is strongly affected by the asymmetric semi-diurnal tides and the discharge is mainly controlled during the dry season by the amount of water released from the Dau Tieng reservoir.

The region falls in a tropical monsoon climate, with an average humidity of 78-82 % and a mean air temperature of 28.5 ± 1.0 °C. The year is divided into two distinct seasons (wet and dry). The wet season has an average rainfall of about 1,800 mm annually (about 150 rainy days year⁻¹), usually beginning in May and ending in November (Figure 3-1b). The dry season lasts from December to April.

Ho Chi Minh City is intrinsically linked with the Saigon River. The megacity's economy has developed considerably over the last ten years and HCMC is now considered the second most dynamic city in the world (World Economic Forum News, January 2017). HCMC resides on the banks of the Saigon River in a lowland floodplain area of the Saigon-Dongnai River basin and belongs to a transitional area between the South-eastern region of Vietnam and the Mekong Delta.

Land use in HCMC is dominated by agricultural activities in the north, urban settlement in the center and Can Gio Mangrove forest to the south. Can Gio forest is located 40 km southeast of HCMC and is recognized as a biosphere reserve by UNESCO. Agricultural land in HCMC was around 32 % in 2016, showing a decrease of 2 % in comparison to 2012 (data from HCMC Statistical Yearbook 2016).

The population of HCMC was 8.4 million inhabitants in 2016, which represents about 8.9 % of Vietnam’s total population. This makes HCMC the largest and most densely populated city in the country (Figure 3-1c). The population of HCMC is mainly concentrated in the heart of the city (as presented in Figure 3-1) and 6.7 million inhabitants live in the nineteen urban districts. The rural population was 1.5 million in the three districts upstream of HCMC and only 230,000 in the two downstream districts. HCMC reached an urban population proportion of 88% in 2004, since then the population of HCMC and its boundary districts has increased rapidly (data from HCMC Statistical Year Book 2016).

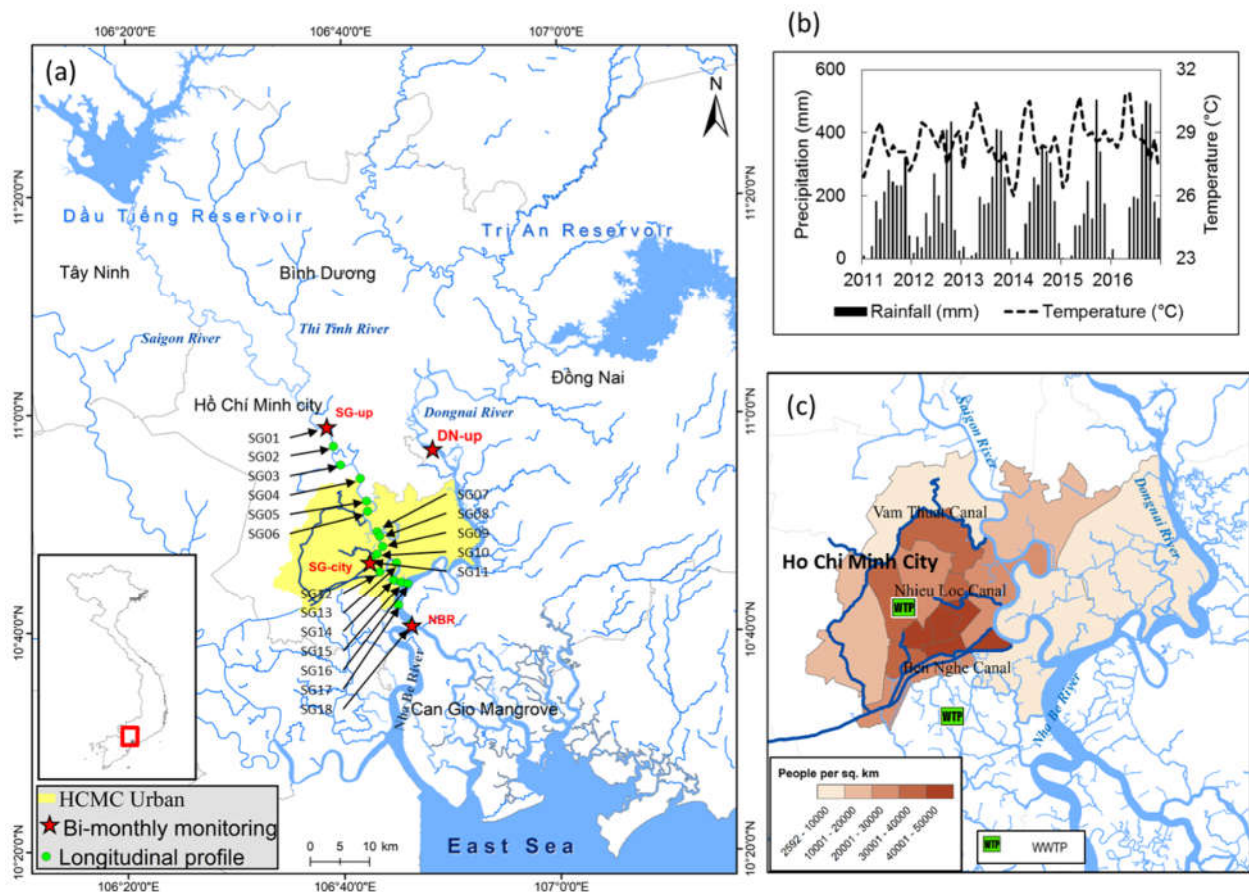


Figure 3-1. (a) Map of Saigon – Dongnai Rivers and location of sampling sites, (b) distribution of monthly rainfall and temperature from 2011 to 2016, (c) population density distribution in Ho Chi Minh City (source: HCMC Statistical Year Book 2016).

Less than 10 % of the domestic waste water is collected and treated before being discharged directly into urban canals or rivers (Marcotullio, 2007). The main wastewater treatment plant (Binh Hung WWTTP) is located south of the high population density urban area (Figure 3-1c). Its

treatment capacity is $141,000 \text{ m}^3 \text{ day}^{-1}$ (426,000 inhabitants). Construction began in 2017 to reach a total capacity of $469,000 \text{ m}^3 \text{ day}^{-1}$ (1,390,000 inhabitants) and authorities have scheduled the building of ten new WWTPs within the next ten years (source: ATLAS Ho Chi Minh City).

2.2. Monitoring and sampling strategy

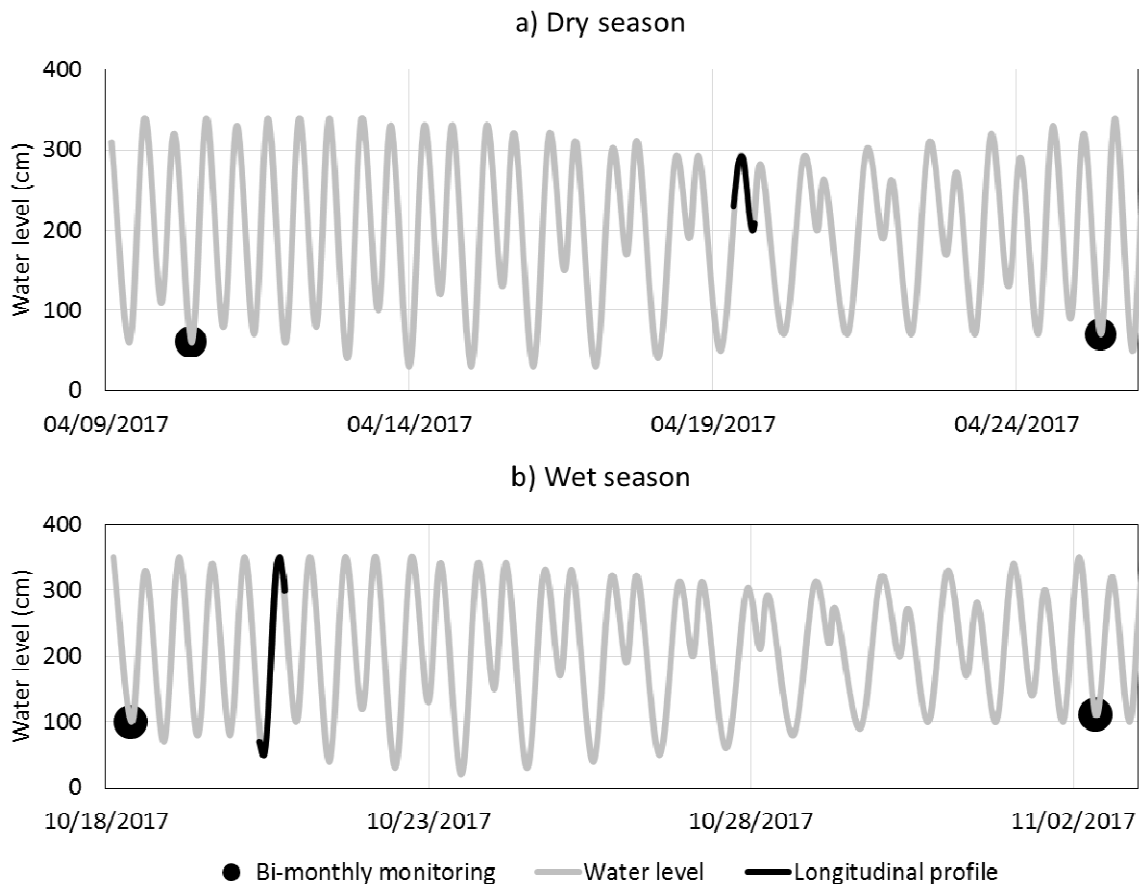


Figure 3-2. Water level during (a) the dry season (April 2017) and (b) the wet season (October 2017) at HCMC (source: tide4fishing.com; black points indicate bi-monthly monitoring and black lines correspond to the time period during which longitudinal profiles were carried out).

A bi-monthly monitoring was undertaken at four sites from July 2015 to December 2017 in the Saigon – Dongnai River branches (Figure 3-1a). Two upstream stations on Saigon River (SG-up) and Dongnai River (DN-up), respectively, characterize the reference water status in the upstream of HCMC. One monitoring station in the city center (SG-city) aimed at highlighting the impact of the city. The last station was placed downstream of the confluence between the two rivers (Nha Be River – NBR) to assess the impact of HCMC on estuarine waters. Water sampling was taken

at low tide for each field survey to consider the dominant fluvial flow from upstream to downstream and is illustrated in Figure 3-2.

Two complementary longitudinal profiles were carried out during the wet and dry seasons to understand the spatial fluctuation of water quality in the Saigon River. Total profile length was 50 km with 18 sampling points (Figure 3-1a). The profiles were conducted on-boat on 19th April 2017 from 8:00am to 4:30pm (dry season) and 20th October 2017 from 9:30am to 6:00pm, starting at Saigon upstream (SG-up – SG01) and ending at Saigon downstream (NBR – SG18) (Figure 3-1a). A GPS was used to mark longitude and latitude values of each point and to calculate the corresponding kilometric point (pK), in which pK = 0 was the confluence between the Saigon and Dongnai Rivers, with negative values upward and positive downward.

For each sample, surface water (0-30cm below the surface) was taken by using a 2.5 L Niskin bottle in the middle of the river from either a bridge or a boat. Immediate measurements of physico-chemical parameters were realized using a multi-parameter probe (WTW 3420[®]), e.g., temperature, pH, conductivity, salinity, and dissolved oxygen concentration and percentage of oxygen saturation. Each water sample was collected in a 5 L polypropylene recipient for Total Suspended Sediment (TSS), C, N, Si, and P measurements and Chlorophyll-*a* analysis.

2.3. Laboratory analyses

Samples were immediately filtered through a Whatman GF/F filter (porosity 0.7 μ m) at the Centre Asiatique de Recherche sur l'Eau (CARE) laboratory for analysis of dissolved nutrients. Unfiltered samples were kept to measure total nutrients. TSS was measured based on the different weight of a GF/F filter after 24 hours of drying at 50 °C. Chlorophyll-*a* and Phaeopigments were measured after filtration through a second Whatman GF/F filter using acetone (90 %) extraction method followed by a spectrophotometry measurement at 665 and 750 nm absorbance (Aminot and K erouel, 2004).

Unfiltered water samples were used to measure Total N and Total P using persulfate digestion procedure and standard colorimetric method (American Public Health Association: APHA, 1995). Dissolved nutrients were analyzed on filtered water samples using standard colorimetric methods (APHA, 1995). The NH₄⁺, NO₃⁻, PO₄³⁻ and Dissolved Si (DSi) were analyzed by using respectively the Nessler reactive method, the cadmium reduction method, the acid ascorbic method and the

silicomolybdate method, respectively. Reproducibility for replicate measurements was better than 5 % for all total and dissolved nutrients.

A TOC (Total Organic Carbon) analyzer apparatus (TOC-V Shimadzu, CARE laboratory, Vietnam) was used to determine Dissolved Organic Carbon (DOC) in filtered water samples with accuracy higher than 5 % (Sugimura and Suzuki, 1988). Particulate Organic Carbon (POC) was measured on suspended matter retained on a GF/F filter (ignited at 550 °C). The POC content (in % of dry weight) was measured by using combustion in a LECO CS-125 analyzer (Laboratory EPOC, Université de Bordeaux, France) with precision better than 5 % (Etcheber et al., 2007). The POC concentration is calculated as the product of POC % and TSS concentration and expressed in mgC L⁻¹.

2.4. Hydrological database

Water discharges data was provided by the Center of Environmental Monitoring (CEM) of the Department Of Natural Resources and Environment of HCMC (DONRE). This national monitoring program allows the measuring of monthly discharges at different locations within our study sites. Discharges are measured every hour during a 24-hour cycle because the hydrodynamic effect of the tide is perceptible up to the dams (Figure 1a). The residual discharge is then calculated as the difference between positive and negative discharges over the tide cycle. The residual discharge corresponds to the net positive water flow of the river from land to the sea. Data was collected for the 2012-2016 period in order to characterize the mean monthly distribution of the discharge at two locations in the Saigon River (SG-city) and in the Dongnai River (DN-up) (Figure 1a).

3. Results

3.1. Seasonal hydrological variability

Water discharge distribution is mainly controlled by precipitation seasonality (Figure 3-1b). The wet season shows a water discharge increase from June to October followed by a decrease at the end of November in both Saigon and Dongnai Rivers (Figure 3-3). The mean inter-annual discharge calculated over the 2012-2016 period for the Saigon River ($50 \pm 20 \text{ m}^3 \text{ s}^{-1}$) is twelve times lower than for the Dongnai River ($613 \pm 220 \text{ m}^3 \text{ s}^{-1}$). The Saigon River discharge can be

artificially controlled during the dry season by releasing water from Dau Tieng reservoir to flush out salt intrusion. Trieu et al. (2014) suggest that water release regulation from the reservoir can be up to $30 \text{ m}^3 \text{ s}^{-1}$ during the dry season. This is in good agreement with the field data provided by DONRE.

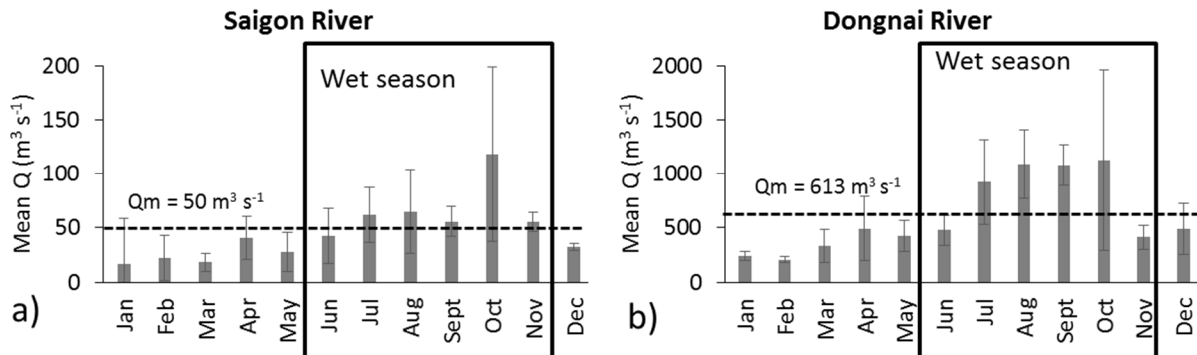


Figure 3-3. Distribution of mean monthly discharge (period 2012-2016) in Saigon River and Dongnai River (data source: DONRE; see Figure 3-1a for the location).

3.2. Interannual variations of water quality in the Saigon River

3.2.1. Physico-chemical parameters

Water temperature in the Saigon and the Dongnai Rivers ranged between 27 to 33 °C, with low fluctuations. Mean value was 30.0 ± 1.2 °C at four stations. As shown in Figure 3-4a, pH increased from upstream to downstream in the Saigon River (6.3 ± 0.2 and 6.8 ± 0.3 at SG-up and NBR, respectively), while mean pH in DN-up was 7.0 ± 0.3 . For salinity, values were almost zero in SG-up and DN-up, whereas higher values and wider ranges of seasonal fluctuations were observed in SG-city and NBR (the highest value observed was 5.0 in SG-city and 8.7 in NBR during the 2016 dry season – Figure 3-4b). Dissolved oxygen (DO) fluctuated at the four stations without specific variability and reached its highest value at DN-up (7.96 mg L^{-1}) and the lowest value at SG-city (0.57 mg L^{-1}) (Figure 3-4c). Average DO concentrations were $6.1 \pm 0.6 \text{ mg L}^{-1}$, $2.5 \pm 0.9 \text{ mg L}^{-1}$, $1.7 \pm 0.8 \text{ mg L}^{-1}$ and $3.6 \pm 1.0 \text{ mg L}^{-1}$ at DN-up, SG-up, SG-city and NBR, respectively. The TSS concentrations evidenced seasonal variations at SG-up and DN-up, (Figure 3-4d) and reached their highest values during the wet season at SG-up and DN-up (140 mg L^{-1} in DN-up and 184 mg L^{-1} in SG-up during the wet season). However, the mean TSS value at SG-city ($98.5 \pm 56.7 \text{ mg L}^{-1}$)

was higher than at the other stations ($36.8 \pm 26.4 \text{ mg L}^{-1}$, $67.3 \pm 36.7 \text{ mg L}^{-1}$ and $33.1 \pm 25.6 \text{ mg L}^{-1}$ at SG-up, NBR and DN-up, respectively).

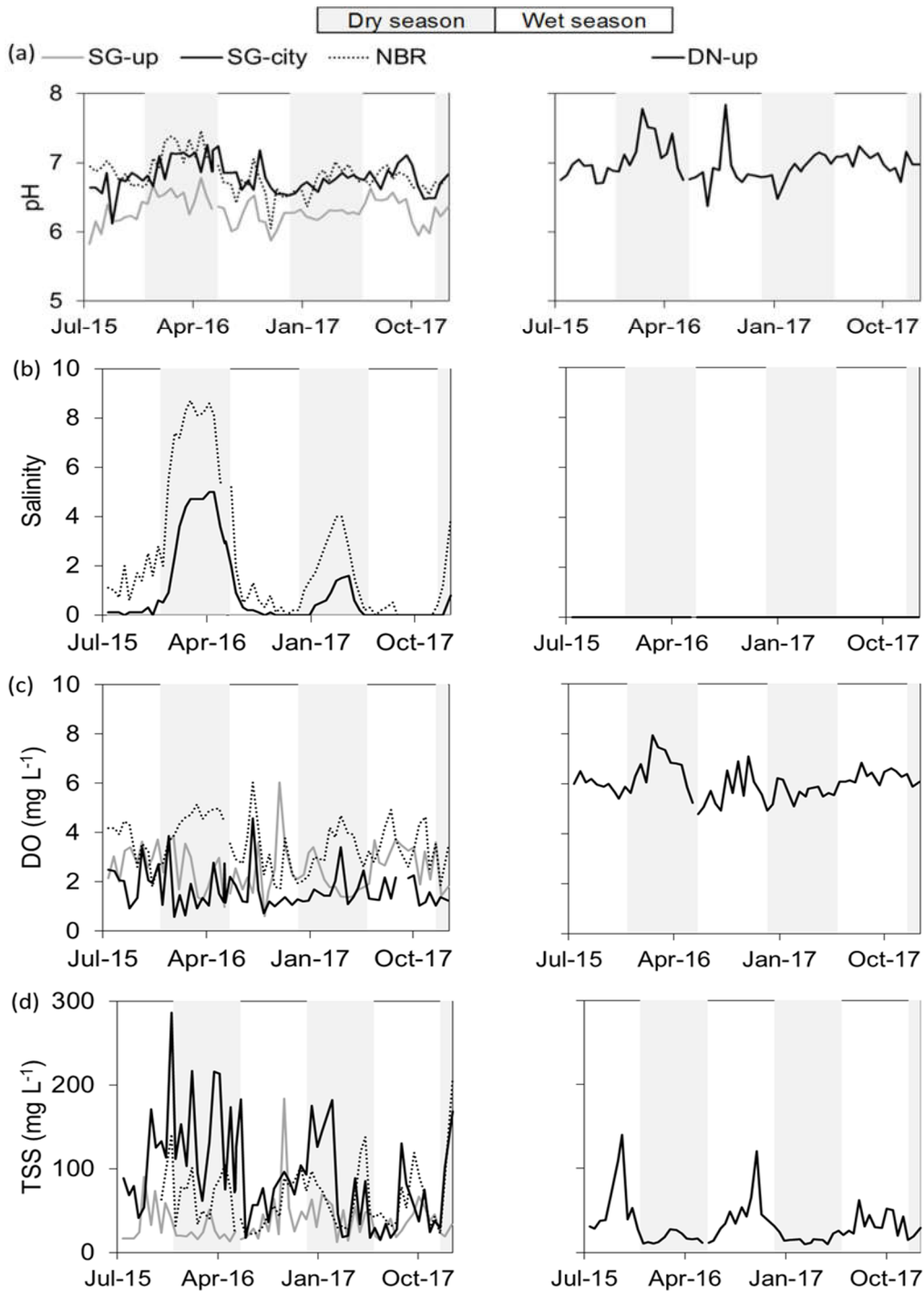


Figure 3-4. Seasonal variations of (a) pH, (b) salinity, (c) DO and (d) TSS at the four stations during the period July 2015 to December 2017. Gray color presents the dry season, which lasts from December to April and white color is the wet season from May to November.

3.2.2. Dissolved and total nutrients

Total N reached its highest value at SG-city ($3.4 \pm 0.9 \text{ mgN L}^{-1}$) and lowest at DN-up ($1.5 \pm 0.4 \text{ mgN L}^{-1}$; Figure 3-5a). Total N tends to significantly increase from upstream (SG-up) to Saigon City Center (SG-city) before slightly decreasing downstream of Saigon (NBR). NO_3^- was lower than 2 mgN L^{-1} in all samples (except one point in SG-city in July 2015), averaging $0.6 \pm 0.3 \text{ mgN L}^{-1}$, $0.7 \pm 0.5 \text{ mgN L}^{-1}$, $0.6 \pm 0.3 \text{ mgN L}^{-1}$ and $0.5 \pm 0.2 \text{ mgN L}^{-1}$ for SG-up, SG-city, NBR, and DN-up, respectively. NO_3^- concentrations at DN-up fluctuated slightly with no evident seasonality. At SG-city, NO_3^- presented much more fluctuation from 0.0 to 2.4 mgN L^{-1} (Figure 3-5b). NH_4^+ was low at DN-up ($< 0.5 \text{ mgN L}^{-1}$), but reached peak value up to over 3 mgN L^{-1} (Figure 3-5c) at SG-city. Average NH_4^+ was $0.3 \pm 0.2 \text{ mgN L}^{-1}$, $0.8 \pm 0.6 \text{ mgN L}^{-1}$, $0.1 \pm 0.1 \text{ mgN L}^{-1}$ and $0.1 \pm 0.1 \text{ mgN L}^{-1}$ at SG-up, SG-city, NBR and DN-up, respectively.

Total P and PO_4^{3-} did not present specific seasonal variations. Total P was dramatically high in SG-city, $0.3 \pm 0.2 \text{ mgP L}^{-1}$ on average and up to 0.8 mgP L^{-1} for peak values (Figure 3-5d). Average Total P was $0.2 \pm 0.1 \text{ mgP L}^{-1}$, $0.1 \pm 0.1 \text{ mgP L}^{-1}$ and $0.1 \pm 0.1 \text{ mgP L}^{-1}$ in SG-up, NBR and DN-up, respectively. The PO_4^{3-} concentration (Figure 3-5e) was low in DN-up (experienced at $0.04 \pm 0.02 \text{ mgP L}^{-1}$ on average). Mean PO_4^{3-} value was $0.04 \pm 0.03 \text{ mgP L}^{-1}$, $0.07 \pm 0.06 \text{ mgP L}^{-1}$ and $0.03 \pm 0.02 \text{ mgP L}^{-1}$ in SG-up and NBR, respectively.

A seasonal DSi concentration fluctuation was observed at the four sites from 6 to 8 mgSi L^{-1} during the wet season down to values below 1 mgSi L^{-1} during the dry season. Average DSi concentrations were $2.6 \pm 2.1 \text{ mgSi L}^{-1}$, $1.7 \pm 1.8 \text{ mgSi L}^{-1}$, $1.8 \pm 1.8 \text{ mgSi L}^{-1}$ and $2.0 \pm 1.8 \text{ mgSi L}^{-1}$ at DN-up, SG-up, SG-city and NBR, respectively (Figure 3-5f).

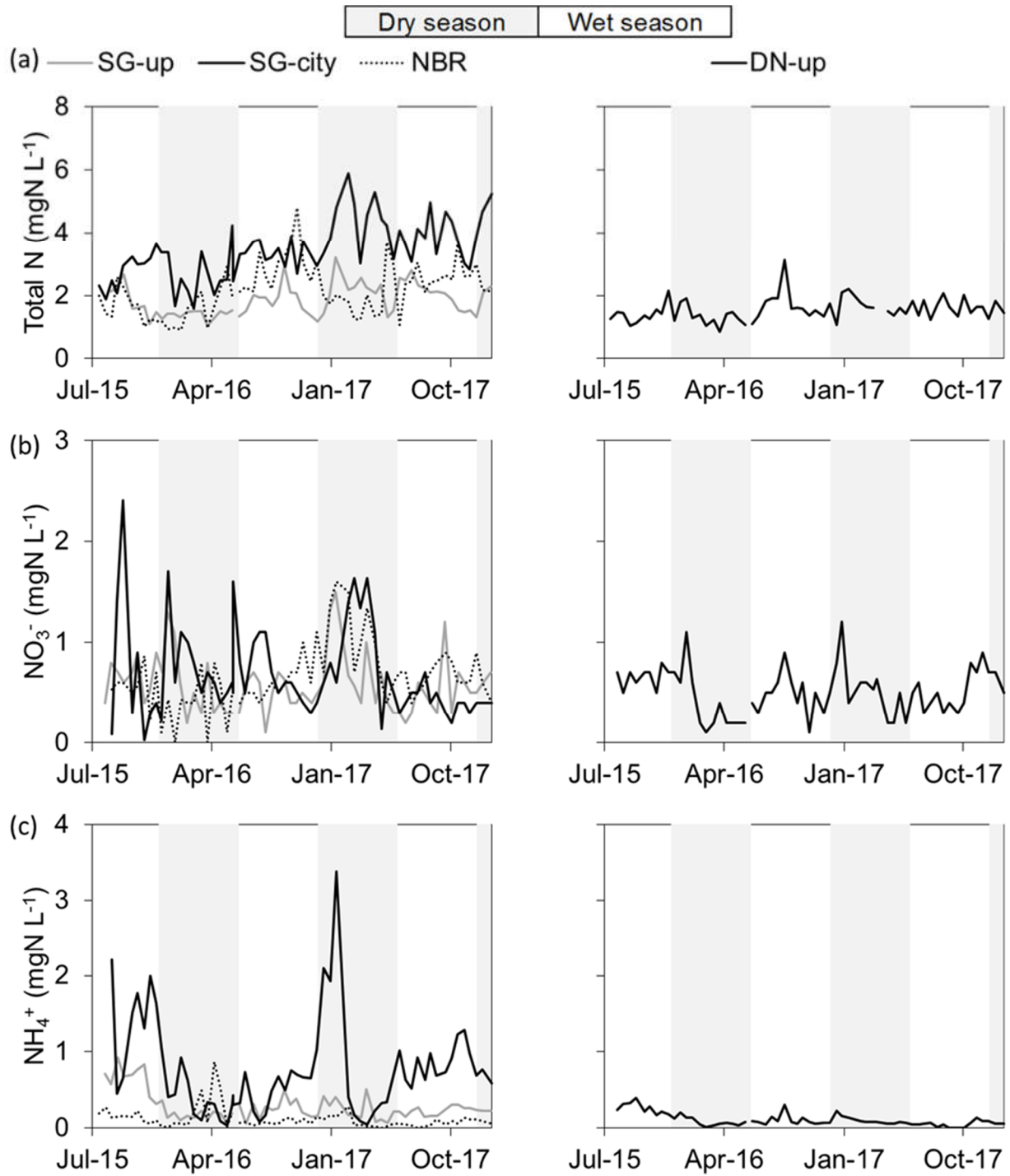


Figure 3-5. Seasonal variations of (a) Total N, (b) NO₃⁻, (c) NH₄⁺, (d) Total P, (e) PO₄³⁻ and (f) DSi measured at the four sites from July 2015 to December 2017.

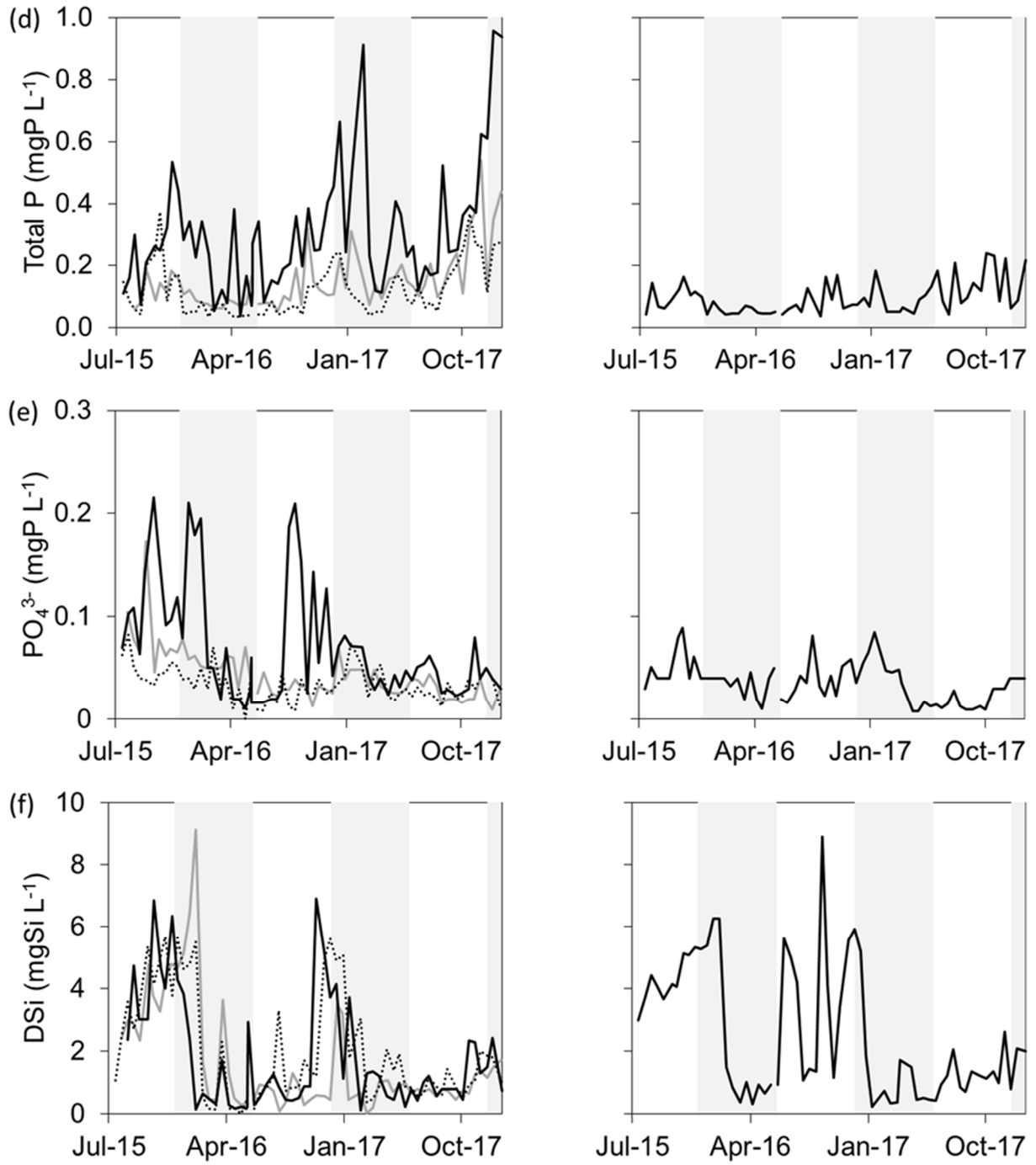


Figure 3-5. Seasonal variations of (a) Total N, (b) NO_3^- , (c) NH_4^+ , (d) Total P, (e) PO_4^{3-} and (f) DSi measured at the four sites from July 2015 to December 2017.

3.2.3. Organic carbon, chlorophyll-*a*, and phaeopigments

The DOC concentrations were $2.4 \pm 0.7 \text{ mg L}^{-1}$, $2.5 \pm 1.9 \text{ mg L}^{-1}$, $3.6 \pm 1.1 \text{ mg L}^{-1}$ and $3.8 \pm 0.7 \text{ mg L}^{-1}$ on average at DN-up, SG-up, SG-city and NBR, respectively (Figure 3-6a). As compared with other parameters measured during this study, DOC concentrations were remarkably stable at the four sites during the observation period, except in March 2017 with a peak of DOC observed in SG-up and DN-up. POC reached its highest value at SG-city, with an average of $4.3 \pm 1.6 \text{ mg L}^{-1}$ and lowest value at the other sites, making up $1.3 \pm 0.5 \text{ mg L}^{-1}$, 1.7 ± 0.8 and 1.6 ± 1.0 on average at DN-up, SG-up and NBR, respectively (Figure 3-6b).

Chl-*a* concentrations varied from $0.01 \text{ } \mu\text{g L}^{-1}$ to $147 \text{ } \mu\text{g L}^{-1}$ at SG-city (Figure 6c). Chl-*a* and phaeopigments exhibited contrasted variability both spatially and seasonally for the whole survey period. The mean value of pigments at SG-city ($29.6 \pm 32.4 \text{ } \mu\text{g Chl-}a \text{ L}^{-1}$ and $19.5 \pm 16.7 \text{ } \mu\text{g phaeopigments L}^{-1}$, respectively) was higher than at DN-up ($4.7 \pm 3.8 \text{ } \mu\text{g Chl-}a \text{ L}^{-1}$ and $5.7 \pm 5.3 \text{ } \mu\text{g phaeopigments L}^{-1}$), SG-up ($3.2 \pm 5.5 \text{ } \mu\text{g Chl-}a \text{ L}^{-1}$ and $4.9 \pm 4.8 \text{ } \mu\text{g phaeopigments L}^{-1}$) and NBR ($1.6 \pm 1.1 \text{ } \mu\text{g Chl-}a \text{ L}^{-1}$ and $2.6 \pm 1.7 \text{ } \mu\text{g phaeopigments L}^{-1}$).

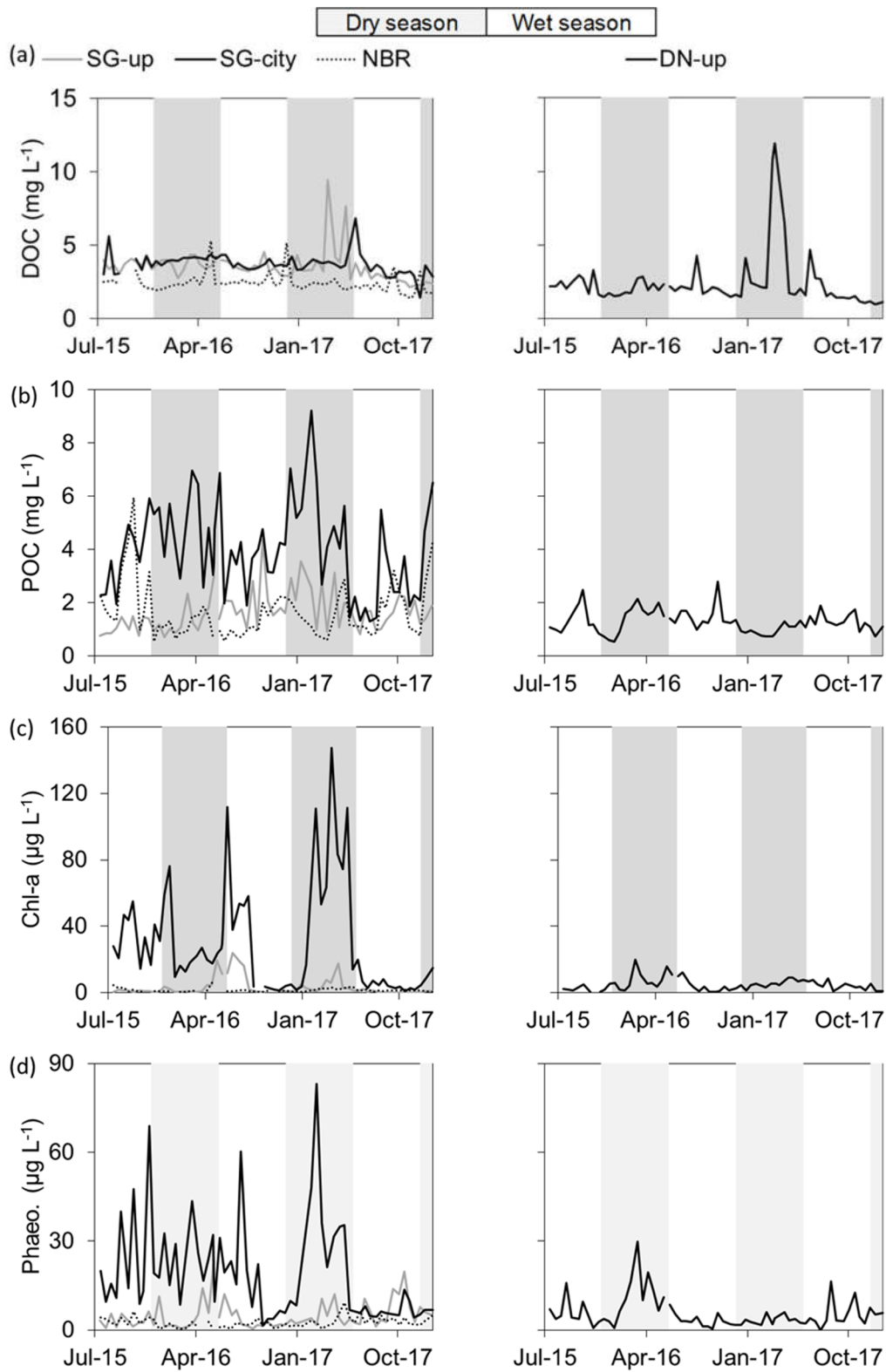


Figure 3-6. Seasonal variations of a) DOC, b) POC, c) Chl-a and d) phaeopigments measured at the four sites from July 2015 to December 2017.

3.3. Longitudinal profile in the Saigon River

The longitudinal profile was divided into three sections (i.e., was defined according to population density distribution) to assess the impact of urbanization on the Saigon River water quality: (i) the upstream section was from Phu Cuong Bridge (SG01) to Thanh Da canal (SG06), (ii) the middle section was representative of the high urban system extending from Thanh Da canal (SG07) to Bach Dang Harbor (SG11) and (iii) the downstream section was from Bach Dang Harbor (SG12) to Binh Khanh Ferry (SG18). The Saigon River water quality changed considerably between the profiles realized during the dry and wet seasons (Figure 3-7). The pH increased gradually from land to the ocean (with more acid water during the wet season from 5.8 to 6.2) than during the dry season (6.3 to 7.2) (Figure 3-7). DO values varied between moderate to severe hypoxic conditions (as low as 0.5 mg L⁻¹ in the upstream section and rose up to 4.7 mg L⁻¹ in the downstream section). During the wet season, DO increased rapidly from extremely low concentrations at upstream of HCMC (SG01 to SG06 during the wet season with DO values below 2 mg L⁻¹) to high values after City Center (point SG13, with DO values above 4 mg L⁻¹). Saline water intrusion during the dry season was observed up to about 40 km from the confluence between the two rivers. Salinity was nearly 0 during the wet season in the three sections. The magnitude of TSS fluctuations was higher during the wet season with a peak of 71.5 mg L⁻¹ at SG03, a minimum of 17.6 mg L⁻¹ at SG09 and a new maximum downstream with a value of 78.6 mg L⁻¹ at SG17.

While Total N and NO₃⁻ concentrations were stable from the upstream to downstream sections, NH₄⁺ tended to increase in the middle section during the wet season. Total P concentrations were high in the upstream and middle section before decreasing in the downstream section, especially during the dry season. The PO₄³⁻ concentrations fluctuated from the upstream to downstream sections during both seasons. The DOC, POC, Chl-*a* and phaeopigments varied in the same range. Chl-*a* content was stable and lower than 2 µg L⁻¹ for Chl-*a* and 5 µg L⁻¹ for phaeopigments during the wet season, while Chl-*a* and phaeopigments concentrations reached their highest values at the transition between upstream and middle sections during the dry season (110.3 µg Chl-*a* L⁻¹ at SG06 and 72.2 µg phaeopigments L⁻¹ at SG07). DSi values were rather low in the upstream section (1mg DSi L⁻¹) but increased locally to 2 mg DSi L⁻¹ in the middle (dry season) or downstream sections (wet season).

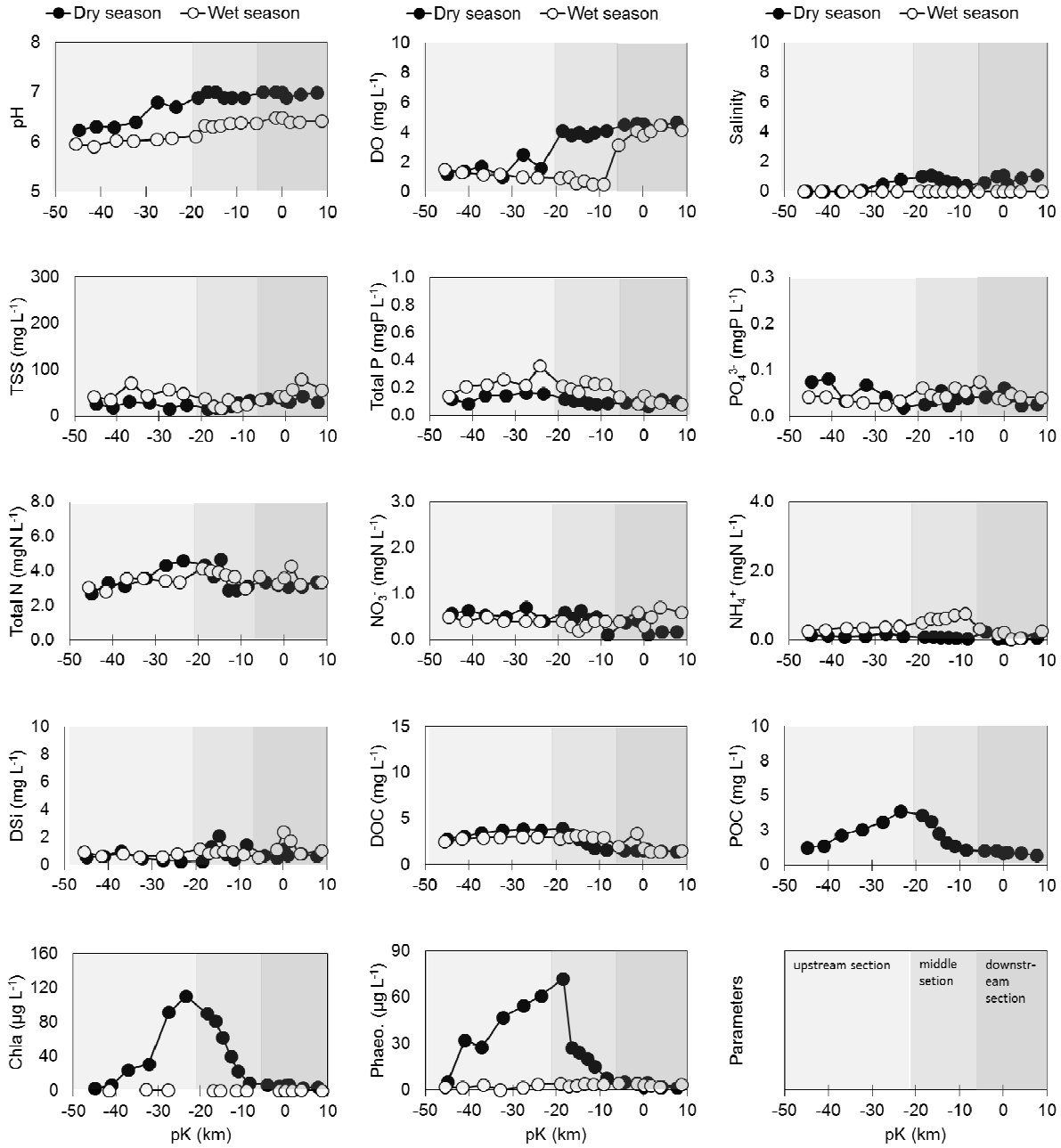


Figure 3-7. Longitudinal profile in the Saigon River during dry season (19th April 2017) and wet season (20th October 2017); from top to bottom and from left to right: pH, DO, salinity, Total N, NO_3^- , NH_4^+ , Total P, PO_4^{3-} , DSi, TSS, DOC, POC, Chl-a and Phaeopigments. Last panel represents the longitudinal profile divided into three sections.

3.4. Nutrient ratios

We have assembled a data set of dissolved nutrients (N, P and Si) concentrations at the four sampling sites in the Saigon and Dongnai Rivers. These dissolved nutrient concentrations were used to calculate the Redfield ratio (N:P:Si ratios). When the graph was plotted according to the Si:N and Si:P ratios, the theoretical lines represented a theoretical nutrients ratio and the graph was separated into four areas (Figure 3-8). Each area represented, respectively, from the lower right quadrant counter-clockwise: an area with N in limiting amounts and Si in excess; an area with N in limiting amounts and Si deficiency; an area with P in limiting amounts and Si deficiency; and an area with P in limiting amounts and Si in excess (Billen and Garnier, 2007). Figure 3-8 showed that the four sampling sites in the Saigon – Dongnai River system were distributed in overlapping clusters. From bi-monthly monitoring, we can observed that the Saigon – Dongnai River system was characterized by a N:P ratio above the Redfield value and by a substantial excess of Si over both P and N. Additionally, the molar Si:P ratio was higher than 15 and reached the maximum value of 200 (Figure 3-8), indicating that Si still exceeded diatom growth requirement, and could therefore limit the development of non-diatoms algae in this environment.

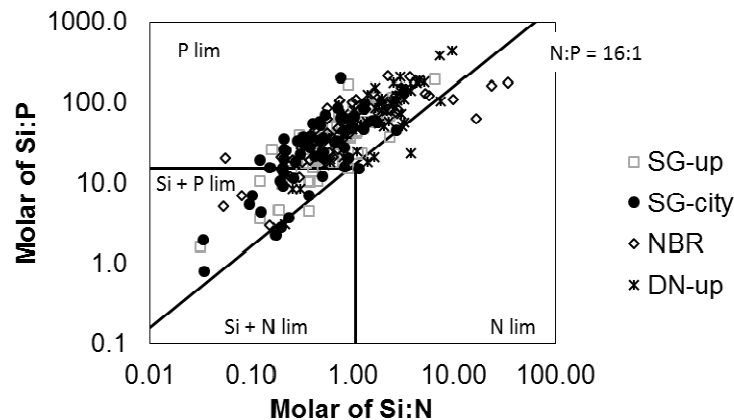


Figure 3-8. Molar ratios of Si:N and Si:P (log scale) at four sampling sites in Saigon – Dongnai River system from July 2015 to December 2017.

4. Discussion

4.1. Level of nutrient contamination

The level of nutrients measured in the Saigon and Dongnai Rivers was compared to the European standard of water quality (Water Framework Directive, 2000). The European standard of water quality has five surface water quality assessment categories (high, good, moderate, bad and very bad status). Vietnamese water quality regulation (QCVN08, 2015) is not complete enough for nutrient assessment and only provide water quality legislation for industrial and domestic usage, but this standard does not include the general health of the ecosystem.

Nitrogen. In the whole Saigon and Dongnai basin, NO_3^- concentration was lower than 2.5 mgN L^{-1} , which is equivalent to good ($0.5\text{-}2.5 \text{ mg. L}^{-1}$) and high status ($< 0.5 \text{ mgN L}^{-1}$) (WFD, 2000). The NH_4^+ concentration was around 0.1 mgN L^{-1} (good status $< 0.4 \text{ mgN L}^{-1}$, WFD 2000) at upstream Dongnai River (DN-up) and downstream Saigon River (NBR). This was not the case for Saigon City Center (SG-City), as it is affected by urban wastewater discharge, NH_4^+ concentration being on average 0.7 mgN L^{-1} and rising up to 3 mgN L^{-1} (moderate, $0.4\text{-}1.6 \text{ mgN L}^{-1}$ to bad status, $1.6\text{-}3.9 \text{ mg L}^{-1}$, WFD 2000). The Saigon River was then highly polluted by ammonium in the dense urban area but less by nitrates, which indicates low emissions from agricultural practices (Grizzetti et al., 2012).

Concentrations of NO_3^- and NH_4^+ in the Saigon River were not only higher than in the very polluted urban Nhue River in Hanoi, Northern Vietnam: $0.0\text{-}0.1 \text{ mgN-NO}_3^- \text{ L}^{-1}$ and $0.02\text{-}0.15 \text{ mgN-NH}_4^+ \text{ L}^{-1}$ (Trinh et al., 2015) but also than in other major tropical rivers, such as the Lower Mekong River (Vietnam): $0.15 \pm 0.05 \text{ mgN-NO}_3^- \text{ L}^{-1}$ and $0.03 \pm 0.03 \text{ mgN-NH}_4^+ \text{ L}^{-1}$ (Li and Bush, 2015), the Senegal River (Senegal): $0.042\text{-}0.17 \text{ mgN-NO}_3^- \text{ L}^{-1}$ and $0.005\text{-}0.03 \text{ mgN-NH}_4^+ \text{ L}^{-1}$ (Mamadou et al., 2016), Chap Phraya River (Thailand): 1.3 mgN L^{-1} and 0.08 mgP L^{-1} (Færge et al., 2001) and the Yangtze River (China): $0.019\text{-}0.027 \text{ mgN-NO}_3^- \text{ L}^{-1}$ and $0.003\text{-}0.005 \text{ mgN-NH}_4^+ \text{ L}^{-1}$ (Liu et al., 2016). Land use in boundary areas of HCMC is dominated by agriculture, while the upstream parts of both the Saigon and Dongnai Rivers are more populated compared to downstream of HCMC; the heart of HCMC being the most densely populated (Figure 1c). NO_3^- originates principally from the leaching of agricultural soils (Billen et al., 2010). However, in rivers influenced by urban wastewater discharges, it can originate from the ammonium nitrification

process (Garnier et al., 2007; Herbert, 1999; Xia et al., 2009). In HCMC, due to the lack of wastewater treatment plants, most of the untreated domestic waste water is discharged directly into the three main urban canals connected to the Saigon River (Figure 1c). Therefore, the untreated domestic waste water becomes the principal source of NH_4^+ .

Indeed, NH_4^+ is a good indicator of domestic pollution among all nitrogen forms. NH_4^+ concentrations increased considerably in Saigon City Center evidencing pollution by effluents from HCMC and its suburbs (Figure 3-7). At the Saigon – Dongnai River confluence, NH_4^+ concentrations returned close to the values observed in upstream Dongnai River showing the spatially limited NH_4^+ pollution of HCMC on its river further downstream. NO_3^- concentrations also decreased at the confluence and we hypothesized that they (NH_4^+ and NO_3^-) played a major role in the stimulation of primary production by phytoplankton in the Saigon River aquatic system (Dugdale and Goering, 1967).

Phosphorus and silica. PO_4^{3-} concentrations ranged from 0.05 up to 0.2 mgP L⁻¹ in Saigon City Center (SG-city) corresponding to good-to-moderate status (WFD, 2000) but Total P concentrations were much critical up to 0.96 mgP L⁻¹ (bad status, WFD, 2000). In the other upstream locations (SG-up) and downstream (NBR) of the Saigon River and upstream of the Dongnai River (DN-up), PO_4^{3-} concentrations were lower than 0.1 mgP L⁻¹ (good status, WFD, 2000) and Total P concentrations were lower than 0.4 mgP L⁻¹ (good to moderate status, WFD, 2000). These results indicated a high level of phosphorus within the city center, especially Total P. The proportion of particulate P in Total P was high (mean of 78 ± 6 % in mean) indicating a strong interaction with suspended sediments. PO_4^{3-} concentrations in the Saigon River were higher than in other rivers, e.g., the Lower Mekong River (Vietnam): 0.02 ± 0.04 mgP L⁻¹ (Li and Bush, 2015) and the Senegal River (Senegal): 0.003-0.013 mgP L⁻¹ (Mamadou et al., 2016). Moreover, we observed that DSi fluctuated between four sampling sites during the 2015-2017 period. The sources of DSi were mainly from diffuse origins (e.g., weathering and erosion of natural rocks) or from point sources (e.g., urban runoff, domestic and industrial waste waters) (Sferratore et al., 2006). Particulate silica can then be dissolved to become a source of DSi, which could explain the fluctuation of DSi observed at the four sampling sites.

Due to the very poor sanitation in HCMC, Total P inputs are mainly from industrial and domestic sources, so that the Total P load in the Saigon – Dongnai River is rather constant over the year.

Less than 10% of domestic effluents and less than 50 % of industrial effluents are treated before being discharged directly into the river (Marcotullio, 2007). The theoretical domestic P concentration can be estimated at 30 mgP L^{-1} from the equivalent per capita ($4 \text{ gP inhab.}^{-1} \text{ day}^{-1}$ and $150 \text{ L inhab.}^{-1} \text{ day}^{-1}$; Lam et al., 2008 and Ministry of Construction, 1998) and total wastewater flux at about $10 \text{ m}^3 \text{ s}^{-1}$ (considering the total unconnected urban population at 6 million inhabitants, i.e., 90% of the 6.7 million inhabitants living in urban districts). Untreated wastewaters would contribute to about 20 % of mean annual discharge of the Saigon River ($50 \text{ m}^3 \text{ s}^{-1}$) and untreated wastewaters P concentrations would be 100 times higher than P concentrations measured in the Saigon River (mean of 0.3 mgP L^{-1} in mean). Domestic inputs are then constant over the time regardless of the season and could maintain P concentrations at a high level within the Saigon River.

During the wet season, the river flow was higher than during the dry season (Figure 3); P input was diluted by rainfall and there was a P depletion with regards to Si. During the dry season, most of the Saigon River water discharge comes from the Dau Tieng reservoir, modifying the nutrients balance. Despite inputs being less diluted during the dry season, the decrease of PO_4^{3-} concentrations, especially effective at this time can be due to algal growth (Causse et al., 2015). In addition, the canal system in HCMC is the main drain for untreated effluents from the urban catchment area so that P can be stored within the sediments. Urban areas include houses, roads, and other public areas, are regularly flooded with a mix of storm water and wastewater leading to the sediments being flushed out by the overflow under rainy conditions (Tran Ngoc et al., 2016). Indeed, high levels of PO_4^{3-} in water and Particulate P in sediment were observed in urban canals in HCMC (Strady et al., 2017) indicating an intense adsorption process of dissolved P onto sediment. This accumulation in the sediment was evidenced in the urban canals of Bangkok (Thailand), which presents similar geography, demography and wastewaters management to HCMC (Færge et al., 2001).

Nutrients proportions. We observed from bi-monthly monitoring data that the dissolved inorganic nitrogen proportion (%DIN) at four sampling sites fluctuated slightly from $38 \pm 24 \%$ at NBR to $49 \pm 22 \%$ at SG-up during the 2015-2017 period. Whilst PO_4^{3-} proportion was lowest at SG-city ($32 \pm 27 \%$), Dongnai River experienced the highest proportion ($49 \pm 31 \%$ at DN-up). The change in proportion of primary nutrients (N, P and Si) can lead to a change in the Redfield ratio N:P:Si,

affecting the risk of eutrophication in aquatic systems. In turn, phytoplankton uptake affects nutrient concentrations (Redfield, 1958 and many others since then); furthermore, the increased anthropogenic nutrient inputs necessarily change the proportion of nutrients for phytoplankton growth (Turner and Rabalais, 1991). It means that the feedback effect between nutrients and phytoplankton uptake, can control phytoplankton dynamics (e.g., dominant species, successions from diatoms to non-diatoms).

The seasonal variability of water quality in the Saigon River was explored from the bi-monthly monitoring. Seasonality is of most important in terms of the eutrophication problem because phytoplankton development is naturally controlled by factors that follow seasonal cycles (Le et al., 2015). Surprisingly, there was no clear seasonality of all observed parameters. The relation between water discharge and nutrient variation is not relevant in this system as the discharge is controlled by the tide and water release from Dau Tieng reservoir (Trieu et al., 2014). The parameters monitored and presented in Figures 3-4 to 3-6 showed that the salinity was by far the one that exhibits the strongest seasonality. We can actually clearly distinguish the two peaks during the 2016 and 2017 dry seasons, but we can also clearly see that their magnitudes are very different. Based on statistical analyses, we observed that some nutrients fluctuated seasonally (p-value < 0.05), e.g., NO_3^- (p-value = 0.015) and NH_4^+ (p-value = 0.04) in SG-up; Total P (p-value = 0.026) in DN-up or NO_3^- (p-value = 0.049) in SG-city. Other nutrients such PO_4^{3-} did not vary seasonally with p-value > 0.05, especially in SG-city. Therefore there is no evidence of hydrological seasonality to explain the variability of most of the nutrient concentrations.

Concerning the spatial fluctuation of nutrients in the Saigon River, our results clearly showed the impact of HCMC on nutrient concentrations, which is presented by an increase of Total N, NH_4^+ , Total P and PO_4^{3-} (Figure 3-7).

4.2. Eutrophication status and limiting factors

According to the eutrophication classes proposed by Dodds et al. (1998) and Dodds and Welch (2000), we classified the water quality of the Saigon and Dongnai River branches upstream and downstream of the City Center as mesotrophic, and being clearly eutrophic in the SG-city site (Table 3-1). To identify the principal limiting factor of algal blooms in the system, we used the N:P molar ratio presented in Figure 3-8. A molar ratio of N:P lower than 16 indicates that P is the

limiting factor for algal growth, while a ratio higher than 16 shows that N is the key element. When observation that the molar N:P ratio is higher than 16:1 in the Saigon – Dongnai River system, P is indicated as the limiting factor. More precisely, the molar N:P ratio in the Saigon River (N:P = 69.7 ± 44.8 on average) was higher than that of the Dongnai River (N:P = 45.5 ± 22.9 on average). In May 2017, the N:P ratio peaked at 258, showing that N was largely in excess in comparison to P. It would seem that in order to reduce eutrophication in the Saigon River, the control of the key element P is necessary.

Table 3-1. Classification of trophic levels of rivers and assessment of levels for the four sampling sites in the Saigon – Dongnai Rivers.

Site	DO (mg L ⁻¹)	Total N (mg L ⁻¹)	Total P (mg L ⁻¹)	Chl-a (µg L ⁻¹)	Trophic status classification
According to Dodds et al., 1998 and Dodds and Welch, 2000					
	>5	<0.7	<0.025	<10	Oligotrophic
	1-5	0.7-1.5	0.025-0.075	10-30	Mesotrophic
	0-1	>1.5	>0.075	>30	Eutrophic
Classification of trophic levels of four sampling sites in Saigon – Dongnai Rivers					
SG-up	2.5	1.8	0.15	3.2	Mesotrophic
SG-city	1.7	3.4	0.30	29.6	Eutrophic
NBR	3.6	2.2	0.12	1.6	Mesotrophic
DN-up	6.1	1.5	0.09	4.7	Oligotrophic

The relationship between POC and Chl-*a* was analyzed at the four sampling sites to verify the origin of organic carbon. This relationship observed with $R^2 < 20\%$ indicates that the main composition of organic carbon in the Saigon River is not phytoplankton. Furthermore, POC/Chl-*a* weight-to-weight ratio values were calculated for each site and were all higher than 200 (e.g., 1573 ± 1600 , 517 ± 770 , 1564 ± 1620 and 524 ± 620 at SG-up, SG-city, NBR and DN-up,

respectively), characterizing a detrital or degraded material (Cifuentes et al., 1988). This means that the origin of the organic matter within the river was principally allochthonous and likely from untreated wastewaters. Moreover, the average C:N ratios calculated at the four sampling sites were all lower than 12 (3.0 ± 2.5 , 4.5 ± 4.7 , 2.6 ± 3.2 and 2.8 ± 4.5 at SG-up, SG-city, NBR and DN-up, respectively). These findings supported the assumption of allochthonous organic matter inputs from wastewaters (Hedges et al., 1986; Hedges and Oades, 1997; Balakrishna and Probst, 2005; Zafirah et al., 2017). Nevertheless, values of POC/Chl-a ratio decreased in SG-city where algal development is maximum. It indicated that in this part of the river, the proportion of autochthonous organic matter is increasing due to eutrophication.

4.3. Impact of HCMC on the water quality of the Saigon River

Since the early 1990's, HCMC has been experiencing an economic boom (Du and Fukushima, 2010; Vo, 2007). This growth has been accompanied with a rapid urbanization, the urban surface area has indeed quintupled between 1990 and 2012; with about 660 km² of surface becoming urbanized during that period (Kongtis et al., 2014). Since the study of Gurby et al. (2002), urbanization has continued to expand along the Saigon River (North and South), with a population density that can exceed 30,000 inhabitants per km² (see Figure 3-1c). This economic and demographic evolution has created new challenges for the city in terms of water quality management (Vo, 2007). Policy makers are now being forced to find new solutions to reduce pollution and restore the aquatic ecosystem of the Saigon River.

The monitoring conducted at our four stations for more than two hydrological years underlines a critical eutrophication observed each year at SG-city. The three episodes of eutrophication observed during our survey lasted several weeks, but were not easily correlated with dry or wet hydrological seasons; even if the highest peak of eutrophication was observed in the middle of the dry season in January 2017 ($\text{Chl-}a > 150 \mu\text{g L}^{-1}$, Figure 3-6c).

Figure 3-9 proposes a synthetic view of population distribution along the Saigon River and the level of the trophic status in order to highlight the connections between urbanization and eutrophication. The snapshot presented in Figure 3-9a shows a rapid deterioration of water quality, from oligotrophic waters in the upper 5-10 km of monitoring (pK-55 to pK -48) to mesotrophic (pK-48 to pK-40) and eutrophic waters downstream (pK -40 to pK -12). The water quality is then

restored quite rapidly and recovers an oligotrophic status around pK-10, i.e., ten kilometers before the confluence of Saigon and Dongnai Rivers.

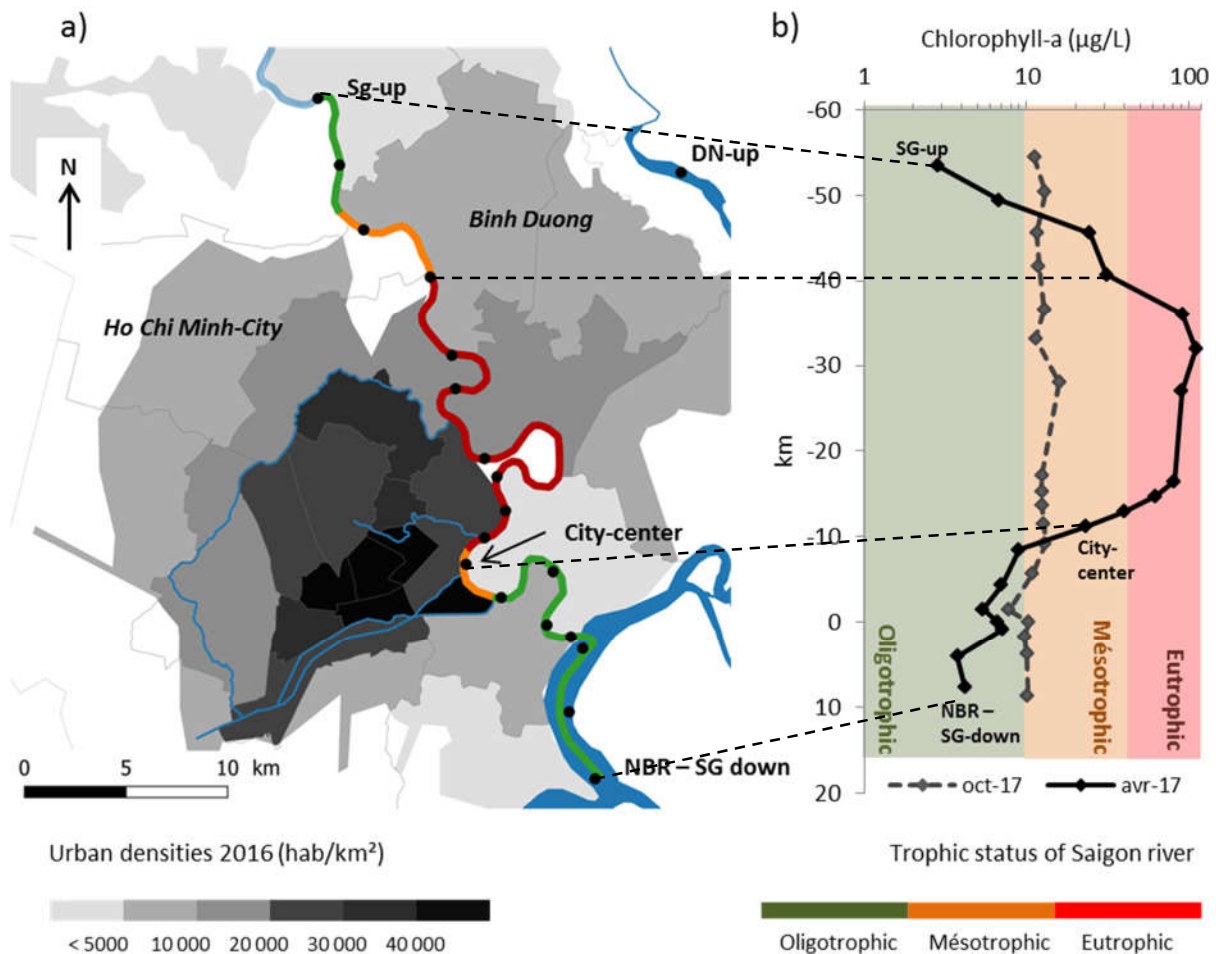


Figure 3-9. Seasonal variations of the trophic status of the Saigon River with the population density. (a) Snapshot of the eutrophic level during the dry season (19th of April 2017); spatial distribution of population density (see also Figure 3-1c). The patch of Chl-a presented in Figure 3-9a is corrected from the upstream-downstream oscillation of water masses with tides. Depending on tidal level, water masses can oscillate with an amplitude of 6 km around their mean locations during spring tides and (b) Intercomparison of snapshot trophic situations for dry (black solid line) and wet (grey dashed line) seasons.

For the upper stretch of the river (from pK-55 to pK-10), Figure 3-9 shows a good match between the increase in eutrophication and the density of inhabitants, which demonstrates the direct effect of untreated urban wastewater release on river quality. The restoration of water quality beyond

pK-10 can appear surprising, but it is supported by two assumptions: first, the restoration of water quality can be attributed (at least partially) to the positive effect of the Binh Hung wastewater treatment plant, which collects and treats a part of the wastewaters from the south of the City Center (district 5, district 7); second, water quality improvement is more likely due to the dilution of polluted water from the Saigon River as it mixes with seawater and the Dongnai River waters. The water quality monitored at station NBR indeed presented a good water quality status ($\text{Chl-}a < 10 \mu\text{g L}^{-1}$) throughout the 30 months of monitoring. During the dry season, this water body mixes daily with the Saigon River and progressively diffuses upstream. This is demonstrated by the saline intrusion observed at SG-city station in Figure 3-4b.

Furthermore, during the dry season survey (19th April 2017), the DO level was high due to photosynthesis processes by a high level of phytoplankton, reaching its highest value at $110 \mu\text{g Chl-}a \text{ L}^{-1}$. During the wet season, the upstream and middle sections of the Saigon River were deoxygenated ($\text{DO} < 2 \text{ mg L}^{-1}$, see Figure 3-7) because of oxygen consumption by biological activities (not only degradation of organic matter but also nitrification). At these locations, the water body contains large amounts of organic matter which is quantified by high DOC and POC concentrations (making up above 3 mgC L^{-1}) and NH_4^+ concentrations higher than 0.2 mgN L^{-1} (cf. Figure 3-7).

In addition to the high level of NH_4^+ in the middle section, NO_3^- was also low in this area (Figure 3-7). There was a slight decrease in NH_4^+ concentrations during the dry season from upstream to downstream of HCMC, while there was a drastic increase in NH_4^+ levels at the City Center during the wet season due to the discharge of untreated domestic waste water from urban zones. By contrast, NO_3^- concentration increased at the City Center during the wet season, whereas their values decreased during the dry season (Figure 3-7). During the dry season, NH_4^+ slightly decreased at the same time as NO_3^- indicating a dilution by the Dongnai River or seawaters. On the contrary, during the wet season, there was a strong nitrification process with a rapid decrease of NH_4^+ at City Center occurring at the same time as an increase in NO_3^- (Figure 3-7). The nitrification process was likely initiated with the input of oxic waters up to pK -10 (Figure 3-7). The result highlights the potential capacity of the estuary to metabolize the pollution emitted by HCMC (Flemer, 2006).

In general, anthropogenic activities (e.g., agricultural and industrial activities, poor domestic wastewater treatment) have led to eutrophication of water bodies with an expected impact on water quality and drinking water production. To deal with these problems, wastewater planning and management is needed. It is critical that wastewater management becomes a fundamental part of urban planning and watershed management. HCMC authorities have already planned to build ten new WWTPs able to treat most of the domestic waters in the next ten years (Tran Ngoc et al., 2016). HCMC must also urgently incorporate principles of ecosystem management from the watershed into the coastal ocean. The natural ecosystem can be used to provide water more sustainably; with this in mind, wetland management is an essential option to sustain the water cycle and hence, to decrease nutrient and organic pollution (Gupta et al., 2016).

5. Conclusions

Our two-and-a-half year survey of water quality was the first time to evaluate global nutrient levels in the Saigon – Dongnai Rivers. As outlined above, the values observed in the upstream of HCMC were close to those observed in the downstream of HCMC, which showed the low anthropogenic impact in the more rural areas. We did not take into account nutrient input from diffuse sources (e.g., agricultural activities) due to lack of data. However, we assumed that nutrient input from diffuse sources is of second order in comparison to high nutrient input from point sources in the urban areas of HCMC. Indeed water in the heart of HCMC was polluted and exhibited high nutrient concentrations (especially phosphorus) due to untreated effluents from residential and industrial areas. The Saigon River experienced extreme eutrophication episodes. We can therefore conclude that there are numerous problems and challenges with regards to water quality management in HCMC. The question we need to ask is how to revise, improve and implement the current legal system regarding the water sector in general, and wastewater management in particular. Appropriate quantification of the eutrophication status is a first step in which scientists and policy makers need to work together using operational modelling tools to move from observation to mitigation. We need to understand how water quality in downstream of HCMC is recovered. This should be further investigated when looking at dilution between the Saigon River, the Dongnai River and seawater within the whole salinity gradient down to the coastal area. A modelling approach of the hydrodynamics and biogeochemistry of the estuary is currently conducted to identify the predominance of each of the internal biogeochemical processes.

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Conclusion of Chapter 3

As outlined above, values observed upstream of HCMC were close to those observed downstream of HCMC, which indicated low anthropogenic impact in these areas. Indeed, water in the heart of HCMC was polluted with high level of nutrients (especially phosphorus) due to untreated wastewater effluents from urban and industrial areas. We need to understand how recovery of good quality water downstream of HCMC. This should be further investigated when looking on a biogeochemical processes and a dilution process between the Saigon River, the Dongnai River and seawater within the whole salinity gradient down to the coastal area. In order to understand these processes, a laboratory experiment set-up of P adsorption-desorption capacity onto SS within the observed salinity gradient was suggested and presented in Chapter 4.

CHAPTER 4. PHOSPHORUS ADSORPTION-DESORPTION PROCESSES IN THE TROPICAL SAIGON RIVER ESTUARY (SOUTHERN VIETNAM) IMPACTED BY A MEGACITY

This section is copied from the paper that will be submitted to Estuarine, Coastal and Shelf Science. Results of the longitudinal and salinity profiles field survey were first used to assess the level of phosphorus in suspended sediment (SS) and deposited sediment (DS) and to explain the spatio-temporal variability of phosphorus in Saigon River salinity gradient, a sector strongly impacted by Ho Chi Minh City (HCMC). This section also presents an original laboratory investigation of P adsorption-desorption capacity onto SS within the observed salinity gradient. The experiments were conducted in a Jar test under controlled levels of SS, salinity and turbulence. Particle Size Distribution (PSD) and flocculation index were examined as possible key physical indicators of changes in phosphorus adsorption processes.

PHOSPHORUS ADSORPTION-DESORPTION PROCESSES IN THE TROPICAL SAIGON RIVER ESTUARY (SOUTHERN VIETNAM) IMPACTED BY A MEGACITY

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Abstract

The Saigon River flows through one of the most growing megacity of South East Asia, Ho Chi Minh City (HCMC, > 8.4 million inhabitants). This tidal river is characterized by a tropical monsoon climate, alternating a wet and a dry season. In the last decades, increased economic and urban developments of HCMC have led to harmful impacts on the water quality of this tidal river, with severe events of eutrophication. This situation results from the conjunction of contrasted hydrological seasons and the lack of upgraded sanitation infrastructures: indeed, less than 10% of the domestic wastewater are collected and treated before being discharged directly into urban canals or rivers. Phosphorus (P) being the nutrient considered to be the key factor controlling freshwater eutrophication, this study focuses on P dynamics. Based on field measurements and original laboratory experimentations, we assessed the phosphorus levels in the river water and sediments, and investigated P adsorption-desorption capacity onto suspended sediment (SS) within the observed salinity gradient. Field surveys showed a clear impact of the HCMC megacity on the total P content in SS that increased three fold in HCMC center, as compared with the upstream

values (0.3-0.8 gP kg⁻¹). Downstream, in the mixed estuarine area, the total P was lower than 0.5 gP kg⁻¹. Laboratory experiments were carried out to characterize the influence of SS concentrations (SS = [0.5-0.9] g L⁻¹), salinity (S = [2.6-9.3]) and turbulence (G = [22-44] s⁻¹) on the sorption capacity of P onto sediment. The size of sediment particles and their propensity to flocculate were also originally measured with a recent developed instrument: the System for the Characterization of Aggregates and Floccs (SCAF®). Under the investigated experimental conditions, SS concentrations affected the most the adsorption of P onto sediment, e.g., P adsorption capacity increased when SS concentrations rose. In contrast, salinity and turbulence modify in a lower importance the adsorption properties of sediments. Among these observed variables, SS concentration was shown to be the main driver for adsorption capacity of P onto SS within salinity gradient. We then discuss in this paper the implication of these findings for the understanding of P dynamics within a high-urbanized, tropical estuary.

Key words. Sediment, tropical region, phosphorus dynamics, eutrophication, flocculation.

1. Introduction

Among all nutrients, phosphorus (P) is generally recognized as playing a crucial role for plant growth in soils and aquatic environments. Since the 1960s, the utilization of mineral fertilizers and polyphosphates in washing powders has increased the P inputs to most of aquatic environments of northern countries with serious ecological consequences. P was banished in washing powders, in the mid-1990s, and its application to soils was also reduced (Némery and Garnier, 2016; Le Noë et al., 2018). In the same period, many scientists continued to pay attention to P concentrations in water (since Vollenweider, 1968), but started to analyze its reactivity with sediment (Heathwaite and Johnes, 1998; Sims et al., 1998; Sui et al., 1999; Richard et al., 2001), and showed that high P values could strongly promote algal growth and increase the risk of eutrophication. It is highlighted that the mobility of P and its sorption capacity can shift from dissolved to particulate phases through biological and physico-chemical processes, with some consequences in P transport and P dynamics. P can be transported from upstream basins to estuaries under form of particulate phosphorus (PP) (Conley et al., 1995; Paludan and Morris, 1999; Martha et al., 2004; Némery and Garnier 2007), but P can also be released from particulate to dissolved form, which leads to the increase of the dissolved P concentration in aquatic system. Dissolved P is then easily available for phytoplankton uptake (Meybeck, 1982; Reddy et al., 1999; Jordan et al., 2008).

Phosphorus can be transformed by adsorption-desorption and by precipitation-dissolution processes (Froelich, 1988). These processes can be controlled by iron (Fe), aluminum (Al) oxides and their organic complexes (Torrent, 1997; Borggaard et al., 2005; Bruland and DeMent, 2009; Hinkle et al., 2015), silicate clays and calcite (Richardson, 1985; House and Donaldson, 1986). The change of the ionic strength within salinity gradient (i.e., between brackish and saline waters) is considered as an important factor for P adsorption-desorption capacity (Sundareshwar and James, 1999; Némery and Garnier, 2007; Jordan et al., 2008; Bruland and DeMent, 2009). The sediment redox potentials, especially the low ones, can also have an impact on the dissolution, and can release P in water (Lai and Lam, 2008).

Because phosphorus can adsorb onto particles, hydrodynamic (river discharge, tides and induced turbulence, salinity) and sediment dynamics (erosion, transportation, flocculation, deposition) conditions need to be characterized prior to any robust assessment of eutrophication risk. While

sediment's transportation depends on water discharge (Craft and Richardson, 1993), dissolved P release is governed by diffusion of P from pore water of underlying sediment (Reddy et al., 1995; Richardson, 1985). Likewise, physical characteristics of sediment like particle size, porosity and flocculation can become the dominant factor for P adsorption capacity. When particle size distribution of SS decreases, P adsorption capacity onto sediments usually increases (Walter and Morse, 1984; Zhang and Huang, 2007) because the proportion of fine clay particles increase. A sediment containing significant proportion of clay minerals, with electromagnetic properties causing the sediment to bind together, is called a cohesive sediment. Small particles can be however easily re-suspended by turbulence (through wave action or tide in estuaries), and have longer residence time in the water column than coarser particles, such as silt and sand that are mostly transported by siltation near bottom. Therefore, cohesive sediments interact more easily with dissolved phosphorus in the water column. Besides, P adsorption capacity can be strongly modified at the interface of bottom sediment and water (Kim et al., 2003; Wang et al., 2006).

In developing countries, such as Vietnam, fast economic growth has affected the quality of aquatic systems (Trinh et al., 2012). Aquatic systems are thus frequently enriched in P by both the diffuse (e.g., agricultural activities) and the point inputs (e.g., domestic and industrial wastewaters discharge) (Camargo et al., 2005). This is particularly the case of Ho Chi Minh City (HCMC), the second biggest city of Vietnam. Industrialization and urbanization have developed rapidly during the last decade while the wastewater treatment was remained basically unchanged (Marcotullio, 2007; Nguyen-Thanh et al., 2017; Strady et al., 2017).

The main objective of this study is to assess the dynamics of phosphorus in particulate and dissolved phases along the tidal Saigon River during the dry and wet seasons for a better understanding of the cause of eutrophication. For this purpose we investigated P adsorption-desorption capacity onto SS at different hydrodynamic conditions (i.e., level of SS concentration, salinity, turbulence) to assess the link between adsorption-desorption capacity and physical properties of sediments (flocculation, particle size).

2. Material and methods

2.1. Study area

The Saigon River, a part of the Saigon – Dongnai River basin located in Southern Vietnam, is about 250 km long with a catchment area of 4717 km² up to the confluence with the Dongnai River (Figure 4-1a). Upstream of the Saigon River, the Dau Tieng reservoir (270 km² and 1580×10⁶ m³) was constructed in 1985 for irrigation, flood protection purposes and the control of the intrusion of saline water (Trieu et al., 2014). When flowing through HCMC, the Saigon River is connected with urban canals and further joins the Dongnai River to become the Soai Rap River flowing through Can Gio mangrove to the South China Sea. The flow direction of the Saigon River is predominantly driven by the asymmetric semi-diurnal tides. The region falls in a tropical monsoon climate. The year is divided into two distinct seasons: the wet season from May to November and the dry season from December to April. During the dry season, the net residual discharge is low (few tens of m³ s⁻¹) and is mainly controlled by the amount of water released from the Dau Tieng reservoir to flush out salt intrusion. The wet season shows a slight increase of the water discharge from June to October, followed by a recession at the end of November (Nguyen et al., 2019). The mean inter-annual discharge calculated over the 2012-2016 period for the Saigon River (50 ± 21 m³ s⁻¹) is around twelve times lower than the one's for the Dongnai River (613 ± 218 m³ s⁻¹) (Nguyen et al., 2019).

HCMC is established in a low elevation coastal zone (LECZ), along the banks of the Saigon River and belongs to a transitional area between the southeastern Vietnam and the Mekong Delta. Economically, HCMC has grown fast over the last ten years, so that now it is considered as one of the five most dynamic cities in the world (World Economic Forum News, January 2017). The population of HCMC was 8.4 million inhabitants in 2016. There is only less than 10 % of domestic wastewater collected and treated before being released directly to the three main urban canals (Figure 4-1b) or to the Saigon River (Marcotullio, 2007). Due to untreated effluents from residential and industrial areas, the Saigon River water episodically suffered severe phases of eutrophication (e.g., high levels of nutrients - N and P - and algal biomass - expressed as chlorophyll *a* concentrations (Nguyen et al., in press). The main wastewater treatment plant (Binh Hung WWTP) is located in district 8, south of the city center (Figure 4-1b). Its treatment capacity

is $141\,000\text{ m}^3\text{ day}^{-1}$ (426 000 inhabitants). In 2017, one extension was under construction to reach a total capacity of $469\,000\text{ m}^3\text{ day}^{-1}$ (1 390 000 inhabitants) (source: ATLAS Ho Chi Minh City) and ten new WWTPs are planned to be constructed in the next decade (Tran Ngoc et al., 2016).

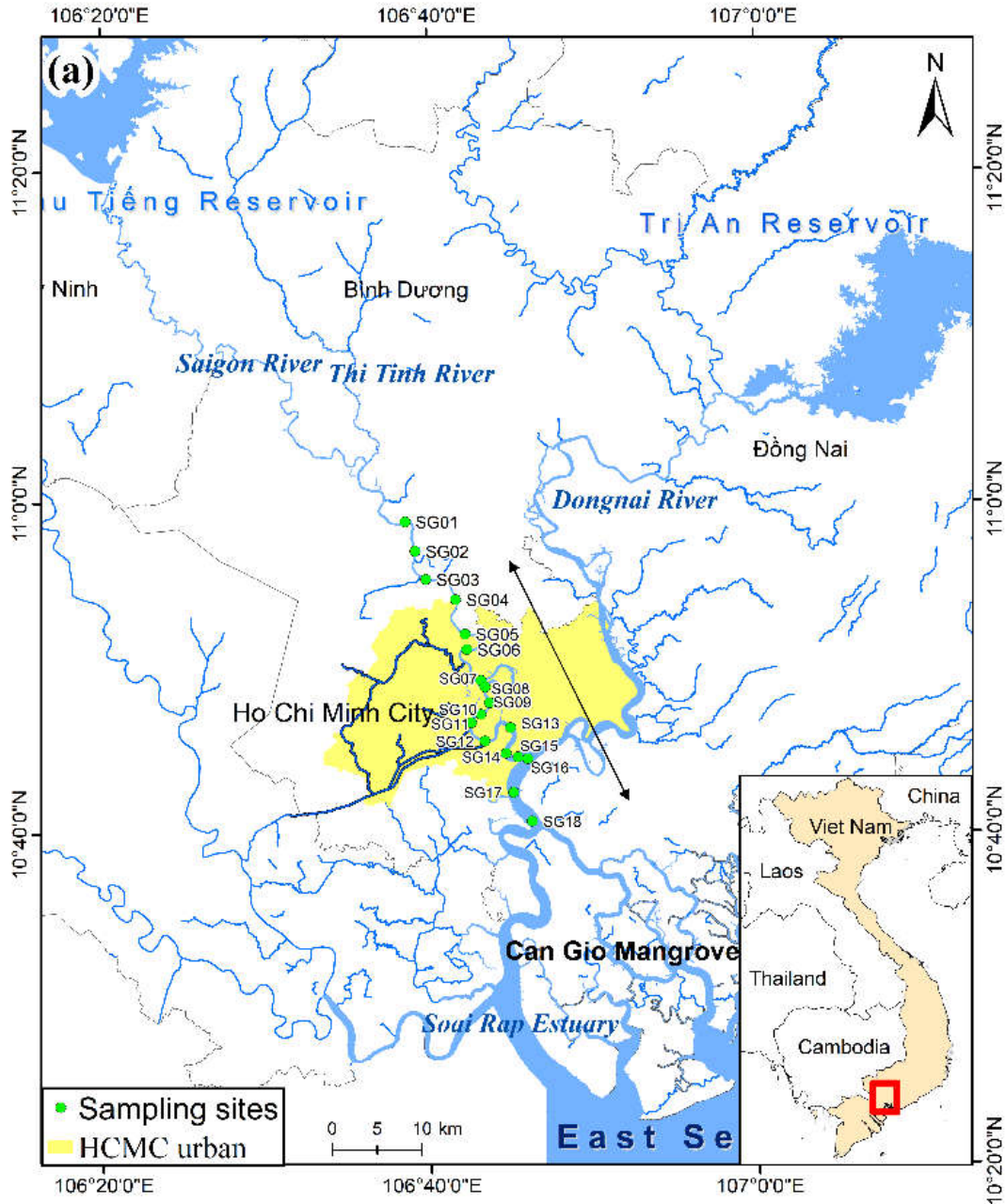


Figure 4-1. (a) Saigon – Dongnai basin catchment and sampling sites along a longitudinal profile, (b) The population density distribution on the zoomed urban central districts (source: HCMC Statistical Year Book 2016). Grey arrow indicates the amplitude of upstream salt intrusion (see Table 1 for details of sampling sites).

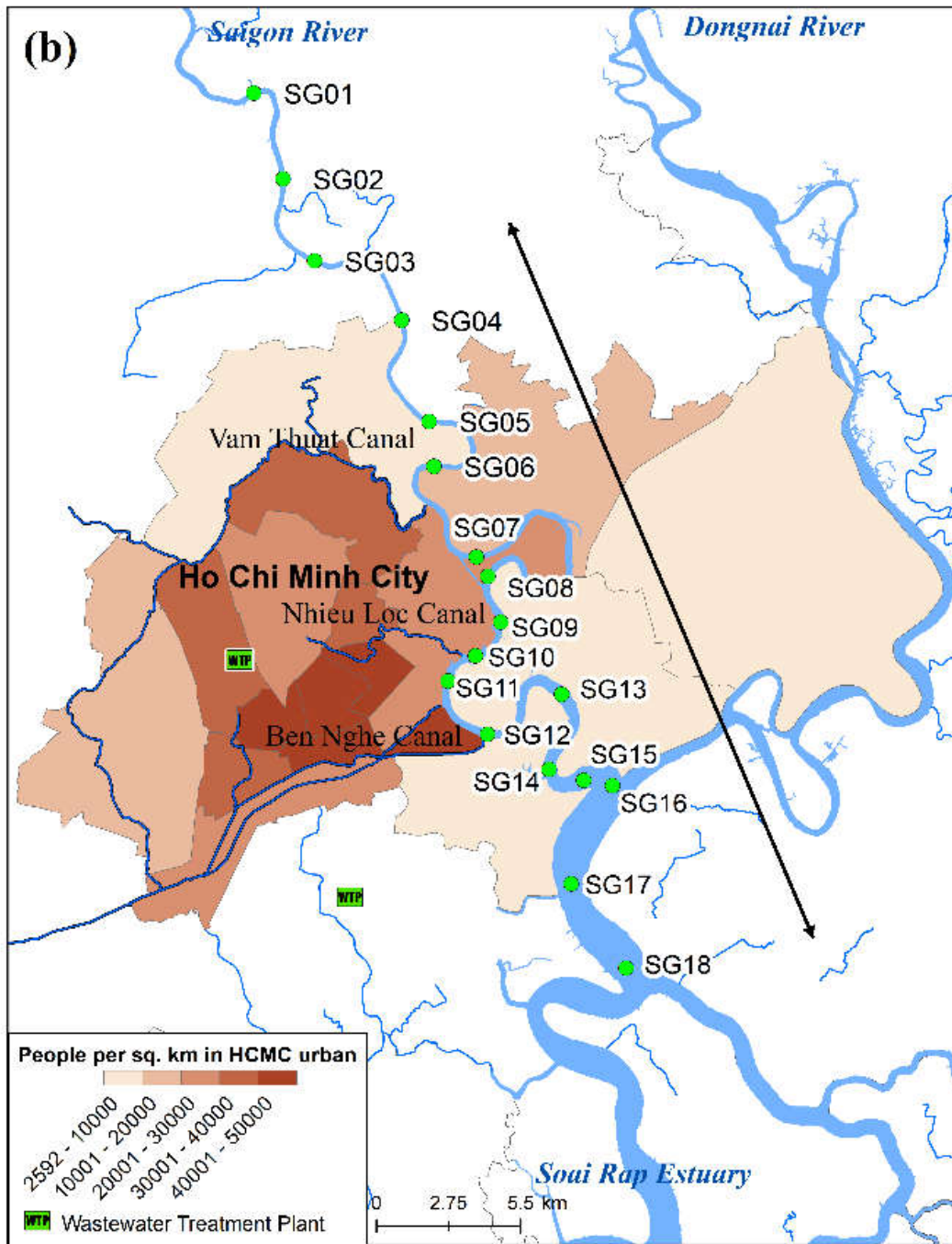


Figure 4-1. (a) Saigon – Dongnai basin catchment and sampling sites along a longitudinal profile, (b) The population density distribution on the zoomed urban central districts (source: HCMC Statistical Year Book 2016). Grey arrow indicates the amplitude of upstream salt intrusion (see Table 1 for details of sampling sites).

Table 4-1. Sampling sites along the longitudinal profile on the Saigon River (pK = 0 corresponds to the confluence between Saigon River and Dongnai River)

Name	Coordinates		pK (km)	Zones
SG01	10°58'50"N	106°38'33"E	-45	Upstream <i>Before dense urban area</i> <i>Possible salt intrusion during dry season</i>
SG02	10°57'04"N	106°39'08"E	-41	
SG03	10°55'21"N	106°39'47"E	-37	
SG04	10°54'06"N	106°41'35"E	-32	
SG05	10°52'00"N	106°42'10"E	-28	
SG06	10°51'07"N	106°42'16"E	-23	
SG07	10°49'16"N	106°43'03"E	-18	Ho Chi Minh City Center <i>Wastewaters release</i> <i>Salt intrusion</i>
SG08	10°48'47"N	106°43'23"E	-16	
SG09	10°47'53"N	106°43'37"E	-15	
SG10	10°47'10"N	106°43'03"E	-13	
SG11	10°46'28"N	106°42'33"E	-11	
SG12	10°45'35"N	106°43'30"E	-8	
SG13	10°46'16"N	106°44'54"E	-4	Downstream <i>Confluence with Dong Nai</i> <i>Salinity gradient</i> <i>Estuarine zone</i>
SG14	10°44'51"N	106°44'44"E	-1	
SG15	10°44'32"N	106°45'23"E	0	
SG16	10°44'13"N	106°45'42"E	1	
SG17	10°42'32"N	106°45'20"E	4	

SG18	10°40'46"N	106°46'02"E	8	
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2.2. Sampling campaigns

Six surveys (two longitudinal profiles and four additional salinity gradient campaigns) were carried out during the dry season (April 2017) and the wet season (October 2017) along the Saigon River (Figure 4-1).

Longitudinal profiles. Two campaigns from upstream to downstream of HCMC were conducted to understand the spatial fluctuation of water quality along the Saigon River. Total length of the profile was 50 km including 18 sampling points (from SG01 to SG18, Figure 4-1b). These longitudinal profiles were conducted on-boat on 19th April 2017 from 8:00 am to 4:30 pm (dry season) and on 20th October 2017 from 9:30 am to 6:00 pm (wet season), starting at Saigon upstream (SG01) and ending at Saigon downstream (SG18) (see Figure 4-1b).

Salinity gradient campaigns. Four additional on-boat surveys were conducted to find out the variability of nutrients within salinity gradient. These 30km long profiles (12 sampling points, from SG07 to SG18, see Figure 4-1b) were carried out on-boat on 21st and 25th April 2017 (two campaigns during the dry season) and on 23rd and 25th October 2017 (two campaigns during the wet season).

A GPS was used to geolocalize each sampling point and for calculating the corresponding kilometric point (pK), in which pK = 0 (i.e., SG15, Figure 4-1b, Table 4-1) is the confluence between the Saigon and Dongnai Rivers, negative values upward and positive downward.

2.3. Measurement and analytical methods

2.3.1. In-situ sampling and measurements

Physico-chemical parameters (e.g., depth, temperature, pH, conductivity, salinity, dissolved oxygen (DO), chlorophyll *a* and turbidity) were measured *in situ* along the vertical profile by using Hydrolab[®] DS5 Multiparameter Probe. Each water sample was collected 0-30cm below the surface in a 5-L polypropylene recipient for further analysis of Total Suspended Sediment (TSS) and Total P and dissolved P.

Deposited sediment samples were taken by using sediment grab at the river bank during the two longitudinal profile campaigns to analysis P content (19th April and 20th October 2017). During the 1st campaign of salinity profile of each season (21st April and 23rd October 2017), 30 liters of surface water were taken to analyze particulate phosphorus (PP) in the SS at 12 sites (from SG07 to SG18). During the 2nd campaign of salinity profile of each season, 30 liters of water were taken at two depths, 0-30 cm below water surface and near water-sediment interface to analyze PP at six sites (from SG13 to SG18).

2.3.2. Sediment preparation

Suspended sediments samples collected in the 30-liter bottles where let to deposit during 24 hours. After siltation, the overlying water was siphoned off. Wet concentrated SS and other deposited sediments were then freeze dried, grinded and sieved through 200 μ m hole size before being stored until their use for experiments and particulate phosphorus analyses.

2.3.3. TSS and phosphorus measurement

In the laboratory, 5L of samples were filtered through GF/F Whatman filter (porosity 0.7 μ m) to analyze PO₄³⁻ using acid ascorbic method (APHA, 1995). Total P was measured on unfiltered samples using persulfate digestion process and standard colorimetric method (American Public Health Association: APHA, 1995). Reproducibility for replicate measurements was better than 5% for all total and dissolved P.

TSS (in mg L⁻¹) were analyzed on a pre-weighed standard glass-fiber filter (GF/F) through which a well-mixed sample were filtered. The residue retained on the filter was dried during about 24 hours at 50°C.

Total particulate phosphorus (TPP) concentration in sediments was measured using a high temperature/HCl extraction technique (Némery and Garnier, 2007) prior to phosphate measurement by colorimetric method (Murphy and Riley, 1962). To determine particulate inorganic phosphorus (PIP), the measurement was similar to the procedure for the analysis of TPP, except the step of high temperature mineralization. Particulate organic phosphorus (POP) was determined by subtracting PIP concentration from TPP concentration ([POP] = [TPP] – [PIP], in gP kg⁻¹).

2.4. Experimental designs

Experiments were conducted in a 2-liter Jar tank to simulate sorption processes of phosphate (PO_4^{3-}) for contrasted conditions of turbulence, salinity and suspended sediment concentrations. The design was close to the one of Keyvani and Strom (2014). Experimental conditions were chosen to be as close as possible as natural conditions. Range of TSS was 0.25 to 0.90 g L^{-1} , range of salinity was 2.6 to 9.3 and range of turbulence was 22 to 44 s^{-1} . To do the experiment, a representative sediment sample was selected on the basis of the analysis of the main patterns of P distribution along the longitudinal profiles. In total, eight hydrodynamic conditions were investigated and each time, ten levels of PO_4^{3-} were considered (Table 4-2). Each duplicate Jar test was stirred during four hours in the lab under 26°C air conditioning stable temperature (Figure 4-2). First, preliminary tests showed that four hours of stirring was long enough to reach steady state equilibrium between particulate and dissolved P. The full set of experiments represented 160 individual Jar tests (80 duplicates). To realize these experiments, hundreds of liters of the Saigon River water were sampled during field surveys and the water was let to deposit during 24 hours to remove suspended sediment and to be used as a medium. The field sample was then filtered through GF/F filter (porosity 0.7 μm) to remove remaining suspended particles and stored in cool room until the use for experiments.

For these experiments, we hypothesized that inorganic phosphorus is mainly present as PO_4^{3-} form, and we considered dissolved organic phosphorus (DOP) as a negligible part of dissolved phosphorus (Nguyen et al., 2019). Phosphate solutions (KH_2PO_4) were prepared for ten different concentrations with demineralized water (from 0 to 3.0 mgP L^{-1} , see Table 4-2) before they were successively added to the Jar test. A gradually increased mass of sediment, obtained from oven-dried, was added to these Jar tests to reach suspended sediment concentrations of $\text{SS} = 0.25, 0.50$ and 0.90 g L^{-1} . One drop of pure chloroform was added to all the samples to stop any possible biological activity during the experimentation (Aissa-Grouz et al., 2016). After four hours of stirring in the Jar tank, the solutions were filtered through a 0.45 μm porosity Millex syringe filter to determine the PO_4^{3-} concentration. Such a design specifically help evaluating whether the kinetics of adsorption-desorption of phosphorus (i.e. variation of the maximal adsorption capacity of P onto SS - P_{ac} and the half-saturation concentration - K_{ps} , see eq. 2 here after) was modified,

or not, by hydrodynamics conditions and to what extent the degree of flocculation of particles may control this kinetic.

Table 4-2. Phosphorus adsorption-desorption experimental conditions

Conditions	SS (g L ⁻¹)	Turbulence (s ⁻¹)	Salinity	PO ₄ ³⁻ added (mgP L ⁻¹) as KH ₂ PO ₄
C1	0.5	44	2.7	0; 0.05; 0.1; 0.2; 0.5; 0.75; 1.0; 1.5; 2.0 and 3.0
C2	0.5	22	2.7	
C3	0.25	44	2.6	
C4	0.25	22	2.6	
C5	0.5	44	9.3	
C6	0.5	22	9.3	
C7	0.9	44	2.7	
C8	0.9	22	2.7	

Another kind of experiments was set up to assess some specific and original physical analyses of sediment properties. The evaluation of the degree of flocculation of particles was done by two ways: a destructive way, by sonication, and a nondestructive way, by sedimentation. The “sonication technique” is based on the analysis of Particles Size Distribution (PSD). A first PSD is realized after sampling in the Jar tank, and a second one is realized on the same sample, but after two minutes of sonication to break flocs. The inter-comparison of the two PSDs provides information on the content of non-cohesive particles (no change in peaks and magnitude of PSDs before and after sonication) and the degree of flocculation of the cohesive particles (shift of the peak to lower particle size bands and decrease of its magnitude in PSD after sonication). According to Lee et al. (2012), we may anticipate the existence of different populations of particles. Flocculi (a single class of particles that corresponds to very fine silt: 8-50 μm), flocs (50-200 μm) and macro-flocs (> 200 μm) are cohesive particles that will be, at least partially, destroyed in primary

particles (0-8 μm) after sonication; silt (2-63 μm) and sand (63 μm to 2 mm) are non-cohesive particles that are not affected by sonication and have virtually no effect on biogeochemistry.

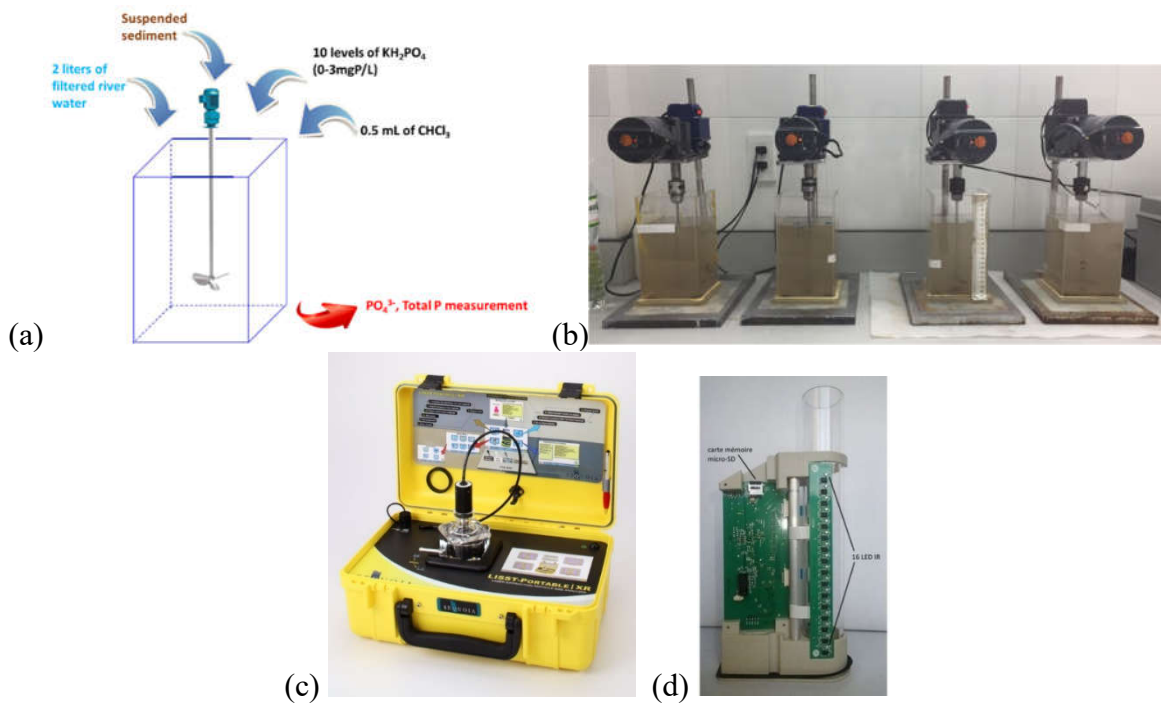


Figure 4-2. (a and b) Experimental designs, (c) The portable granulometer of LISST XR system and (d) System for characterization of aggregates and flocs (SCAF).

To realize this destructive test, we used a LISST Portable XR instrument from Sequoia[®]. The principle of the operation is based on laser diffraction. This instrument provides a semi-log distribution of the volumetric concentration of particles over a 44 bands spectrum from 0.4 to 460 μm .

Further, the “sedimentation technique” was realized using a System for the Characterization of Aggregates and Flocs (SCAF), which was recently developed in our laboratory (Gratiot et al., 2015). SCAF consists in an optical settling column, equipped with a vertical array of 16 turbidity sensors to provide light transmission through a suspension during quiescent settling. With SCAF, the index of flocculation of particles is obtained as the ratio between settling velocities measured in the upper part and near the bottom of the SCAF settling tube (Wendling et al., 2015). For a solution of non-cohesive particles, settling velocity remains constant in the SCAF and the flocculation index is close to zero. For a solution with cohesive particles, flocculation enhances sedimentation and the flocculation index increases. A flocculation index equaling one means that

the average settling velocity of flocs population has doubled during sedimentation in the 16 cm high SCAF settling tube. For further details on the characterization of hydrodynamics and flocs properties in the Jar test, one can refer to Gratiot et al. (2017).

2.5. Sorption isotherm

Langmuir isotherm was used to evaluate the adsorption and desorption of P onto sediment (Wang and Li, 2010; Rossi et al., 2012; Vilmin et al., 2015; Aissa-Grouz et al., 2016); the adsorbed P onto SS is described by:

$$\frac{[PIP]}{[SS]} = P_{ac} \times \frac{[DIP]}{[DIP] + K_{ps}} \quad (1)$$

PIP is particulate inorganic phosphorus (mgP g^{-1}); SS is suspended sediment concentration (mg L^{-1}); DIP is PO_4^{3-} concentration (mgP L^{-1}); P_{ac} is the maximal adsorption capacity of P onto SS (mgP g^{-1}); K_{ps} is the half-saturation concentration of PO_4^{3-} (mgP L^{-1}). The concentration of PIP can be estimated by subtracting the content of total phosphorus (TIP) to DIP in the water ($\text{PIP} = \text{TIP} - \text{DIP}$). Replacing the TIP value in the above equation, DIP concentration can be deduced by the following equation:

$$DIP = \frac{TIP - P_{ac} \times SS - K_{ps} + \sqrt{(-TIP + P_{ac} \times SS + K_{ps})^2 + 4 \times TIP \times K_{ps}}}{2} \quad (2)$$

Phosphorus sorption isotherms are used to identify the capacity of sediment to sorb P and also to compare the sorption capacity for the different experimental conditions. The two parameters (P_{ac} and K_{ps}) of the Langmuir isotherm were determined by fitting relation (2) to the measured DIP vs TIP, using the criterion of least square deviation (Aissa-Grouz et al., 2016). An example of the result is shown in Figures 4-7a and 4-7d.

3. Results

3.1. Physico-chemical parameters distribution along the Saigon River

The Saigon River depth fluctuates between 5 to 18 meters with a deeper channel in the upstream section and a shallower channel at the confluence between Saigon and Dongnai Rivers. On the whole, the vertical distribution of physico-chemical parameters was rather homogeneous within

the water column except toward the confluence, where a near bottom stratification of TSS developed (Figure 4-3). The water temperature was stable along the Saigon River, averaging 31.1 ± 0.1 °C and 28.6 ± 0.2 °C during the dry season and the wet season, respectively.

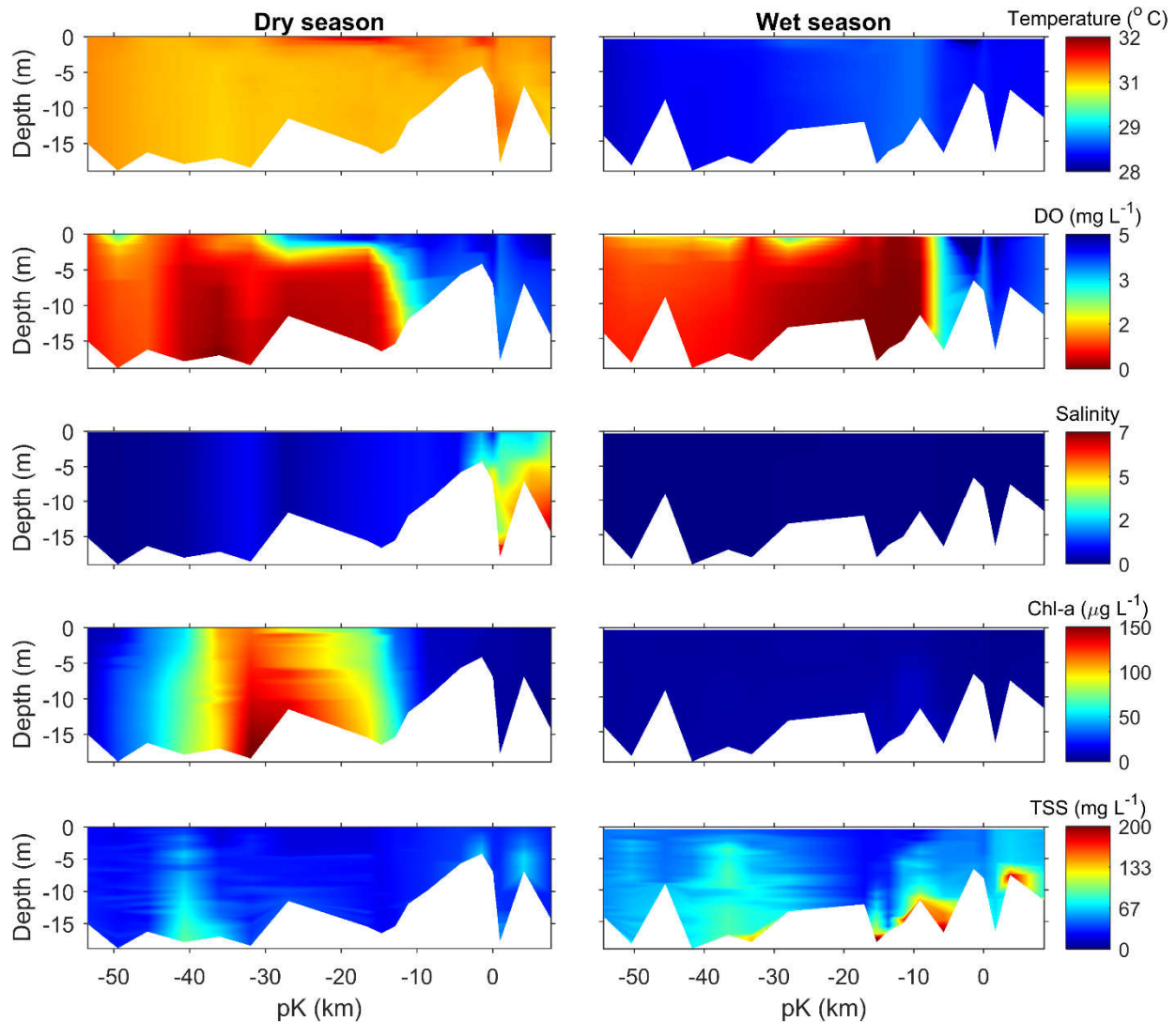


Figure 4-3. Temperature, dissolved oxygen (DO), salinity, Chlorophyll a (Chl-a) and total suspended sediment (TSS) variations from upstream to downstream of HCMC during the dry season (April 2017) and the wet season (October 2017).

Dissolved oxygen (DO) values varied between moderate to severe hypoxic conditions (as low as 0.5 mg L^{-1} in the upstream stretch of HCMC and rose up to 4.7 mg L^{-1} in the downstream one). During the wet season, DO level increased rapidly from extremely low concentration upstream of HCMC (pK -10 km, SG11 with DO value below 2 mg L^{-1}) to higher value downstream (pK 1 km,

SG16, with DO value above 4 mg L⁻¹). Saline water intrusion during the dry season was observed up to pK -40 km. Salinity was zero during the wet season along the Saigon River (Figure 4-3). Chl-a concentration was clearly higher in HCMC Center (from 8.8 µg L⁻¹ at surface in pK -18 km to 150µg L⁻¹ in pK -8 km) during the dry season) than upstream (pK -45 km to -23 km) and downstream (pK -4 km to 8 km) of HCMC. The magnitude of TSS fluctuations was lower during the dry season (from 16 mg L⁻¹ to 38 mg L⁻¹ near surface and from 23 mg L⁻¹ to 95 mg L⁻¹ near bottom) than during the wet season (from 24 mg L⁻¹ to 67 mg L⁻¹ near surface and from 31 mg L⁻¹ to 297 mg L⁻¹ near bottom) (Figure 4-3).

3.2. Phosphorus levels along the Saigon River

Deposited sediment. Throughout the two seasonal surveys, TPP and PIP respectively averaged 0.7 ± 0.4 and 0.6 ± 0.3 gP kg⁻¹ during the dry season and 1.1 ± 0.5 and 0.9 ± 0.4 gP kg⁻¹ during the wet season (Figure 4-4a and 4-4b, respectively). For all sampling sites, PIP fraction dominates TPP and accounted for 88 ± 7 % for both wet and dry seasons.

Suspended sediment. TPP contents ranged from 0.4 to 2.5 gP kg⁻¹ during the dry season (Figure 4-4a) and from 0.7 to 2.6 gP kg⁻¹ during the wet season (Figure 4-4b). P in SS decreases from upstream to downstream, especially during the dry season.

Water column. Total P fluctuated slightly from the upstream to the downstream sector of HCMC during the dry season, and ranged from 0.065 to 0.162 mgP L⁻¹ (Figure 4-4c). During the wet season, Total P was higher in the urban center of HCMC (0.213 ± 0.03 mgP L⁻¹ in average from pK -18 km to pK -8 km, SG07 to SG12), and decreased after the confluence between the Saigon and the Dongnai River (from SG16 to SG18) down to 0.09 mgP L⁻¹ in average (Figure 4-4d). Total P was clearly higher during the wet season in comparison to the dry season of the year 2017, 0.19 ± 0.07 and 0.11 ± 0.03 mgP L⁻¹, respectively. While proportion of PP highly varied along the Saigon River (from 6.4 to 87.7 %) during the dry season, PP in Total P was systematically higher than 45 % during the wet season. These high percentages of PP show propensity for adsorption of P onto sediments (see below).

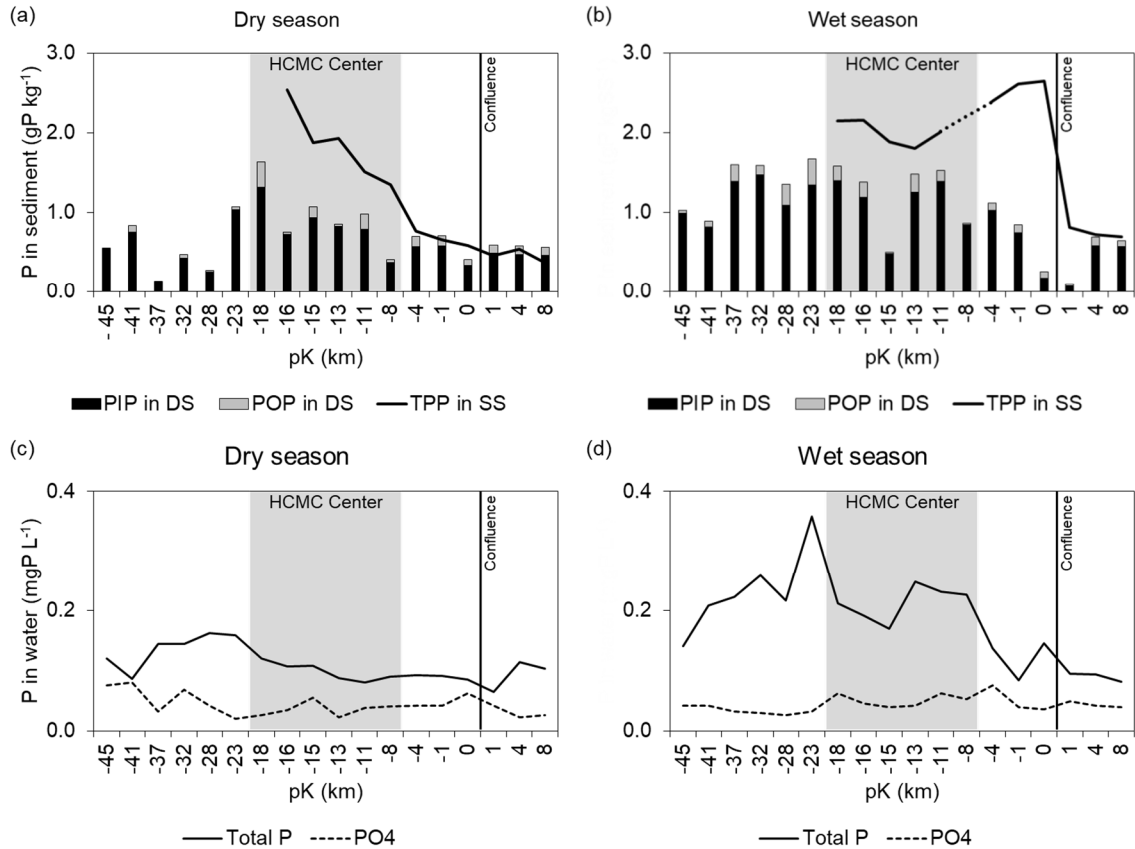


Figure 4-4. (a and b) Particulate phosphorus in deposited sediment (DS) and suspended sediment (SS) and (c and d) Total phosphorus and PO₄³⁻ concentrations in the water column during the dry season and the wet season. The vertical line indicates the confluence with the Dongnai River.

3.3. Phosphorus distribution in the salinity gradient

The Figure 4-5 describes the observed distribution of particulate and dissolved phosphorus concentrations within the salinity gradient, which expressed the transformation (adsorption or desorption) of dissolved phosphorus during the mixing of fresh water and salt water throughout the estuary. Because we did not observe the salinity gradient (salinity ≈ 0) during the wet season, we showed data only during the dry season. We clearly observed that TPP content in SS decreased within the salinity gradient, from 2.54 gP kg⁻¹ (for salinity below 1) to 0.48 gP kg⁻¹ (for salinity up to 8) (Figure 4-5a). The decrease in TPP content in SS resulted mainly from a decrease in PIP concentration (from 1.5 gP kg⁻¹ at salinity below 1 to 0.4 gP kg⁻¹ at salinity up to 8, while POP remained stable (0.2 ± 0.1 gP kg⁻¹) (Figure 4-5b). Consequently, PIP/POP ratio equaled to 4.4 in

average for salinity lower than 1 and decreased to a value close to 2.6 for the maximum salinity observed (Figure 5c). Such PIP behavior could represent a desorption of P from SS, in order to equilibrate the PO_4^{3-} concentrations in the river water, diluted by seawater poorer in P, as shown in the PO_4^{3-} profiles within the salinity gradient (Figure 4-5d).

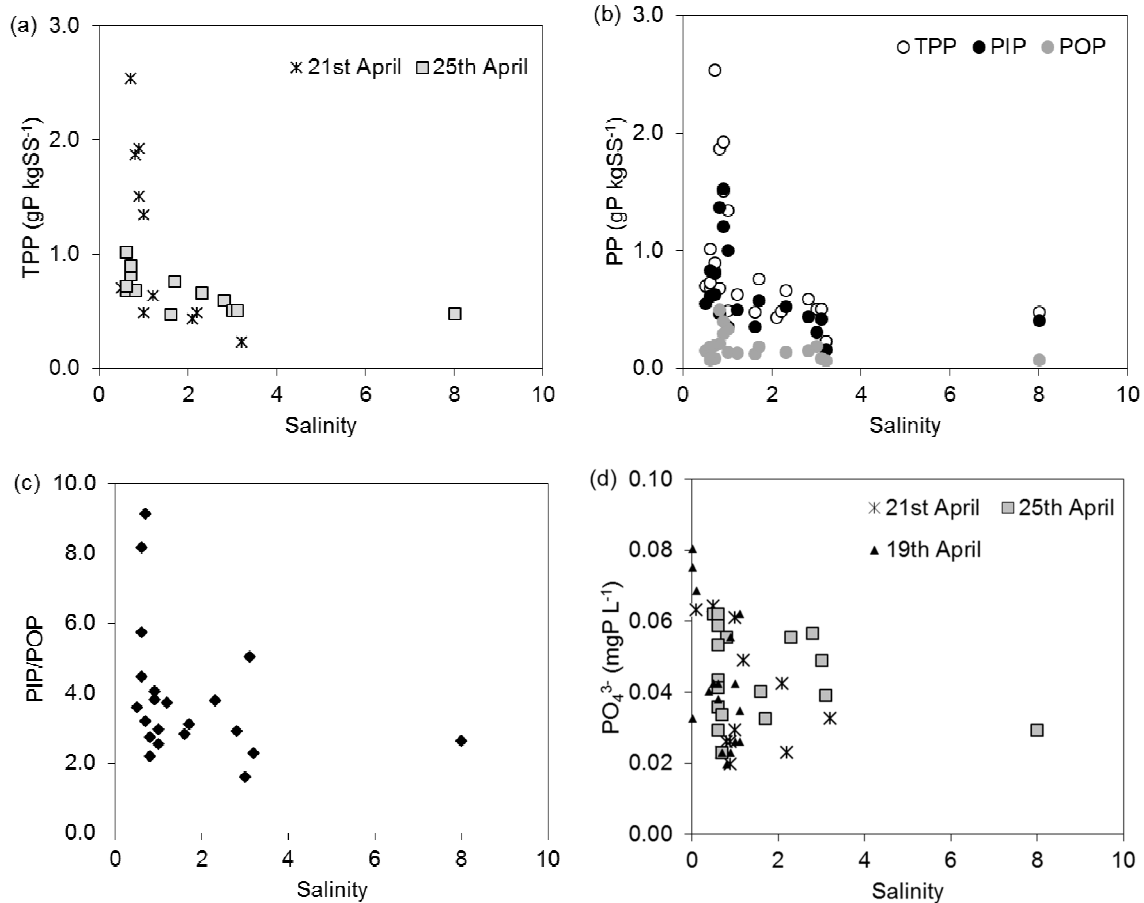


Figure 4-5. (a) Distribution of Total particulate phosphorus, (b) Distribution of particulate phosphorus, (c) PIP/POP ratios and (d) PO_4^{3-} concentration within the salinity gradient during the dry season.

3.4. Phosphorus adsorption capacity onto sediment from laboratory experiments

Phosphorus level. The selected sediment was sampled in the downstream part (at SG13; pK - 4 km, see Figure 4-1b) where P transformations were evidenced between salinity 0.6 to 8.0 during the dry season survey. Initial P content of sample SG07 was 0.81 and 0.90 gP kg^{-1} for PIP and TPP, respectively. The change in P sorption capacity onto suspended sediment was linked to the

levels of P added. While original P in SS was released at condition without P added (%DIP adsorbed < 0), at P added concentrations higher than 0.1 mgP L⁻¹, P was adsorbed (%DIP adsorbed > 0) (Figure 4-6). The adsorption capacity was high for small concentration of added P. At a low concentration of additional P (< 0.1 mgP L⁻¹), the amounts of P adsorbed varied between 0.062 gP kg⁻¹ and 0.064 gP kg⁻¹ (i.e. 15.5 % and 64.1 % of P added). However, the amounts of P adsorbed were 0.109 gP kg⁻¹ to 0.520 gP kg⁻¹ (i.e., 0.9 % to 15.6 % of P added) at the highest P added concentration (3.0 mgP L⁻¹) (Figure 4-6). This shows that (i) a greater proportion of the added P is adsorbed at low P concentrations and (ii) the reduction of this proportion is related to the increase of additional P.

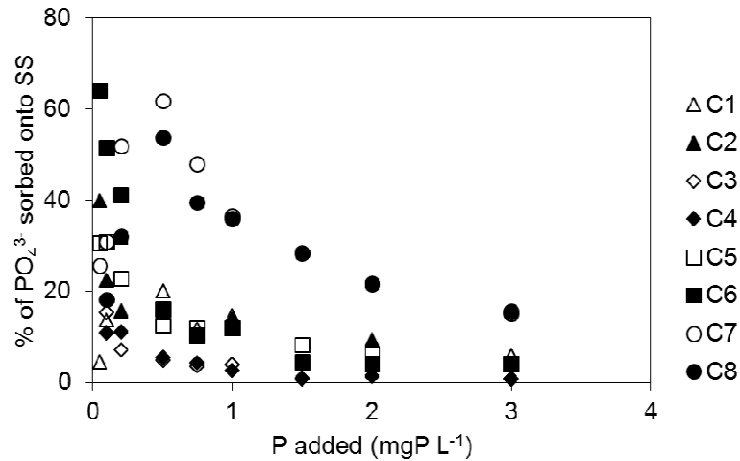


Figure 4-6. The percentage of PO_4^{3-} adsorbed onto suspended sediments (SS) at different hydrodynamic conditions (see Table 1 for C1 to C8 conditions). Sediment was sampled at site SG13 during the dry season (25th April 2017).

We also observed that when SS increased from 0.25 – 0.5 – 0.9 g L⁻¹, the percentage of P added adsorbed onto SS increased from 13 – 36 – 58 % at low P added (< 0.1 mgP L⁻¹) to 1 – 8 – 16 % at highest P added (3.0 mgP L⁻¹), respectively (Figure 4-6).

For all hydrodynamic conditions experimentally tested (SS, turbulence, salinity), TIP increased rapidly with the additional PO_4^{3-} , and then reached a plateau; the kinetics of sorption of P was typical of Langmuir isotherm curves (Figure 4-7a and 4-7d). The fitting of experimental points to Langmuir isotherm equation was good (R^2 higher than 0.98), allowing to examine the variation of P_{ac} and K_{ps} for all hydrodynamic conditions considered (Table 4-3). The maximal adsorption of P

onto SS (P_{ac}) and the half-saturation concentration of PO_4^{3-} (K_{ps}) ranged from 1.075 to 1.301 $gP\ kg^{-1}$ and from 0.004 to 0.035 $mgP\ L^{-1}$, respectively.

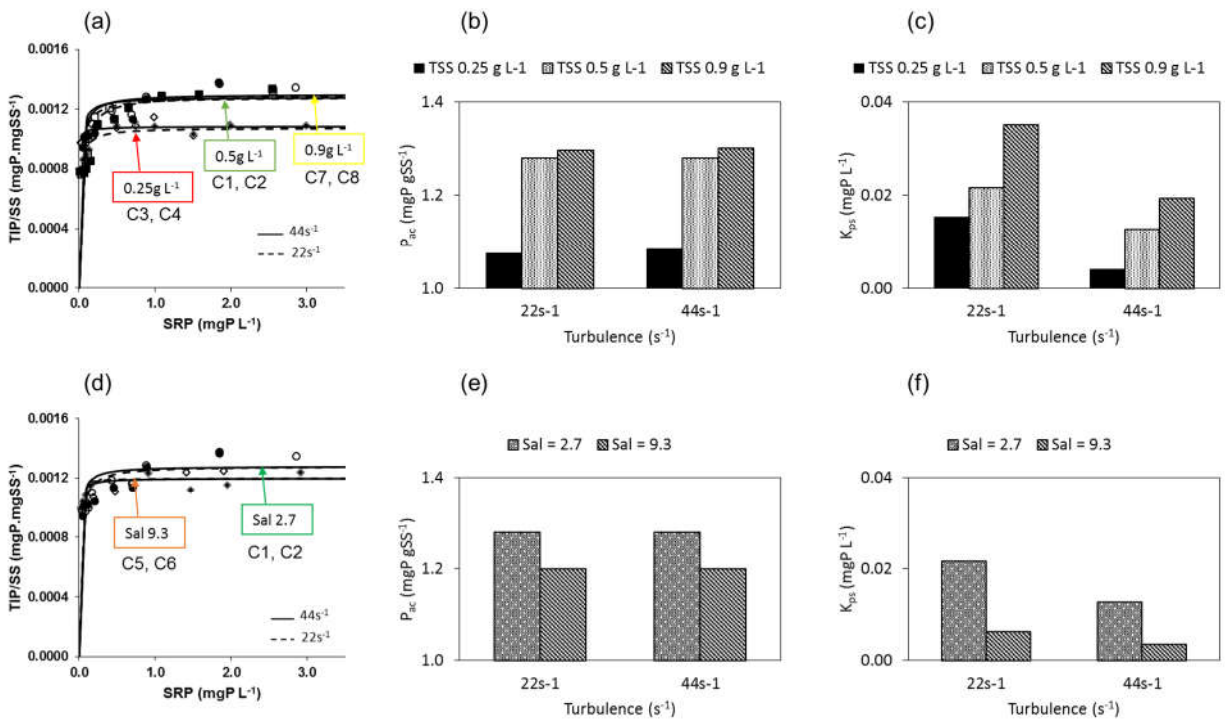


Figure 4-7. (a) and (d) Kinetics of sorption of phosphorus and (b), (c), (e) and (f) the sorption parameters (P_{ac} , K_{ps}) for all hydrodynamic conditions tested. Correlation coefficient (R^2) for each parameter determination is shown in Table 3.

The evolution of sorption parameters with turbulence (22s⁻¹ and 44s⁻¹) for three gradually increased values of suspended sediment concentration (SS represented by bars in Figure 4-7b and 4-7c) pointed out a positive influence of SS on sorption capacity (P_{ac} increased by nearly 20% when SS increases, without effect according to the intensity of turbulence (Figure 4-7b). From the results of P_{ac} and K_{ps} values presented in Table 3, SS was the main factor driving the adsorption capacity. For example, P_{ac} and K_{ps} values increased from 1.084 to 1.301 $mgP\ gSS^{-1}$ and 0.004 to 0.109 $mgP\ L^{-1}$ respectively, within the range of SS [0.25– 0.9 $g\ L^{-1}$].

However, the increasing turbulence had a decreasing effect on the half-saturation content for all SS tested (Figure 4-7c). Indeed, K_{ps} increased gradually with SS but decreased from 0.013 to 0.022 $mgP\ L^{-1}$ with turbulence increasing from 22 s⁻¹ to 44 s⁻¹ (see condition C1 and C2 in Table 2).

Regarding the effect of salinity, the maximum sorption capacity P_{ac} decreased at higher salinity, and similarly for the two turbulence levels (Figure 4-7e). K_{ps} value was lower for high salinity, and lower at higher turbulence levels (Figure 4-7f). For example, at turbulence 44 s^{-1} , P_{ac} and K_{ps} decreased with salinity respectively from 1.280 to 1.199 mgP gSS^{-1} and from 0.013 to 0.004 mgP L^{-1} when salinity increased from 2.7 to 9.3.

To summarize, our experimental set-up showed a predominant effect of suspended sediment (SS) on sorption processes followed by salinity and then turbulence which solely modified the half saturation constant.

Table 4-3. Maximal adsorption capacity of phosphorus onto suspended sediment (P_{ac}), half-saturation concentration of PO_4^{3-} (K_{ps}), correlation coefficient (R^2) of the fitting and the percentage of PO_4^{3-} absorbed onto SS at the different experiment conditions (in 1st column: C1, number of samples for each treatment).

	SS	Turbulence	Salinity	P_{ac}	K_{ps}	R^2	% of PO_4^{3-} absorbed	
	g L^{-1}	s^{-1}		mgP g^{-1}	mg L^{-1}		Max	Min
C1	0.5	44	2.7	1.280	0.013	0.991	31.9	5.8
C2	0.5	22	2.7	1.280	0.022	0.991	40.0	9.4
C3	0.25	44	2.6	1.084	0.004	0.999	15.5	0.9
C4	0.25	22	2.6	1.075	0.015	0.999	10.9	0.9
C5	0.5	44	9.3	1.200	0.006	0.994	31.1	4.1
C6	0.5	22	9.3	1.199	0.004	0.996	64.1	4.1
C7	0.9	44	2.7	1.301	0.019	0.987	61.8	15.3
C8	0.9	22	2.7	1.296	0.035	0.991	53.9	15.6

3.5. Physical characteristics of sediments

A first analysis of the Particle Size Distributions (PSDs) confirmed the presence of both cohesive and non-cohesive sediments. The high majority of particles (55-95% in volume) were cohesive sediments, which are partially or almost completely disaggregated after two minutes of sonication. Conversely, the non-cohesive particles (i.e. no effect of sonication: silt, sand and others) only represented 5 to 45 % (by volume concentration) of particles' population, without any evident trend between trials C1 to C8. These macro-particles (higher than 200 microns) form a population that is assumed to be poorly active in sorption processes, contrarily to the cohesive ones. Figure 4-8, synthesizes the results for cohesive particles populations.

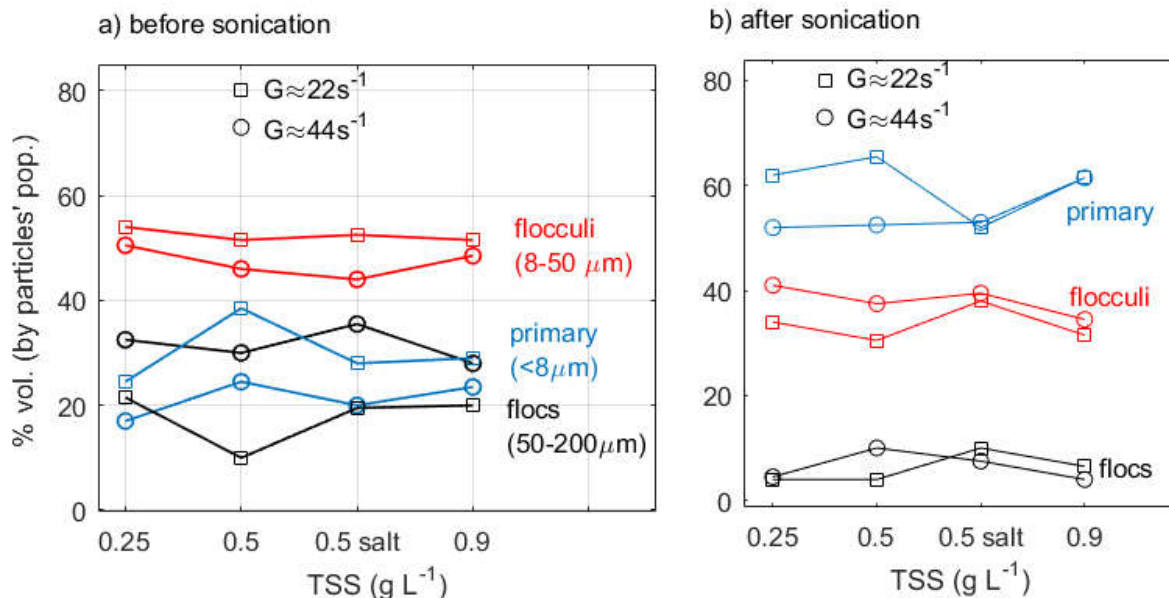


Figure 4-8. Variation of the volumic concentration of particles (primary $< 8 \mu\text{m}$, flocculi $[8-50 \mu\text{m}]$ and flocs $[50-200 \mu\text{m}]$) with TSS, for the experiments C1 to C8 (see Table 2).

Whatever the trial considered (C1 to C8), flocculi population predominates. It corresponds to 44-54% of the samples while primary particles and flocs correspond to [17-38] and [10-35] of the volumic concentration, respectively. The large predominance (65-90%) of the smallest particles implies that sediments of the Saigon River stay a very long time in suspension in the water column, have a high specific surface (a key parameter for biogeochemical processes) and will deposit only for very low flow conditions.

After sonication, 75% of flocs population and 40% of flocculi were broken. The intercomparison of the volumic concentration of primary particles before (~25%) and after (~60%) sonication, shows that they are highly mobile and directly flocculate on both flocculi and flocs.

Concerning the flocculation under nondestructive “sedimentation technique”, the pattern showed a clear and gradual increase of flocculation index (FI) with SS concentration, which doubled from FI = 0.4 to 0.8 when SS rose from 0.25 to 0.9 g L⁻¹ (Figure 4-9). Salinity also favored flocculation but in a less efficient way (FI increased by 20 %). The turbulence level existing in Jar test before putting the particles in the SCAF settling tube did not affect flocculation.

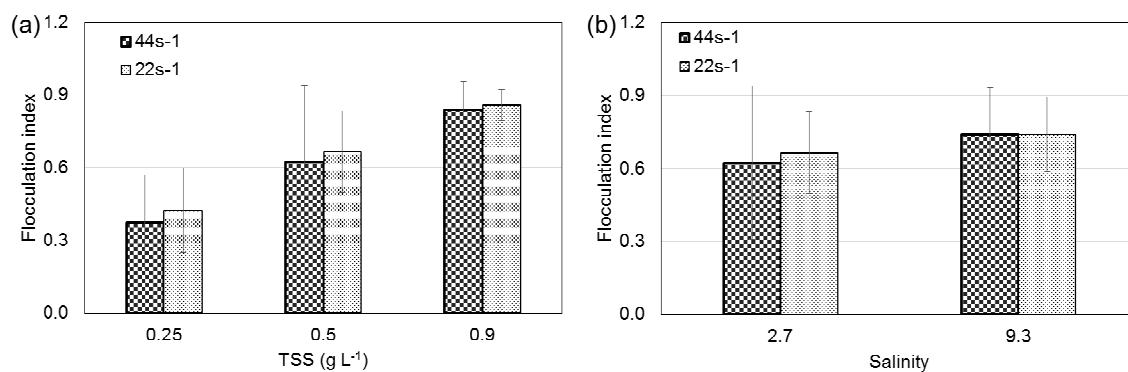


Figure 4-9. Effect of suspended sediment concentration and salinity on flocs and flocculation index at different experimental conditions.

4. Discussion

4.1. The effect of physical characteristic of sediment on the P adsorption capacity

The capacity of SS to sorb phosphorus is originally assessed in this paper, by coupling field measurements, sorption experiments, Langmuir isotherm and the examination of physical characteristics of sediment. The values of P_{ac} ranged from 1.07 to 1.30 gP kg⁻¹ in the Saigon River. This P_{ac} value is lower than those in the sediments of Mai Po Marshes in Hong Kong (from 1.64 to 3.58 gP kg⁻¹), which have been attributed to the availability of abundant amorphous Fe, Al and organic matter (Lai and Lam, 2008). In contrary P_{ac} in the Saigon River was higher than in sediment of Indian River lagoon, Florida (from 0.032 gP kg⁻¹ under anaerobic conditions to 0.132 under aerobic conditions gP kg⁻¹, Pant and Reddy, 2001), Kissimmee River wetlands (0.011 – 0.826 gP kg⁻¹, Reddy et al., 1998) and in the same range of values found in the Seine River with

an impact of the release of the treated wastewaters (Aissa-Grouz et al., 2016). Higher value was associated with the high concentration of SS.

In the Saigon River, the higher P_{ac} values obtained at low salinity rather than at high salinity evidenced higher P adsorption capacity on SS in low than in more saline water. Beside higher competition between anions, increasing salinity beyond a threshold value would lead to decrease positive charges of metal hydroxides in SS, and inhibiting P sorption (Barrow et al., 1980). The change of turbulence, from 22 s^{-1} to 44 s^{-1} , typical of natural estuaries (Marion et al., 2016), did not affect P_{ac} values which remained stable. The particle concentration effect (i.e., the change in the adsorption isotherm in response to changing particle concentration) could explain PO_4^{3-} desorption in the salinity gradients.

As described above, SS concentration controls the flocculation of cohesive sediments. The change in the flocculation index with increasing SS can thus explain the higher P content in SS. It means that sediment dynamics could be one of the main driver of the fate of P in the Saigon River estuary.

4.2. Impact of HCMC on the P concentrations

All samples collected within the salinity gradient campaigns during the dry and wet seasons in 2017 allowed to analyze the P content in SS for two vertical distributions: surface and bottom of water column (Table 4). A difference in P content (TPP, PIP and POP) was observed between the suspended sediment and bottom suspended sediment samples (e.g., higher values in surface samples in comparison to bottom samples during the wet season, 1.45 ± 0.77 and $1.03 \pm 0.39 \text{ mgP kg}^{-1}$, respectively). This result would indicate that the adsorption/desorption of P occurred principally within the water column.

During both 2017 season surveys, a reduction of TPP concentration in SS from the HCMC Center (pK -18 to -8 km) to the downstream section was observed and more during dry than wet season. During the dry season, it can be explained by the fact that the increasing salinity in the downstream of HCMC (pK -4 to 8 km) resulted in a greater competition among anions for positively charged binding sites due to the increase in Cl^- in tidal waters (Stumm and Morgan, 1981). We also observed that the amount of P sorbed onto bottom sediments decreased toward the saltwater end of the gradient (as reported by Sundareshwar and James, 1999). These results were also reported

by Berner and Rao (1994) with a significant decrease of TPP within the salinity gradient in the Amazon River.

Similarly to TPP, PIP in SS were also lower during the dry season than those values during the wet season (Table 4-4). Same trend was observed with Total P in waters. It may be related to the flushing of sediment from urban canals of HCMC with high content of phosphorus during the wet season; a similar hypothesis of high flushing in the beginning of wet season was recently proposed by Babut et al. (2019). It is also shown that during the dry season P was largely uptaken by algae, especially between pK -40 and -10 km where chlorophyll a concentrations raised up to $150 \mu\text{g L}^{-1}$ (Figure 4-4). The eutrophication of the Saigon River was clearly stated in a previous study that also identified P among other key nutrients (nitrogen and silica) as the main limiting factor of algal growth (Nguyen et al., 2019).

By dividing the Saigon River into three sections (upstream: pK -45 to -23 km, HCMC Center: pK -18 to -8 km, downstream: pK -4 to 8 km), we can assess the impact of urbanization on the P content variability in the sediment along the Saigon River (Table 4-4). Deposited sediment in the HCMC Center were characterized by high TPP and PIP contents during both dry and wet seasons, resulting in a P enrichment from untreated domestic wastewater discharges (Nguyen et al., 2019). We assume that this high enrichment is due to the PO_4^{3-} adsorption from wastewater discharge, which was also reported as a typical process by Némery and Garnier (2007) from a study conducted in the Seine River. The effect of high residence time of water and particles due to low net discharge and the effect of high water mixing by tides can also lead to high potential retention of P. This is particularly the case in the urban canals, which were under contaminated conditions by the release of untreated wastewater, like in Nhue River, near Hanoi (Trinh et al., 2009). High concentration of PO_4^{3-} in water and TPP in sediment were observed in urban canals (Strady et al., 2017) indicating an intense adsorption process of dissolved P onto sediment. This kind of accumulation in the sediment was also evidenced in the urban canals of Bangkok (Thailand), which presents geography, demography and wastewaters management characteristics similar to our case study (Færge et al., 2001). It appeared that Total P in water is mainly composed of particulate fraction, making up $73 \pm 20 \%$ ($n = 98$ samples). PIP fraction generally dominated in suspended sediment column as well as in the bottom sediment of the river (i.e., $75.8 \pm 8 \%$ of TPP values). POP represented a small fraction (less than 25 %) of the TPP. POP proportion in SS was in the same

range (25 ± 6 % in both the HCMC Center and the downstream of HCMC), which indicated that TPP decrease was due to PIP decrease. A stability of POP content indicated that TPP decrease is due to PIP decrease. The high variability of TPP is more related to that of PIP than to the variability of POP. The PIP/POP ratio fluctuates according to the PIP variations.

Table 4-4. Mean (\pm SD) of total particulate phosphorus (TPP), particulate inorganic phosphorus (PIP) and particulate organic phosphorus (POP) content in suspended sediments (SS) during both dry season (April 2017) and wet season (October 2017).

	Number of sample	TPP (gP kg ⁻¹)	PIP (gP kg ⁻¹)	POP (gP kg ⁻¹)
		Dry season (April 2017)		
Suspended sediment				
Surface	18	1.14 (0.73)	0.77 (0.47)	0.24 (0.14)
Bottom	7	0.72 (0.20)	0.60 (0.17)	0.12 (0.05)
All samples	25	0.98 (0.61)	0.70 (0.37)	0.19 (0.12)
Deposited sediment				
Upstream	6	0.56 (0.35)	0.52 (0.33)	0.04 (0.03)
HCMC Center	6	0.95 (0.41)	0.82 (0.31)	0.12 (0.12)
Downstream	6	0.59 (0.11)	0.48 (0.09)	0.10 (0.02)
		Wet season (October 2017)		
Suspended sediment				
Surface	18	1.80 (0.73)	1.34 (0.61)	0.46 (0.18)
Bottom	7	1.16 (0.50)	0.88 (0.41)	0.28 (0.13)

<i>All samples</i>	25	1.55 (0.71)	1.16 (0.57)	0.39 (0.18)
Deposited sediment				
Upstream	6	1.35 (0.33)	1.18 (0.26)	0.17 (0.11)
HCMC Center	6	1.22 (0.44)	1.09 (0.36)	0.13 (0.09)
Downstream	6	0.60 (0.38)	0.52 (0.36)	0.08 (0.03)

4.3. Implication for the understanding of P dynamics within estuaries

Sediment play an important role in the adsorption-desorption of P in the estuaries (Zhang et al., 2004). The reactivity of the particles with respect to phosphorus is therefore dependent on estuarine hydrodynamics and more precisely on the variation of the sediments physical properties (Statham, 2012). For the case study of the Saigon River, we hypothesize that the decrease in P concentration in the downstream of HCMC sector were due to (i) a PIP desorption within the salinity gradient with respect to the changes in the physical characteristics of sediments and (ii) a dilution by water with low P concentration coming from the mixed waters from Dongnai River.

Our experimental results showed here an important role of SS concentrations in the sorption capacity of P onto sediments. Sedimentary dynamics in estuaries (especially mechanisms that control the flocculation of cohesive sediments) is indeed influenced by both the gradient of salinity and the level of turbulence (Keyvani and Strom, 2014). These two variables are governed by tidal cycles and seasonal variations in water flows that are highly contrasted in the tropical climate of Vietnam (Lefebvre et al., 2012, Mari et al., 2012). The change in the flocculation index with increasing TSS, especially in wet season as observed in our study (Figure 4-7b), can thus explain the higher P content in TSS at that time (Figure 4-4b). It means that sediment dynamics could be one of the main driver of the fate of P in the Saigon River estuary.

Besides, the decrease of P content downstream could also be due to the dilution effect by water with low P concentration coming from the Dongnai River. Nguyen et al. (2019) indeed observed that mean annual discharge in the Dongnai River was twelve times higher than in the Saigon River, while mean annual Total P concentration was about $0.09 \pm 0.05 \text{ mgP L}^{-1}$ in the Dongnai River,

lower than in the Saigon River ($0.30 \pm 0.20 \text{ mgP L}^{-1}$). Dilution from the Dongnai River supported the second hypothesis and showed that the confluence between the two rivers may play an important role in the P dynamics within the estuary.

Within the salinity gradient, we can divide P dynamics into two phases based on PO_4^{3-} concentration (presented in Figure 4-5d): a stable of PO_4^{3-} concentration in low salinity (0-4) and a decrease of PO_4^{3-} concentration in high salinity (>4). At low salinity, PIP desorption process, which resulted in an increase of PO_4^{3-} concentrations in the water column, would be stronger than the dilution effect by water with low P concentration coming from the Dongnai River. Both waters created a stable zone of PO_4^{3-} concentration in low salinity (0-4). By contrast at high salinity, phosphorus desorption increased much (see the decrease in P_{ac}). It means that a dilution by water from the Dongnai River was stronger than PIP desorption process. This lead to decrease in PO_4^{3-} concentration in high salinity (> 4 , see Figure 4-5d).

Table 4-5. Phosphorus availability within salinity gradient.

Sector	Salinity	TSS level	Flocculation	P adsorption capacity	Dilution	P availability
HCMC Center	+	+++	+++	+++	+	++
Downstream before confluence	++	++	++	++	++	++
Downstream after confluence	+++	+	+	+	+++	+

Table 4-5 synthesizes and conceptualizes the observations made on the Saigon River. The Saigon River suffers from eutrophication in its medium section, which is a logical response to untreated wastewater discharges (Nguyen et al., 2019). However, sediment dynamics appear to effectively buffer excess phosphorus concentrations, particularly through flocculation and sediment adsorption capacity. To this is added a strong dilution capacity by the main confluence (the Dongnai River). Also the Saigon River estuary seems to be spared by the impact of untreated wastewater discharges from HCMC (low algal development, good oxygenation, lower phosphorus levels). This situation is rather unusual compared to estuaries subjected to the same anthropogenic

pressures (Flemer and Champ, 2006) and indicates the great potential for metabolizing of these tropical estuaries

5. Conclusions

Phosphorus concentration in HCMC center was about three times higher than in upstream and downstream of HCMC. Particulate inorganic P represented 80% of total particulate P, as a result of adsorption of PO_4^{3-} released by untreated wastewater discharges. Our results illustrated the strong influence of SS concentrations on flocculation and on the P adsorption onto sediment, e.g. P adsorption capacity (P_{ac}) and adsorption velocity (K_{ps}), increase when SS concentrations rose followed by salinity. In contrast, turbulence modified very slightly adsorption properties of sediment. The Saigon River estuary appeared to be a very reactive biogeochemical system which is able to metabolize an important proportion of P emitted by the megacity of HCMC. The downstream part of the estuary seems to be slightly affected by the urban discharges, due to both positive interactions between P and sediment and because of dilution effect with Dongnai River at the confluence.

As a perspective, mathematical formulation of these experiments will be implemented in the biogeochemical model Carbon-Generic Estuary Model (C-GEM) for a better representation of P processes (Volta et al. 2014). This tool was developed to model the biogeochemical dynamics of carbon and associated bio-elements (N, P and Si) in the tidal estuarine system. The modelling applications are appropriate to solve the spatio-temporal variability of nutrient inputs in the estuarine systems by reproducing the hydrodynamics and the biogeochemical processes. This model is on the way to be implemented in the Saigon River.

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Conclusion of Chapter 4

Phosphorus concentration in the center of HCMC was higher than in upstream and downstream of HCMC. Our results illustrate the strong influence of SS concentrations on the P adsorption onto sediment. In contrast, turbulence modified very slightly adsorption properties of sediment. The Saigon River estuary appeared to be a very reactive biogeochemical system which is able to metabolize an important proportion of P emitted by the megacity of HCM. Because of the rapid development of population and economic in the HCMC and neighboring provinces, the water quality is deteriorating by a significant increase in nutrient fluxes. In order to better understand the current changes and their impacts, we provide a database to calculate stoichiometric ratios and nutrient fluxes to evaluate the influence of human activities on water quality, which will be presented in Chapter 5.

**CHAPTER 5. NUTRIENTS BUDGET IN THE SAIGON – DONGNAI RIVERS BASIN:
PAST TO FUTURE INPUTS FROM THE DEVELOPING MEGACITY OF HO CHI
MINH (VIETNAM)**

This chapter is prepared to be submitted to the “Journal River Research and Application (Special Issue IS Rivers meeting 4-8 June 2018 Lyon, France)”. This chapter is the results from the collaboration between CARE-HCMUT and CEM-DONRE, the operational Center of Ho Chi Minh City in charge of the monitoring of air and water pollution. The database of water discharge and nutrient concentrations from 2012 to 2016 was used to calculate suspended sediment and nutrient fluxes within the Saigon – Dongnai Rivers system and interpret its evolution. This chapter is divided in three sections that (i) provide a complete database gathering information on the water discharge and on the concentration of nutrients that are rejected by HCMC to the Saigon River (including wastewater quality from domestic and industrial activities); (ii) elucidate the complete water circulation and calculate synthetic spatial water budget; and (iii) evaluate the inputs of nutrients from domestic discharge, industrial activities, atmospheric deposition and activities from upstream part of the Saigon River.

NUTRIENTS BUDGET IN THE SAIGON – DONGNAI RIVERS BASIN: PAST TO FUTURE INPUTS FROM THE DEVELOPING MEGACITY OF HO CHI MINH (VIETNAM)

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Abstract

Ho Chi Minh City (HCMC, Vietnam) is one of the fast-growing megacity in the world. In this paper, we attempt to analyze the dynamics of nutrients (nitrogen, phosphorus and silica), suspended sediments and water discharges in the aquatic systems which are connected with the megacity, from 2012 to 2016. The work focuses on nine sites along the Saigon River and one site in the Dongnai River to identify the reference water status before the urban area and the increasing fluxes from upstream to downstream. The calculated fluxes allow drawing a first sediment and nutrients budget at the scale of the watershed and discussing the contribution of each sub basins to the total fluxes to the estuarine and coastal zones. Seasonal and inter annual variability of these fluxes are also presented and discussed to assess HCMC contributions and its impacts downstream. Based on a monthly monitoring of the water discharge (Q), nutrients (phosphorus and nitrogen) and suspended sediment (SS) concentration, we estimated the mean annual SS and nutrients (NH_4^+ and PO_4^{3-}) fluxes at $292 \times 10^3 \text{ tonSS yr}^{-1}$, 791 tonN yr^{-1} and 111 tonP yr^{-1} for the Saigon River and $1175 \times 10^3 \text{ tonSS yr}^{-1}$, $2132 \text{ tonN yr}^{-1}$ and 806 tonP yr^{-1} for the Dongnai River, respectively. Seasonal and spatial variations are clearly linked to hydrology. Nutrient fluxes provide an indicator

of coastal eutrophication potential (ICEP), using nutrient stoichiometry ratios. The distinct shift of nutrient ratios (negative P-ICEP value), indicated a Phosphorus (P) limitation. Finally, we analyzed the future trends (2025-2050) for the nutrients inputs scenarios, based on expected population growth in HCMC and improvement of wastewaters treatment capacity.

Keywords: Sediment, nutrients, flux calculation, budgets, present and future scenarios

1. Introduction

In the 21st century, the impact of humans on ecosystems has become a major concern. The application of fertilizers in agriculture and the discharge of untreated domestic and industrial wastewaters into aquatic systems is significantly increasing (Seitzinger et al., 2010; Turner et al., 2003). As a direct consequence, human activities have changed the cycle of nitrogen (N) and phosphorus (P) at global scale (Meybeck, 1982; Seitzinger et al., 2005, 2010; Van Drecht et al., 2003), as well as at regional scale, with alarming consequences in various large rivers like Yangtze River (Liu et al., 2016), Mississippi River (Turner and Rabalais, 1994) and numerous European rivers (Ludwig et al., 2009, 2010).

An excess of nutrients leads to serious impacts on ecosystems, such as eutrophication in reservoirs, lakes and rivers with hypoxia and fish mortality (Conley et al., 2009; Smith, 1998). Algal blooms can also cause water quality problems such as unpleasant odors and an increase in pH and dissolved organic carbon (Carpenter et al., 1998; Shen et al., 2003; Smith, 1998). In addition, some algal species such as cyanobacteria can release harmful toxins that may affect livestock and human health (Duong et al., 2010a; 2010b; 2012).

Numerous studies have reported that the main cause of eutrophication in aquatic system were due to the increase of nutrient concentrations in both dissolved and particulate forms, but also to the changes in the nutrient stoichiometry, i.e. their molar ratios (Conley et al., 2009; Smith, 1998; Turner et al., 2003; Langenberg et al., 2003; Winter et al., 2002). Examining simultaneously the concentrations, forms and ratios is now recognized as a useful tool to predict the community component of undesirable algae production, both globally and regionally (Redfield et al., 1963; Turner et al., 1998, 2003).

According to this view, an Indicator of Coastal Eutrophication Potential (ICEP) was proposed by Billen and Garnier (2007). This indicator is based on a riverine N, P and Si inputs, and allow determining the possible production of non-siliceous algae. While N:P ratio is used to identify the limiting factor, the ICEP gives information about the role of Si in determining potential eutrophication impacts of changing element stoichiometry.

Several studies have dealt with the nutrient fluxes in tropical areas (Herrera-Silveira and Comin, 1995; Matson, 1993), Nui Coc reservoir catchment (Le et al., 2014), the Lower Mekong River (Li

and Bush, 2015) and Red River basin (Le et al., 2005, 2007; Luu et al., 2010) but so far, no nutrient budget has been done on the Saigon-Dong Nai aquatic system, despite the fact that this region, which hosts the megacity of HCMC, is strategic in terms of ecological and economic issues.

Ho Chi Minh City (HCMC) is the most dynamic city in Vietnam and one of the most dynamic in the world (World Economic Forum News, January 2017). Because of the urban sprawl, the quality of the water is deteriorating, which endangers the resource and the human activities that depend on it (Nguyen et al., 2011; Strady et al., 2017; Nguyen et al., 2019). The Saigon River not only provides water for HCMC water supply, but is also vital for irrigation, aquaculture and navigation.

In order to better understand the current changes and their impacts, we used a database provided by the Center of Environment Monitoring (CEM) of DONRE and compared it with data collected by our group, every fifteen days, from July 2015 to December 2017. The objectives of this paper are (i) to establish the annual budget of water, Total Suspended Sediments (TSS) and nutrients in the Saigon and Dongnai rivers system; (ii) to characterize the relative contribution of local sources (domestic discharge, industries, atmospheric deposition) and upstream sources to the global budget (iv) to assess possible shift of nutrients stoichiometric ratio to the estuarine zone and finally (v) to assess the sustainability of the region in 2025 and 2050 for realistic scenarios of population growth and urban development policy, with the establishment of new wastewater treatment plants.

2. Material and methods

2.1. Description of the Saigon – Dongnai River basin

The Saigon – Dongnai River basin (Figure 5-1a) is low lying area located in South-East Asia (from 10°40'N to 12°00'N and 106°10'E to 106°40'E) and drains an area of 31 216 km². The Saigon River (surface area of 4717 km²) originates from Phum Daung in south-eastern Cambodia, flows to the Dau Tieng Reservoir (120-270km²; 470-1680 million m³) before to reach the estuarine area. The regulation of Dau Tieng reservoir's water discharges aims at preventing the intrusion of saline water below the Phu Hoa water production plant and mitigating flood risks (Figure 5-1). Downstream the reservoir, the Saigon River flows through Ho Chi Minh City where the river is connected to urban canals and confluences with the Dongnai River to form the Soai Rap River, which then flows through the Can Gio mangrove and discharges into the coastal waters of the East Sea, 20km North of the Mekong Delta (Figure 5-1). Saigon River is highly influenced by the Dongnai River and coastal waters, as a response of the tidal forcing that leads to the flow current

inversion twice a day up to Dau Tieng reservoir. The Dongnai River originates from south-west plateau and its surface area is 26 499 km² at the confluence with the Saigon River. The Tri An reservoir located in the middle course of the Dongnai River was built in 1986 for hydroelectric production (surface 323 km², 2 700 million m³) (Dao et al., 2016).

Ho Chi Minh City (HCMC) is developing in this low elevation coast zone. The HCMC falls in a tropical monsoon climate, with an average humidity of 78-82%. The year is divided into two distinct seasons. The rainy season usually begins in May and ends in November. The dry season lasts from December to April (Tran Ngoc et al., 2016).

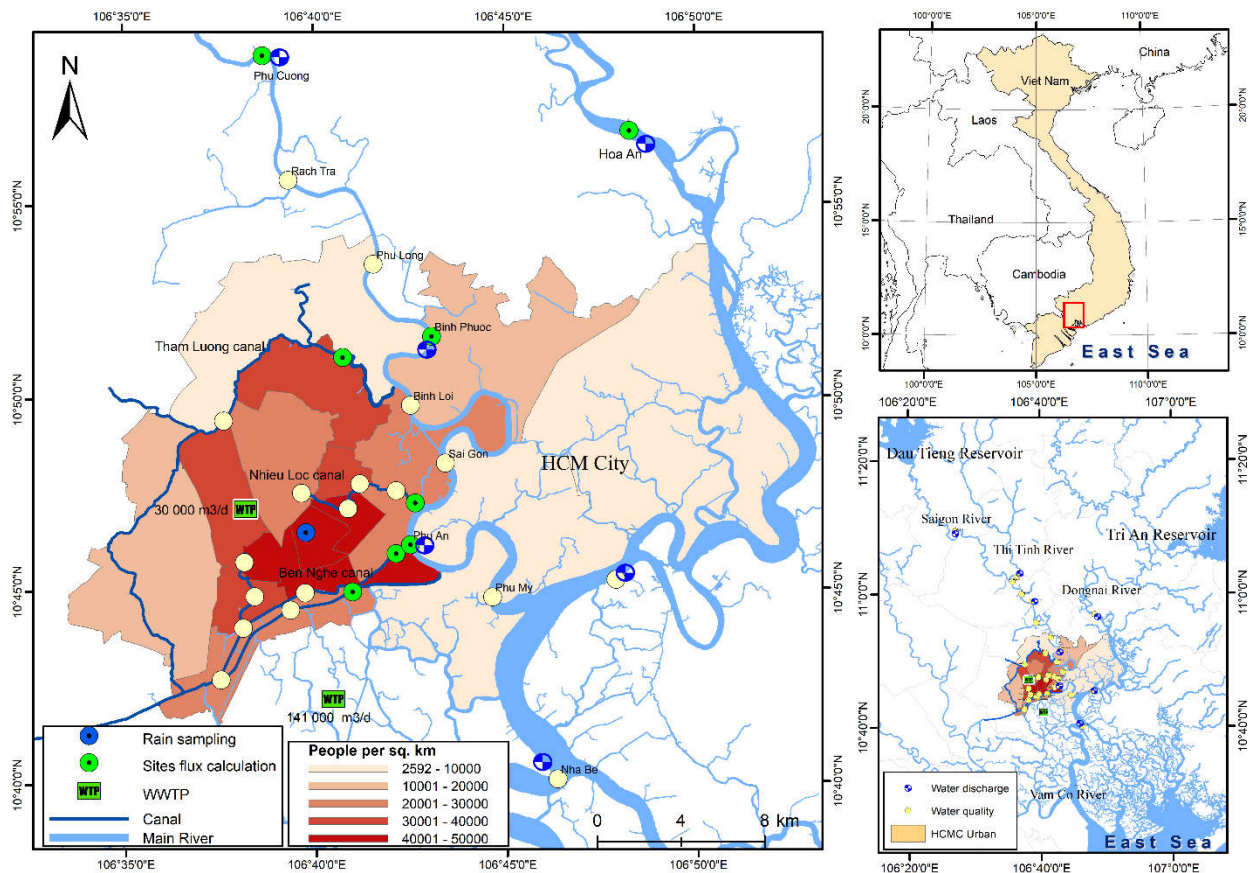


Figure 5-1. The Saigon – Dongnai rivers basin and population distribution in HCMC and location of the sampling sites

Land use is very diverse from the North to the South of HCMC: for example, agricultural activities (e.g. paddy rice) dominated in the north, while the heart of the city is dominated by urban

settlement. Can Gio Mangrove, which is located at South-south east of the city center, is a forest dominated area that has been recognized as biosphere reserve by UNESCO.

The population in HCMC was 8.4 million inhabitants in 2016, making it the highest population-concentrated city in Vietnam. The population density in different districts varied significantly from 2000 inhabitants per km² in rural districts up to 50 000 inhabitants per km² in urban districts (Figure 5-1). Nowadays, less than 10% of domestic waste water is collected and treated before being released directly to urban canals and rivers (Marcotullio, 2007). Three main urban canals (Tham Luong canal in north, Nhieu Loc canal in center and Ben Nghe canal in south of HCMC) receive untreated wastewaters (Figure 5-1). The main wastewater treatment plant (Binh Hung WWTP) is located in the south of the urban center (Figure 5-1).

2.2. Database from the Vietnamese water quality survey

Water discharges and water quality parameters data were provided by the Center of Monitoring (CEM) of the Department Of Natural Resources and Environment of HCMC (DONRE). This national monitoring program allows measuring monthly discharges and bi-monthly water quality parameters at different locations within the Saigon – Dongnai River basin and urban canal network (Figure 5-1). Since 2005, CEM is in charge of 26 sites for water quality monitoring, every two weeks and 15 sites for river discharge measurements every month in the Saigon – Dongnai River basin. In urban sectors, 16 canals are also monitored for water quality. This low elevation coastal zone is under the direct influence of asymmetric and semi-diurnal tides. Their hydrodynamic effect propagates up to the Dau Tieng dam. As a consequence, the estimation of the residual river discharges requires to measure hourly discharge data over a minimum period of 24 hours (two tidal cycles). The mean rivers discharge is then deduced from the integrative flux between the tide inflow and outflow. The residual discharge corresponds to the net positive flow of the river from land to the sea. Even with the low frequency measurement of the discharge (i.e. once per month), the hydrological seasonality is observed (Nguyen et al., 2019). Concerning water quality parameters, river and urban canals water samples were manually collected twice per month at the reference sites, filtered using pre-weight filters and analyzed according to the Vietnamese water quality standard (QCVN08, 2015). Water samples were collected during low and high water levels at 20-30 cm bellow water surface and at 4 m from the riverbank. Many parameters, e.g. temperature, pH, DO, Total Suspended Sediment (TSS), salinity, turbidity, NH₄⁺, PO₄³⁻, were

measured and selected for our study among others (COD, BOD₅, Pb, Cu, Cd, Hd, Mn, oil, Coliform and E.Coli). Study years were chosen based on the availability and reliability of data. The selected period 2012-2016 allows getting complete and suitable database of water discharge, TSS, NH₄⁺ and PO₄³⁻ in order to characterize the mean monthly distribution and fluxes calculation.

2.3 Additional monitoring program

To complete the CEM data base, we carried out bi-monthly monitoring at four sampling stations along the Saigon – Dongnai River system to analyze additional parameters (Total P, Total N, NO₃⁻, PO₄³⁻, NH₄⁺, DSi and TSS) in order to achieve the N and P fluxes. For each bi-monthly sample campaigns, surface water was sampled with a Niskin bottle in the middle of the river, either from a bridge or from a boat and stored in cool box at 4°C before to be analyzed in the laboratory.

TSS was measured by weighing of GF/F filters after 24 hours of drying at 50 °C in the Centre Asiatique de Recherche sur l'Eau (CARE) laboratory. All samples were filtered through Whatman GF/F filters (porosity 0.7µm) to analyze dissolved nutrients. Dissolved nutrients were analyzed on filtered water samples using standard colorimetric methods (APHA, 1995). The NO₃⁻ and DSi were analyzed by using respectively the cadmium reduction method and the silicomolybdate method. Unfiltered waters were used to measure Total N and Total P using persulfate digestion process and standard colorimetric method (American Public Health Association: APHA, 1995). Reproducibility for replicate measurements was better than 5 % for all total and dissolved nutrients.

2.4. Suspended sediment and nutrient budgets calculation

TSS and nutrients budget included nutrient export by rivers and tributaries, industrial inputs, domestic inputs and atmospheric depositions. We calculate fluxes of TSS, Total N, dissolved inorganic nitrogen (DIN including NO₃⁻ and NH₄⁺), Total P and dissolved inorganic phosphorus (DIP was PO₄³⁻).

2.4.1. Nutrients export by rivers

Table 5-1. Mean annual discharges and concentrations (\pm SD) at the sampling sites within the Saigon and Dongnai River (period 2012-2016).

Sites	Ben Suc	Phu Cuong	Binh Phuoc	Bach Dang	Thi Tinh	Hoa An
pK (km)	151	106	88	64	118	53
Discharge ($\text{m}^3 \text{s}^{-1}$)	18 \pm 14	37 \pm 33	41 \pm 32	51 \pm 39	20 \pm 11	632 \pm 446
TSS (mg L^{-1})	54.8 \pm 18.7	62.0 \pm 21.9	102.2 \pm 46.3	186.3 \pm 127.1	70.8 \pm 35.1	57.8 \pm 25.7
Total N (mgN L^{-1})	-	1.9 \pm 0.5	-	3.4 \pm 0.9	-	1.6 \pm 0.3
N-NH ₄ ⁺ (mgN L^{-1})	0.09 \pm 0.07	0.25 \pm 0.21	0.45 \pm 0.52	0.42 \pm 0.55	0.27 \pm 0.25	0.12 \pm 0.12
N-NO ₃ ⁻ (mgN L^{-1})	-	0.63 \pm 0.29	-	0.81 \pm 0.38	-	0.14 \pm 0.10
Total P (mgP L^{-1})	-	0.13 \pm 0.06	-	0.29 \pm 0.14	-	0.08 \pm 0.04
P-PO ₄ ³⁻ (mgP L^{-1})	0.08 \pm 0.04	0.06 \pm 0.03	0.08 \pm 0.04	0.07 \pm 0.08	0.09 \pm 0.05	0.04 \pm 0.04

We used data of nutrient and discharge provided by CEM-DONRE to calculate nutrient flux in the Saigon – Dongnai River system. Among the whole monitoring, we chose six sampling sites (see Figure 5-1 for location), in which, four sampling sites (Ben Suc, Phu Cuong, Binh Phuoc, Phu An)

are on the main course of the Saigon River. One sampling site (Hoa An) is used as the reference for Dongnai River inputs and the last site (Thi Tinh) is used as an indicator of input from river tributaries with agricultural and industrial activities. We have built a dataset of monthly mean discharge, TSS and nutrient (NH_4^+ and PO_4^{3-}) concentration and calculated fluxes of TSS, NH_4^+ and PO_4^{3-} . Due to lack of some data in the period of monitoring considered, we used data from our additional monitoring and data from the study of Strady et al. (2017) for input of Total N, NO_3^- and Total P (Nguyen et al., 2019) (see Table 5-1). Based on the method proposed by Walling and Webb (1985), the fluxes of TSS and nutrients have been calculated as follow:

$$Flux_{annual} = \sum_{i=1}^n Flux_{monthly}, \text{ with } Flux_{monthly} = \frac{C_{monthly} \times Q_{monthly} \times a \times 24 \times 3600}{10^6} \quad (1)$$

Where:

- $Flux_{annual}$ is the annual flux of TSS, Total N, NO_3^- , NH_4^+ , Total P or PO_4^{3-} (ton yr^{-1}).
- $Flux_{monthly}$ is the monthly flux of TSS, Total N, NO_3^- , NH_4^+ , Total P or PO_4^{3-} (ton month^{-1}).
- $C_{monthly}$ is the mean monthly concentrations (mg L^{-1}).
- $Q_{monthly}$ is the mean monthly discharge for the recorded period ($\text{m}^3 \text{ s}^{-1}$).
- a is number of day per month ($a = 28$ to 31) and n is the number of month ($n=12$).

2.4.2. Domestic inputs

2.4.2.1. Gross nutrients inputs from urban areas

To account for Total N and Total P fluxes from human excreta, we assumed the per capita human N emission ranges from 6 to 12 $\text{gN capita}^{-1} \text{ day}^{-1}$ and from 0.8 to 4.0 $\text{gP capita}^{-1} \text{ day}^{-1}$ for P emission (Triet et al., 2008). These values are in good agreement with the values proposed by Sung (1995) for Vietnamese (10.08 $\text{gN capita}^{-1} \text{ day}^{-1}$ for N and 1.7 $\text{gP capita}^{-1} \text{ day}^{-1}$ for P) and Meybeck et al. (1989) for global scale (9.04 $\text{gN capita}^{-1} \text{ day}^{-1}$ for N and 1.1 $\text{gP capita}^{-1} \text{ day}^{-1}$ for P). Gross nutrients inputs were calculated with the per capita human nutrients emission and population of HCMC (equation 2).

$$Flux (\text{ton yr}^{-1}) = \frac{E \times Inhab \times 365}{100000} \quad (2)$$

Where:

- Flux is the annual flux of Total N and Total P (ton yr^{-1}).

- E is Total N and Total P in pollutant emission (g capita⁻¹ day⁻¹).
- Inhab is total population of HCMC (capita).

2.4.2.2. Net nutrients inputs from WWTPs

To assess urban inputs, we recognized that untreated and treated domestic wastewaters from household and Wastewater Treatment Plants (WWTPs) were two main sources of TSS and nutrients. In HCMC, only about 8 % of the population has a direct connection to sewer network while 92 % of population released directly wastewater to urban canals (Table 5-2). The two wastewater treatment plants (Binh Hung Hoa and Binh Hung WWTPs) are located in the south of the high population density urban area (Figure 5-1). Their treatment capacities are 141 000 m³ day⁻¹ (426 000 inhabitants) for Binh Hung WWTP and 30 000 m³ day⁻¹ (120 000 inhabitants) for Binh Hung Hoa WWTP. WWTP inputs were calculated with WWTPs discharge capacity and mean output concentrations after treatment (see Table 5-2 and equation 3).

$$\text{Flux (ton yr}^{-1}\text{)} = \frac{C \times Q \times 3600 \times 24 \times 365}{10^6} \quad (3)$$

Where:

- Flux is the annual flux of TSS, Total N, NO₃⁻, NH₄⁺, Total P or PO₄³⁻ (ton yr⁻¹).
- C is the mean annual concentrations (mg L⁻¹).
- Q is the mean annual discharge for the recorded period (m³ s⁻¹).

2.4.2.3. Net flux from urban canals to the rivers

Inputs were estimated for the three main urban canals (Tham Luong, Nhieu Loc and Ben Nghe canal), which receive untreated domestic wastewater (Figure 5-1). Influent loading was estimated using the annual domestic water discharges and the mean annual TSS and nutrient concentrations within the canals, which were calculated from the data supported by CEM-DONRE (period of 2012-2016). The untreated wastewater discharge was estimated from the population living near these three canals (HCMC Statistical Yearbook 2016) and the per capita water consumption of ~150 L inhab⁻¹ day⁻¹ proposed by Triet et al. (2008). Annual flux was the product of annual discharge and mean inter-annual mean concentration at the low tide in urban canals (see Table 5-2 and equation 2).

Table 5-2. Mean annual discharges and concentrations (\pm SD) from 2012 to 2016 at the outlet of the three main urban canals and in WWTPs

Canals	Tham Luong	Nhieu Loc	Ben Nghe	WWTPs
Population (inhabitant)	2 494 983	769 103	2 409 606	546 000
Discharge ($\text{m}^3 \text{s}^{-1}$)	4.33	1.34	4.18	1.98 ^a
TSS (mg L^{-1})	59 \pm 31	56 \pm 23	56 \pm 34	10 \pm 10 ^a
N-NH ₄ ⁺ (mgN L^{-1})	11 \pm 8	3.0 \pm 1.5	3.0 \pm 2.6	9.2 ^b
P-PO ₄ ³⁻ (mgP L^{-1})	1.0 \pm 0.4	0.15 \pm 0.07	0.15 \pm 0.12	-
Total N (mgN L^{-1})	-	-	-	12.9 ^b
Total P (mgP L^{-1})	1.11 ^c	0.18 ^c	0.19 ^c	0.98 ^b

^a Data from technical visit in Binh Hung WWTP 29/05/2017.

^b Data from Center for Environmental Technology and Management in 8/2014 and from Technical report of WWTPs in 3/2015.

^c Data from Strady et al. (2017).

2.4.3. Industrial inputs

Several large industrial sites exist in the Saigon – Dongnai rivers basin with different activities (Figure 5-2). The contribution of each industrial activity to TSS, N and, P loading to the hydrosystem is extremely difficult to evaluate. To estimate the industrial inputs, we gathered information on representative enterprises within HCMC concerning their production, discharge effluents and water quality variables (e.g. TSS and nutrients). Our approach involved (i) the selection of the most significant enterprises of activity in terms of organic and nutrient pollution (like paper industry, chemical/fertilizers industry, food process, textile industry, pharmaceutical chemistry and cosmetic, detergents, care product) and (ii) the elaboration of a questionnaire sent to all chosen companies. The questionnaire was constructed and sent to about 183 companies in 13 industrial zones in HCMC, with the request to give information about (i) wastewater effluent, number of workers and production, (ii) quality of wastewater discharge (TSS and nutrient concentrations) and (iii) how effluent were discharged (directly to the river, into canal or stored in basin). We received 41 answers, in which there were only 11 answers usable for our calculations.

Table 5-3. Mean daily discharges and concentrations of six sectors in industrial zones (data synthesis from questionnaires)

Industrial sector	Discharge (m ³ day ⁻¹)	TSS (mg L ⁻¹)	Total N (mgN L ⁻¹)	Total P (mgP L ⁻¹)
Paper industry	25	65	24	5
Chemical industry	42	75	30	2
Food industry	87	40.7	32.29	3.975
Pharmaceutical chemistry	25	5	7.5	3.5
Textile industry	-	-	-	-
Cosmetics, detergents, care products	-	-	-	-

We assumed that industrial inputs can be estimated by two approaches: (i) based on data from questionnaires to calculate nutrient input from a group of paper industry, chemical industry, food

industry and pharmaceutical chemistry sectors (1st group) and (ii) based on HCMC Export Processing and Industrial Zones Authority (HEPZA) report data and on the Vietnamese industrial water quality regulation (QCVN40, 2011) to calculate input from a group of textile industry and cosmetics, detergents, care products sectors (2nd group).

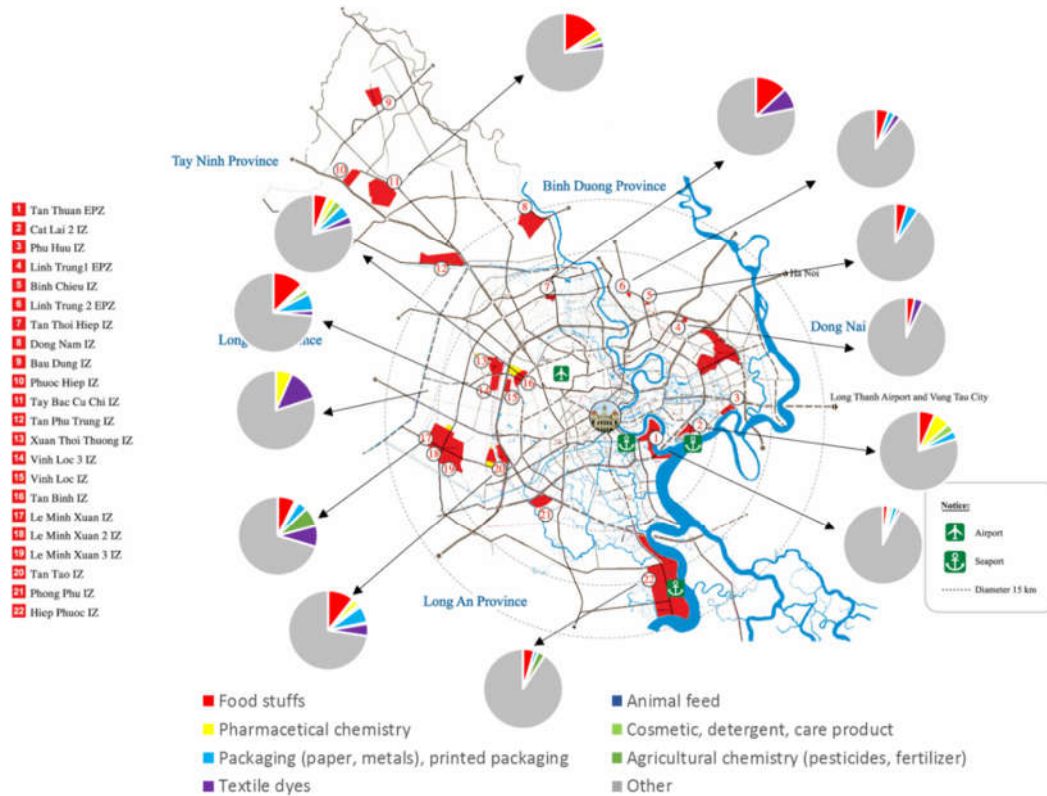


Figure 5-2. Location of industrial sites and proportion of target sector industries (source: Publication Du Catalogue Des Livrets Du PADDI: www.paddi.vn)

For the 1st group, annual nutrient fluxes was the product of mean wastewater discharge and mean TSS and nutrient concentration for each sector (see Table 5-3 and equation 3). For the 2nd group, we hypothesized that daily discharges can be estimated based on the discharge of each industrial zone and percent of textile industry or cosmetics, detergents, care products enterprise (see Figure 5-2 and equation 4). TSS and nutrient fluxes were calculated from daily discharges and concentrations based on equation 3, in which, TSS and nutrient concentrations were extracted from QCVN40 (2011) (see Table 5-4).

$$Q = Q_{IZ} \times \frac{N_{sector}}{N_{IZ}} \quad (4)$$

Where:

- Q and Q_{IZ} are the daily discharge of enterprise and industrial zone (m³ day⁻¹).
- N_{sector} and N_{IZ} are the numbers of textile industries or cosmetics, detergents, care products enterprises and total number of enterprises in each industrial zone.

Table 5-4. Levels of TSS and nutrient concentrations released by domestic and industrial wastewaters based on National Technical Regulation on Domestic and Industrial Wastewater

	For domestic wastewater QCVN14 (2008)	For industrial wastewater QCVN40 (2011)
TSS (mg L ⁻¹)	50	50
Total N (mgN L ⁻¹)	-	15
NO ₃ ⁻ (mgN L ⁻¹)	30	-
NH ₄ ⁺ (mgN L ⁻¹)	5	5
Total P (mgP L ⁻¹)	-	4
PO ₄ ³⁻ (mgP L ⁻¹)	6	-

2.4.4. Atmospheric depositions

Atmospheric depositions were evaluated through the collection of rain water during the rainy season 2017 at the roof of the CARE laboratory in HCMC Center (Figure 5-1). This monitoring was started from May 2017 to December 2017 (Rain at this period represents more than 90 % of yearly precipitations). The volume of rain water was collected using Owen-type gauge (AFNOR NF X 43-014) after each rain event. The samples from Monday to Sunday were combined together to have one composite sample per week. These samples were filtered to analyzed TSS and nutrients (total and dissolved N and P). Rain intensity and concentrations of TSS, Total N, NO₃⁻, NH₄⁺, Total P or PO₄³⁻ were measured. We considered that the impervious surface of HCMC was 27 142

hectares in 2016 (Nguyen et al., 2017). Annual flux is calculated as the product of weekly cumulative rain and weekly concentrations (see Table 5-5 and equation 4).

$$Flux_{annual} = \frac{27142 \times \sum_{i=1}^n Flux_{weekly}}{1000}, \text{ with } Flux_{weekly} = \frac{C_{weekly} \times Q_{weekly} \times 10^4}{10^6} \quad (4)$$

Where:

- Flux_{annual} is the annual flux of TSS, Total N, NO₃⁻, NH₄⁺, Total P or PO₄³⁻ (ton yr⁻¹).
- Flux_{weekly} is the specific flux of TSS, Total N, NO₃⁻, NH₄⁺, Total P or PO₄³⁻ (kg ha⁻¹).
- C_{weekly} is the weekly concentrations (mg L⁻¹).
- Q_{weekly} is the weekly cumulative rain for the recorded period (mm).
- n = 35 and 27142 (ha) is impervious surface of HCMC in 2016 (Nguyen et al., 2017).

Table 5-5. Mean annual concentrations and specific flux of TSS and nutrients from atmospheric deposition in 2017

	Concentration (mg L ⁻¹ ±SD)	Specific flux (kg ha ⁻¹) *
TSS	7.94±8.43	4.189
N-NH ₄ ⁺	0.21±0.16	0.131
N-NO ₃ ⁻	0.30±0.13	0.244
Total N	1.66±0.71	1.168
P-PO ₄ ³⁻	0.01±0.004	0.007
Total P	0.03±0.015	0.018

* Cumulative rain in 2017 of 1969 mm (from May to December).

2.4.5. Assumption on TSS and nutrients budgets

Fluxes of TSS and nutrients within the Saigon – Dongnai Rivers system were calculated based on industrial input, domestic inputs (treated and untreated nutrient input) and atmospheric deposition, which were described above. The estimations relied on a wide variety of sources of different reliability, as well as on several hypotheses. Values deduced from questionnaires may be inaccurate because all answers from enterprises were not always correctly registered. Moreover,

due to the lack of data in the study area, many fluxes, such as fluxes of Total N and Total P in domestic inputs, were estimated from data reported in the literature.

A last important data to establish our budget is the flux of NO_3^- in WWTPs outlet and industrial discharge, and Total N and Total P of canal discharge. The estimation of these fluxes was based on two assumptions. Generally, nitrogen exists in organic and inorganic forms; Total N concentration is the sum of dissolved inorganic nitrogen (NO_3^- and NH_4^+), dissolved organic nitrogen (DON), particulate organic nitrogen (PON) and particulate inorganic nitrogen. However, according to QCVN14 (2008) and QCVN40 (2011), level of organic matter or organic nitrogen in domestic and industrial wastewater after treatment is closed to 0.003 mg L^{-1} . We can ignore organic nitrogen pollutants in treated domestic and industrial wastewater. We assumed then that nitrogen in wastewater discharge only comprises NO_3^- and NH_4^+ forms. The different of Total N and NH_4^+ input is NO_3^- input from WWTPs and industrial discharge into river system.

2.4.6. Expected nutrients inputs in 2025, 2040 and 2050

To estimate nutrients fluxes in the future, we assume that HCMC population increases by 3% per year (according to ADB 2010 and HCMC Statistical Yearbook 2016) and water consumption of inhabitant was about $200 \text{ L inhab}^{-1} \text{ day}^{-1}$ in 2025 (Nguyen et al., 2011). According to HCMC authority, the ambitious environmental sanitation projects aims at building a drainage basin inside the city and establishing ten new wastewater treatment plants (Tran Ngoc et al., 2016). All the current sewage treatment plants in HCMC use the conventional activated sludge without specific biological or chemical nutrient removal (QCVN 14, 2008). In that case, the average Total N and Total P removal are about 40-50% and 10-15%, respectively (Metcalf and Eddy/AECOM, 2014). Gross nutrients inputs were calculated based on the per capita human nutrients emission and population of HCMC (equation 2). Net nutrients inputs by WWTPs were calculated based on the expected population connected to WWTPs, the per capita human nutrients emission and nutrients removal efficiency (equation 5 and Table 5-6).

$$\text{Flux (ton yr}^{-1}\text{)} = \frac{E \times \text{Inhab}_{\text{WWTP}} \times 365 \times C}{100000} \quad (5)$$

Where:

- Flux is the annual flux of Total N and Total P (ton yr^{-1}).

- E is Total N and Total P in domestic emission ($\text{g capita}^{-1} \text{ day}^{-1}$).
- $\text{Inhab}_{\text{WWTP}}$ is the population of HCMC which is connected to WWTPs (capita).
- C is the nutrients removal efficiency (%).

Table 5-6. Treatment capacity and nutrients removal efficiency of WWTPs from past to future

Year	2012-2016	2025	2040	2050
Population (inhabitant) ^a	8 441 902	11 014 767	17 160 649	23 062 447
WWTPs number ^b	2	4	12	12
Treatment capacity ($\text{m}^3 \text{ day}^{-1}$) ^b	171 000	1 253 000	2 813 000	2 813 000
% population connected to sewage treatment	10 %	57 %	82 %	61 %
Nutrients removal efficiency ^c				
<i>Nitrogen</i>	40-50 %	40-50 %	40-50 %	40-50 %
<i>Phosphorus</i>	10-15 %	10-15 %	10-15 %	10-15 %

^a Population growth of 3% per year from ADB 2010, HCMC Statistical Yearbook 2016

^b Planification of the building of new WWTPs from Tran Ngoc et al., 2016

^c Nutrients removal efficiency in conventional active sludge treatment process from Metcalf and Eddy/AECOM, 2014

3. Results

3.1. Seasonal variation of TSS and nutrients concentrations

The spatial variation of TSS and nutrients, observed during dry and wet season of the period 2012-2016 are reported in Figure 5-3a. Values were averaged at each site and for both season and are shown with error bars. Panels all showed a similar trend. The temporal variability is very high especially in the heart of HCMC. The spatial variation of TSS concentration clearly showed sectors of increase, also in the heart of HCMC. The highest (averaged) concentration measured was about 215 mg L^{-1} , for TSS at the city center of HCMC. Nutrient concentrations, given by NH_4^+ and PO_4^{3-} concentrations, were varying slightly from upstream to downstream along the Saigon River with

no difference between dry and wet season in 2017. The high value of NH_4^+ ($0.48 \pm 0.67 \text{ mgN L}^{-1}$) showed little variations along the Saigon River, although smaller values were observed in the downstream part of HCMC. PO_4^{3-} concentration fluctuated slightly from upstream to downstream during dry season and wet season.

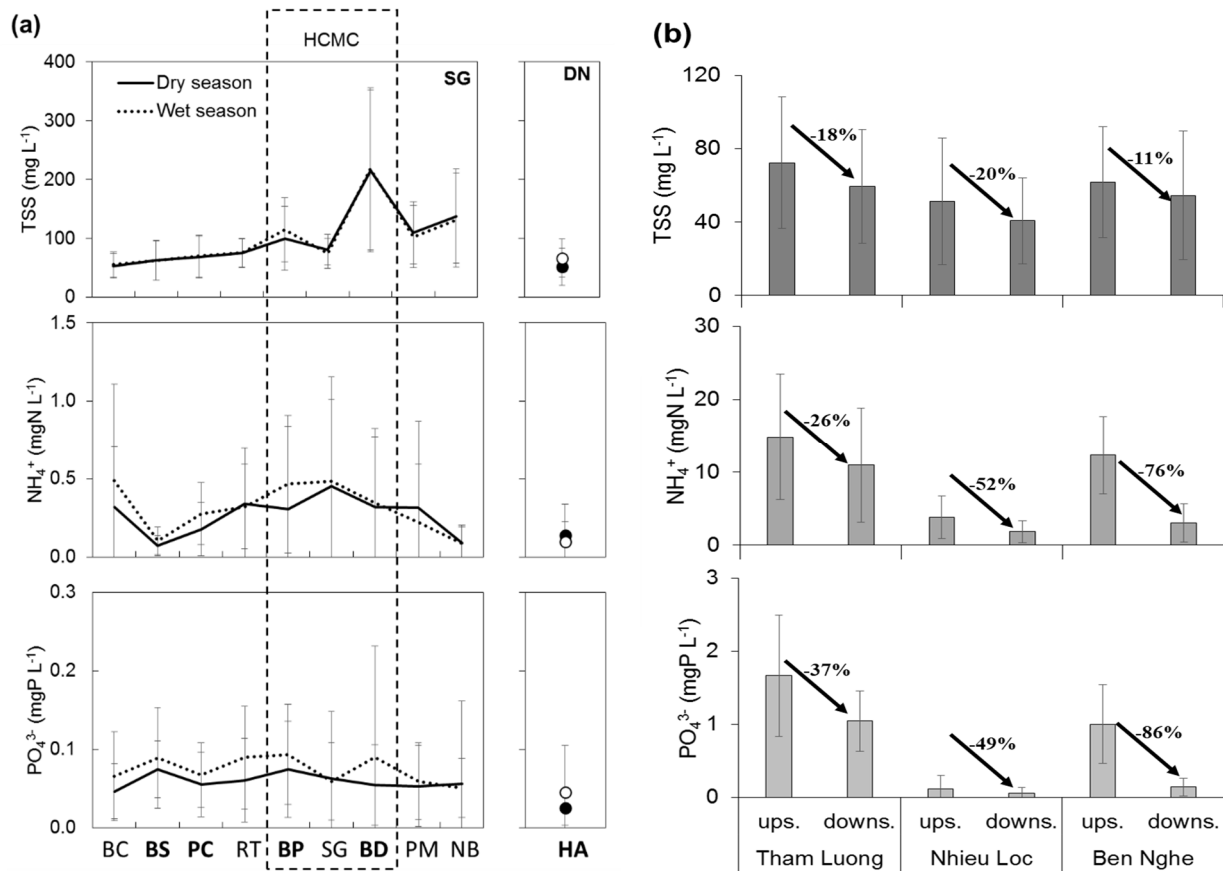


Figure 5-3. Mean TSS, NH_4^+ and PO_4^{3-} concentrations over the period 2012-2016: (a) along the Saigon River (SG) and the Dongnai River (DN) during the wet and dry seasons and (b) in urban canals of HCMC. In which, BC, BS, PC, RT, BP, SG, BD, PM, NB and HA are 10 sampling sites within the Saigon – Dongnai River basin (Ben Cui, Ben Suc, Phu Cuong, Rach Tra, Saigon, Bach Dang, Phu My, Nha Be and Hoa An respectively). In Figure 5-3a, bold legends represent sampling sites which are chosen for budget calculation.

High concentrations of nutrients were found in urban canals with value ten to twenty times higher than the ones measured in the Saigon and Dongnai Rivers for PO_4^{3-} and NH_4^+ , respectively (Figure 3b, versus Figure 5-3a). This undoubtedly indicates domestic untreated wastewaters inputs. With such high concentrations, Tham Luong canal and Ben Nghe canal can be considered as a

wastewater collector draining HCMC. Both of these two canals (Tham Luong and Ben Nghe) represented a significant source of pollution to the Saigon River, (see Figure 5-1 for location). A decrease in TSS and nutrients (NH_4^+ and PO_4^{3-}) concentration was observed from upstream to downstream of each canal, which indicated that urban canals are efficient to retain a proportion of the domestic pollution emitted (Figure 5-3b).

3.2. Seasonal and inter-annual variations of river fluxes and the effect of climatic conditions on the river fluxes

3.2.1. Seasonal variation of river fluxes

Results presented in Figure 5-4 shows the seasonal fluctuations of TSS and nutrient fluxes. One can distinguish two distinct hydrologic phases for both the Saigon and Dongnai Rivers system, with a wet season that lasted 7 months, from May to November, and a dry season characterized by low-flow condition from December to April (5 months). Seasonality is much more pronounced for the Dongnai River than for the Saigon River.

While water discharge in the Saigon River ranged from $4.18 \text{ m}^3 \text{ s}^{-1}$ to $222 \text{ m}^3 \text{ s}^{-1}$, in the Dongnai River this values fluctuated from $85.4 \text{ m}^3 \text{ s}^{-1}$ to $2524 \text{ m}^3 \text{ s}^{-1}$ (Figure 5-4a). The mean discharge is thus twelve times lower in the Saigon River ($50 \text{ m}^3 \text{ s}^{-1}$) than in the Dongnai River ($632 \text{ m}^3 \text{ s}^{-1}$).

During the monitoring period considered (2012-2016), TSS flux in the Saigon River ranged from $1.79 \times 10^3 \text{ tonSS month}^{-1}$ to $346 \times 10^3 \text{ tonSS month}^{-1}$; in the Dongnai River this value varied from $4.36 \times 10^3 \text{ tonSS month}^{-1}$ to $545 \times 10^3 \text{ tonSS month}^{-1}$ (Figure 5-4b). The maximum flux of TSS also extends from May to November and is strongly driven by the monthly water discharge for both rivers. The peak of TSS flux accounted for 77 % of the annual TSS flux in the Saigon River and 80 % of the annual TSS flux in the Dongnai River during this period.

The high values of Total N, NO_3^- and NH_4^+ fluxes extends during the period of May to November for the two rivers (Figure 5-4c to 5-4e). During the wet season, the Saigon River delivered 3.4 tonN of Total N, 0.7 tonN of NO_3^- and 0.8 tonN of NH_4^+ , which was 68 %, 66 % and 82 % of total annual fluxes, respectively. They were six, three and two times lower than fluxes of Total N, NO_3^- and NH_4^+ exported by the Dongnai River. Figure 5-4f and 5-4g showed the monthly fluxes of Total P and PO_4^{3-} in the two rivers. Most of the annual flux is exported during the wet season, i.e. 67 %

and 83 % of total annual Total P and PO_4^{3-} fluxes for the Saigon River and 71 % and 81 % for the Dongnai River, respectively.

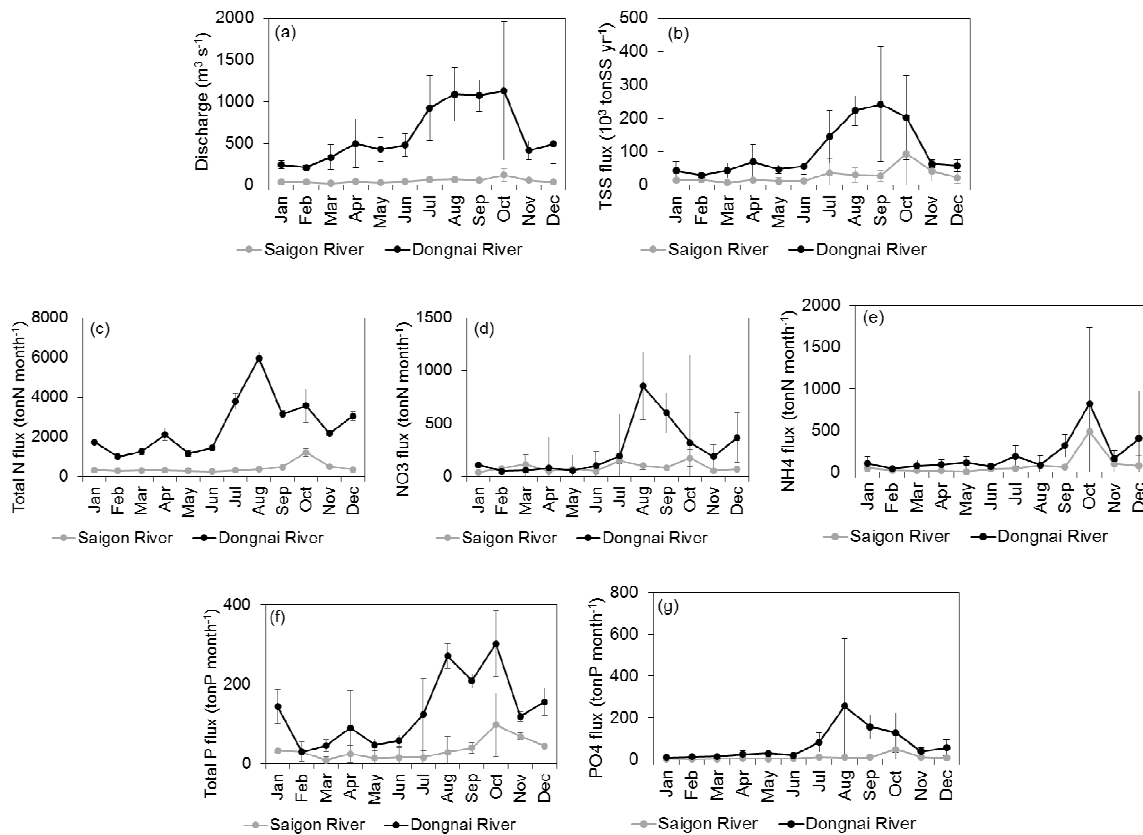


Figure 5-4. Suspended sediment and nutrient fluxes from 2012 to 2016 in the Saigon (Bach Dang station) and Dongnai (Hoa An station) Rivers system.

3.2.2. Inter-annual variations of river fluxes

Our estimations of interannual TSS and nutrient fluxes within the Saigon and Dongnai River basin during the 2012-2016 time period are presented in Figure 5-5. As there is a high variability between sites, data are represented on a log scale for water discharge and TSS flux. Whatever the year considered (gray bars for all stations), the difference between Saigon and Dongnai tributaries predominates on interannual variations.

During the observed period, the mean fluxes of TSS, NH_4^+ and PO_4^{3-} in the Dongnai River were five, six and three times higher than the ones in the Saigon River, respectively. The lowest fluxes of TSS ($19 \times 10^3 \text{ tonSS yr}^{-1}$), NH_4^+ (23 tonN yr^{-1}) and PO_4^{3-} (32 tonP yr^{-1}) were observed in Ben Suc in 2012. The maximal yearly fluxes were $1.43 \times 10^6 \text{ tonSS yr}^{-1}$ (2013) for TSS, $4.65 \times 10^3 \text{ tonN}$

yr⁻¹ (2016) for NH₄⁺ and 1.44×10³ tonP yr⁻¹ (2016) for PO₄³⁻. This is not surprising, as the most of the sampling sites had their lowest fluxes in 2012, when the minimal water discharge was recorded. TSS and NH₄⁺ fluxes increased from 2012 to 2016 from upstream to downstream along the Saigon River while PO₄³⁻ flux decreased from 2012 to 2016 (Figure 5-5). On an interannual fluxes basis, the Saigon River transported about 60 % DIN and 20 % DIP while the Dongnai River transported only 24 % DIN and 63 % DIP.

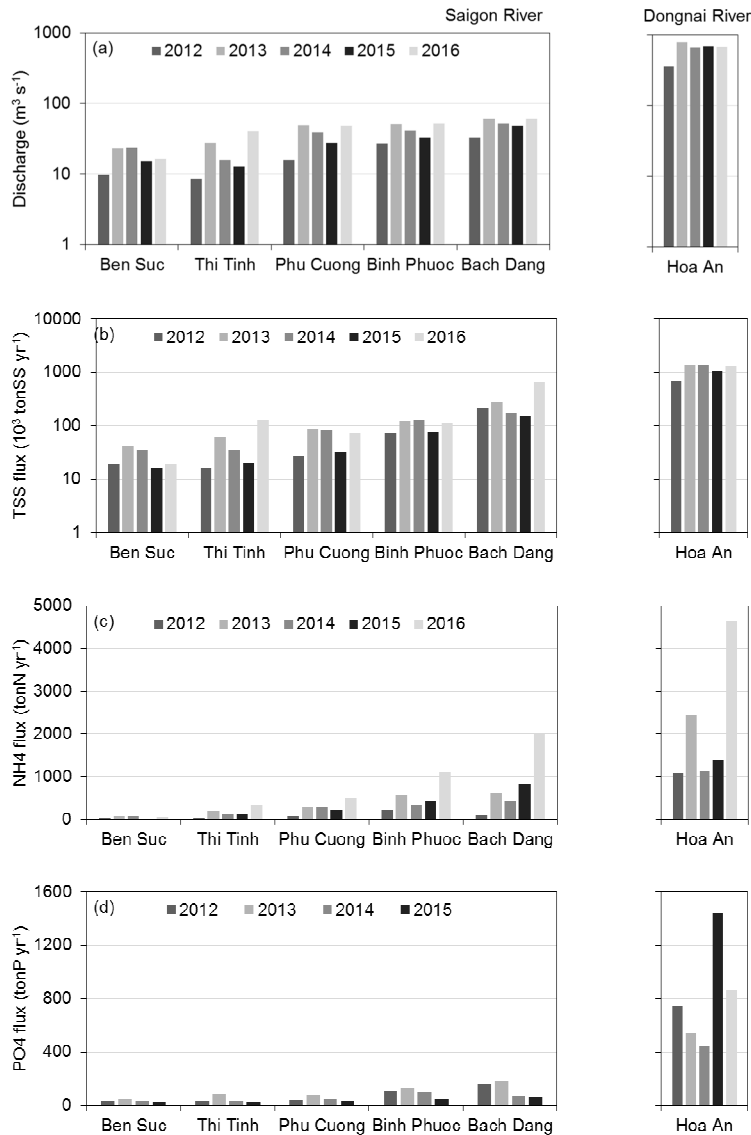


Figure 5-5. Interannual discharges and fluxes of TSS, NH₄⁺ and PO₄³⁻ in the Saigon – Dongnai Rivers system from 2012 to 2016.

3.3. Mean TSS and nutrient budgets in the Saigon River

Water budget.

Discharge contribution is calculated independently for each site (and tributaries) and appears balanced (Figure 5-6). Urban waters contribute to 20 % of the Saigon River discharge in mean over the period 2012-2016 indicating a potential impact of untreated waste waters. Knowing that the Saigon river is around 200m wide and 10m deep in the city, and considering that the water discharge can frequently oscillates between $-1200 \text{ m}^3 \text{ s}^{-1}$ and $+1500 \text{ m}^3 \text{ s}^{-1}$ with tides, the mean residual water discharge of the Saigon River is very small ($51 \text{ m}^3 \text{ s}^{-1}$).

TSS budget.

Concerning suspended sediment, the mean annual TSS flux to the estuary by the Dongnai River was around $1.18 \times 10^6 \text{ tonSS yr}^{-1}$, about four times higher than the values exported by the Saigon River. Through the crossing of HCMC, TSS flux increased by over tenfold to reach $290 \times 10^3 \text{ tonSS yr}^{-1}$ in comparison to watershed upstream value ($26 \times 10^3 \text{ tonSS yr}^{-1}$) (Figure 5-7).

Nitrogen budget.

Regarding N fluxes, the Saigon River delivered $5323 \pm 378 \text{ tonN yr}^{-1}$, $1928 \pm 59 \text{ tonN yr}^{-1}$ and $791 \pm 727 \text{ tonN yr}^{-1}$ for Total N, NO_3^- and NH_4^+ fluxes during period of 2012-2016, whilst the Dongnai River exported $31030 \text{ tonN yr}^{-1}$, $2963 \text{ tonN yr}^{-1}$ and $2132 \text{ tonN yr}^{-1}$, respectively (Figure 5-8).

Phosphorus budget.

Concerning P fluxes, the Saigon – Dongnai Rivers system delivered $2103 \text{ tonP yr}^{-1}$ and 917 tonP yr^{-1} for Total P and PO_4^{3-} fluxes, in which the Saigon River exported 21 % and 13 % of total annual flux of Total P and PO_4^{3-} , respectively (Figure 5-9).

Silica budget.

Concerning Silica fluxes, the Saigon – Dongnai Rivers system delivered $66500 \text{ tonSi yr}^{-1}$, in which the Dongnai River exported 94 % of total annual flux of DSi, respectively (Figure 5-10).

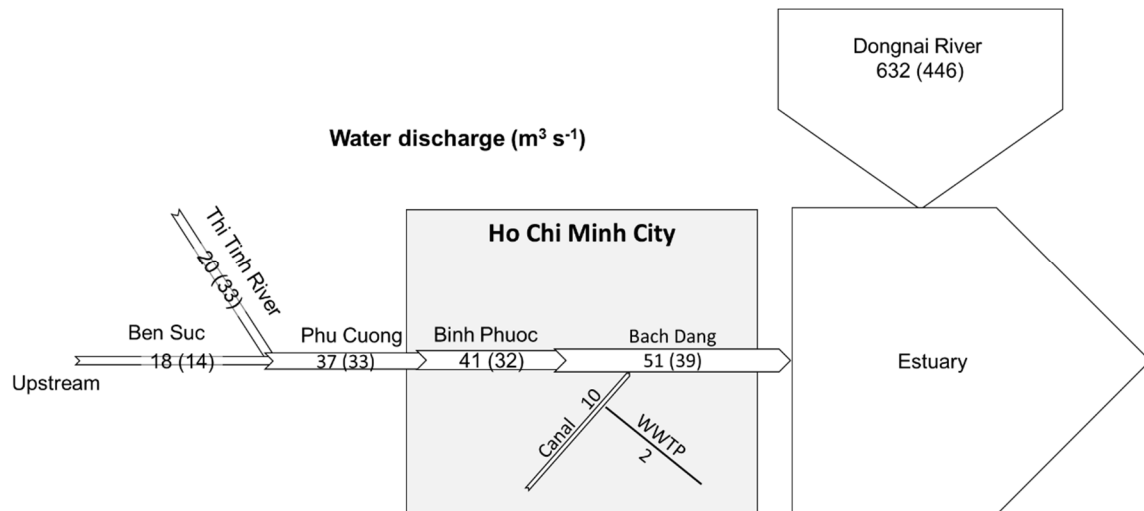


Figure 5-6. Mean Annual water discharge (\pm SD) from 2012 to 2016 within the Saigon and Dongnai River system.

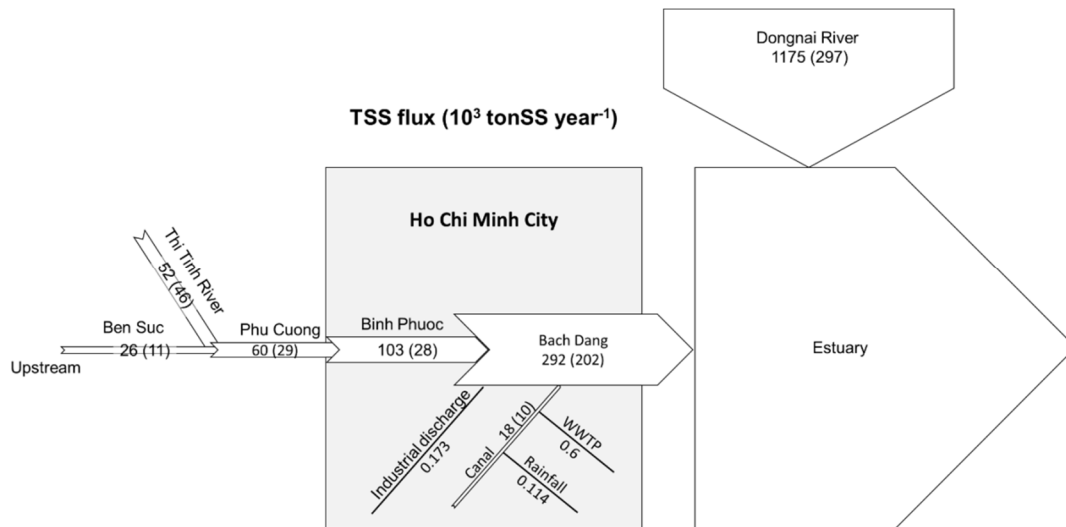


Figure 5-7. Annual suspended sediment (\pm SD) budget from 2012 to 2016 within the Saigon and Dongnai River system.

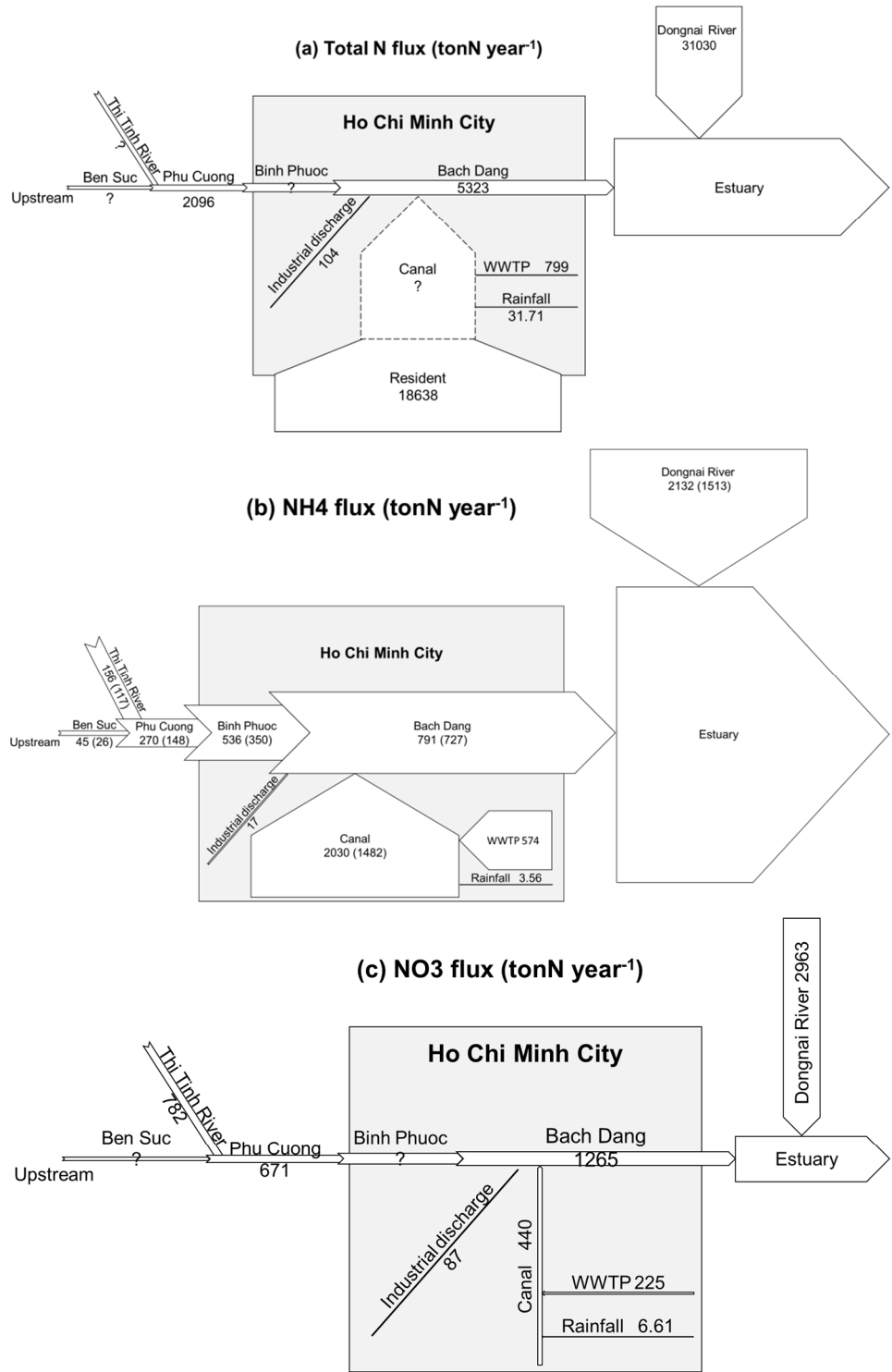


Figure 5-8. Annual nitrogen (\pm SD) budget from 2012 to 2016 within the Saigon and Dongnai River system.

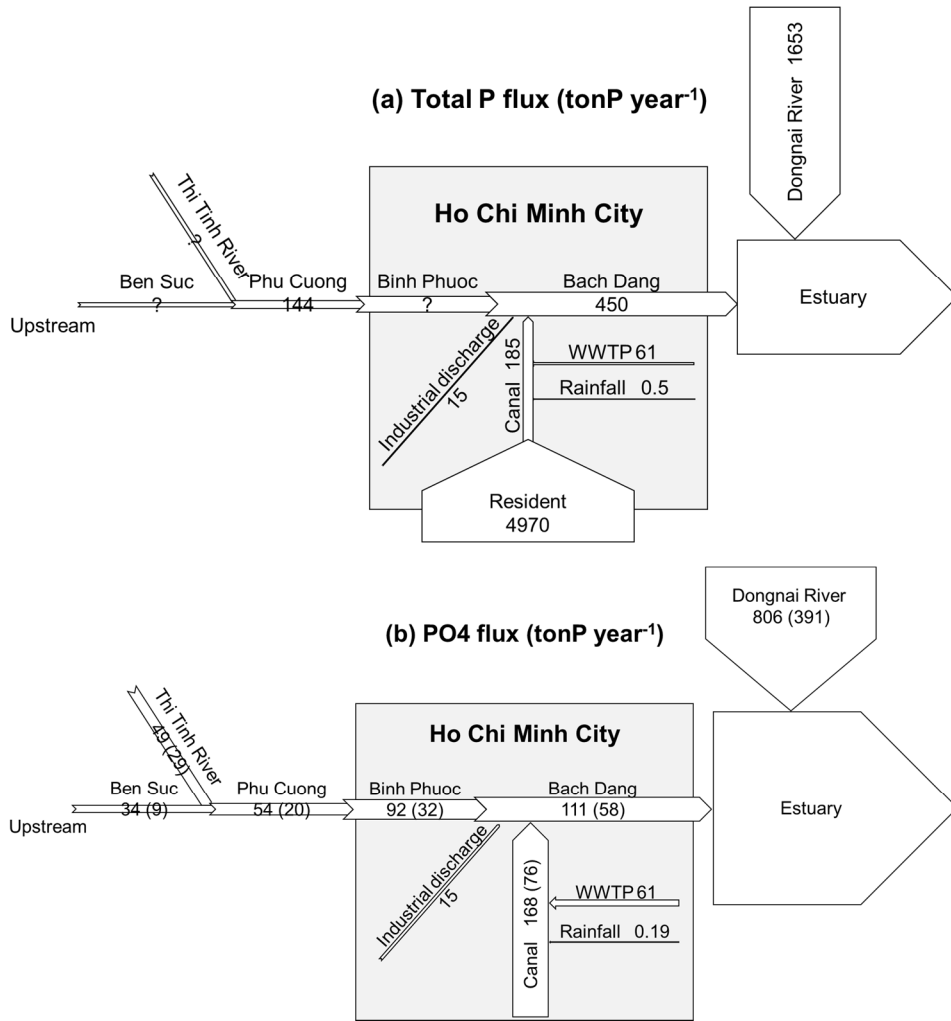


Figure 5-9. Annual phosphorus (\pm SD) budget from 2012 to 2016 within the Saigon and Dongnai River system.



Figure 5-10. Annual dissolved silica (\pm SD) budget from 2012 to 2016 within the Saigon and Dongnai River system.

3.4 Comparison between past and future nutrients inputs

Table 6 presents the estimated population in HCMC from 2016 to 2050. The population should increase by about 270% between 2016 and 2050. During period 2012-2016, the proportion of urban households connected to the main sewage treatment system is low (only 10%). It is estimated that around 57% of the city's dwellers will have access to sewage treatment systems in 2025; it will reach about 82% in 2040. For a stable water treatment capacity, the population in HCMC connected with sewage treatment systems will decrease to 61% in 2050, because of the increase of inhabitants. The higher population will also affect nutrient gross, which is expected to increase remarkably, from 28 tonN yr⁻¹ in 2016 to 76 tonN yr⁻¹ in 2050; and from 7 tonP yr⁻¹ in 2016 to 20 tonP yr⁻¹ in 2050. Besides, there is a decrease of net nutrients fluxes from urban canals to river from 2016 to 2040 prior to an increase one from 2040 to 2050 (see Table 5-7).

Table 5-7. Gross nutrients inputs and net nutrients inputs from

Year	2012-2016	2025	2040	2050
Gross nutrients inputs				
<i>Total N flux (tonN yr⁻¹)</i>	27 732	36 184	56 373	75 760
<i>Total P flux (tonP yr⁻¹)</i>	7 395	9 649	15 033	20 203
Net inputs nutrients from WWTPs				
<i>Total N flux (tonN yr⁻¹)</i>	799	9 055	20 330	20 330
<i>Total P flux (tonP yr⁻¹)</i>	61	4 601	4 529	9 053
Net nutrients fluxes from canals to rivers				
<i>Total N flux (tonN yr⁻¹)</i>	18 638	15 603	10 169	29 557
<i>Total P flux (tonP yr⁻¹)</i>	4 970	4 161	2 712	7 882

4. Discussions

4.1. Budgets in contemporary Ho Chi Minh City 2012-2016

Water discharge during the wet season contributes to 73 % of the annual discharge at the Saigon River and 73 % for the Dongnai River. Indeed, water discharge in the Saigon and Dongnai Rivers varied seasonally. Since the Dau Tieng and Tri An reservoirs were built in the Saigon and Dongnai Rivers, the discharge in each river has been well regulated for water demand and usage (Trieu et al., 2014). The discharge by Dau Tieng reservoir is above $30 \text{ m}^3 \text{ s}^{-1}$ during the dry season to prevent saline intrusion from the estuary with potential consequences on water production and agricultural activities (Trieu et al., 2014).

Based on the temporal changes of water discharge and TSS and nutrient concentrations during the period study, TSS and nutrients did not follow any flushing effect. At the contrary, TSS and nutrients increased with increasing water discharge, in almost similar proportion. In the Saigon and Dongnai Rivers, the seasonal variability of the hydrology played an important role in determining the TSS and nutrients export into the estuary. As discussed above, the highest TSS and nutrient fluxes occurred during the wet season and is very probably induced by the rainfall.

TSS budget.

Runoff during heavy rain, which leads to TSS urban inputs from canals cannot totally explain the global TSS increase. Also, industrial discharge, WWTPs and rainfall input remain negligible, as depicted in Figure 7. Thus, local bank erosion is likely to be responsible of the difference, as observed on site sediment trapping probably occurring in the Dau Tieng reservoirs could be compensated downstream by “hungry waters” and river bank erosion (Kondolf, 1997).

To compare our flux estimations with other river, it appeared that the specific TSS flux in the Saigon – Dongnai Rivers system for period of 2012-2016 was around $47.1 \text{ tonSS km}^{-2} \text{ yr}^{-1}$, and was higher than the values of Congo River (Coynel et al., 2005) and lower than those in Magdalena River (Restrepo and Kjerfve, 2000), Mekong River (Wolanski et al., 1996), Yangtze River (Liu et al., 2016) and Amazon River (Edmond et al., 1981) (see Table 5-8). The TSS specific flux in the Saigon – Dongnai Rivers appeared to be in the low range of those observed worldwide. It can be explained by the high potential sediment retention within the main reservoirs on the main course of the Saigon River and Dongnai River but also by the storage within this low elevation coastal

area as previously observed in the Red River basin in Vietnam for instance (Luu et al., 2010). Recent study has demonstrated high coastal erosion downstream of the Saigon River estuary attributed, which would most probably be the consequence of sediment trapping by dams (Marchesiello et al., 2019, submitted).

Table 5-8. Specific fluxes of TSS and nutrients in the Saigon – Dongnai Rivers system and some rivers in the world.

River	Area	Specific discharge	TSS specific flux	Specific TN flux	Specific TP flux	References
	km ⁻²	10 ³ m ³ km ⁻² yr ⁻¹	tonSS km ⁻² yr ⁻¹	kgN km ⁻² yr ⁻¹	kgP km ⁻² yr ⁻¹	
Saigon River	4 717	335	62	1 129	95	This study
Dongnai River	26 499	751	45	1 171	62	
Saigon – Dongnai Rivers	31 216	689	47	1 165	67	
Amazon River	6 144 000	1 025	187	224	50	Edmond et al., 1981
Changjiang River (China)	1 808 500	9 285	4 819	346	145	Zhang, 1996; Liu et al., 2016
Mekong River	790 000	595	215	238	24	Wolanski et al., 1996; Li and Bush, 2015
Red River basin	156 451	1 104	280	855	325	Le et al., 2005, 2007

Nitrogen budget.

The specific Total N flux in the Saigon – Dongnai Rivers system was $1164 \text{ kgN km}^{-2} \text{ yr}^{-1}$, a value higher than the Red River Delta (Le et al., 2005, 2007), Changjiang River, Minjiang River, Jiulongjiang River and Zhujiang River in China (Zhang, 1996; Liu et al, 2016), Amazon River (Edmond et al., 1981) and lower than those in Jiazhou River (Liu et al., 2005). This indicates the high impact of the megacity of HCMC on the total N flux to the coastal area.

It is clear, from Figure 8 that untreated domestic wastewaters were responsible of high level of nitrogen input, in link with the very high concentrations observed in urban canals (Strady et al., 2017). Untreated domestic wastewaters dominated the fluxes of NO_3^- and NH_4^+ , accounted for 77 % and 58 % of total urban fluxes, while industrial discharge and rainfall were negligible. The fluxes in urban canals are extremely high in respect with the fluxes from other inputs and they unbalanced the budgets of Total N and NH_4^+ . Indeed, the flux calculated at the outlet of the river was clearly smaller than the sum of the fluxes inputs. Our study also estimated Total N from human excreta. The results showed that resident lived near urban canals released $18638 \text{ tonN yr}^{-1}$. We hypothesize that a proportion of Total N was stored in urban canals or in individual septic tank that are common in old urban center (Marcotullio, 2007). It means that urban canals become an efficient sink for nitrogen.

Once N enters into the river system, in-stream N transformations could occur as a result of both physicochemical and biological processes. In-stream N transformations are primarily due to biogeochemical processes, which remove N from the stream water by transferring it to the biota, atmosphere, or stream sediments (Van Drecht et al., 2003). The corresponding nitrogen retention within the aquatic system represents about 70 % of all the inputs of NH_4^+ . Nitrification and algal uptake are suspected to occur in a high rate to explain the assimilative capacity of NH_4^+ in the Saigon River. The low level of NH_4^+ in river compared with urban canals is suggestively related to nitrification process (Nguyen et al., 2019) and warm weather conditions in this tropical environment (Nguyen et al., 2007; Luu et al., 2010).

Phosphorus budget.

The specific Total P fluxes were $67.4 \text{ kgP km}^{-2} \text{ yr}^{-1}$, and was higher than the values of Amazon River (Edmond et al., 1981) and lower than in Red River Delta (Le et al., 2005, 2007), Changjiang

River, Minjiang River, Jiulongjiang River, Zhujiang River and Jiazhou River in China (Zhang, 1996, Liu et al., 2005, 2016). This finding can be explained by a high retention of particulate P within the rivers systems, i.e. in the two reservoirs or in the low lands. Indeed P is known to adsorb easily onto suspended material and the fate of particulate P is closely related to the TSS flux. As an illustration, Red River Delta retained 50 % of the incoming flux of P from upstream (Luu et al., 2010). It appeared from Figure 9 that Total P and PO₄³⁻ fluxes exported by untreated domestic discharge accounted for 71% and 69% of the total annual fluxes of urban area into river. Based on data per capita, we calculated Total P flux released by resident into urban canals. We observed that resident released 4970 tonP into canals annually. However, less than five percent (3.7 %) of Total P flux was transported into the Saigon River by the urban canals. It means that over 96 % of P was stored in canals. In addition, PO₄³⁻ flux showed the storage of 55 % in canals. Adsorption of PO₄³⁻ onto sediment could explain the disequilibrium in the PO₄³⁻ flux budget. Contaminated conditions resulted from the release of untreated wastewater are likely to occur in such urban impacted river, like in Nhue River, near Hanoi (Trinh et al., 2009). Accumulation of phosphorus occurred where adsorption onto suspended sediments and sedimentation are high in low land rivers (Trinh et al., 2012). Such accumulation in sediment was evidenced in the urban canals of Bangkok, which presents similar geography, demography and wastewaters management as HCMC (Færge et al, 2001).

4.2. Nutrients fluxes ratios as an indicator of potential eutrophication

From the aforementioned budgets, it is interesting to propose ways of management. The indicator for coastal eutrophication potential (ICEP) proposed by Billen and Garnier (2007), is particularly relevant to evaluate the new production of non-siliceous biomass potentially sustained in the estuarine zone by either nitrogen or phosphorus delivered in excess over silica (Garnier et al., 2010). The ICEP was based on the Redfield molar C:N:P:Si = 106:16:1:20 (Redfield et al, 1963). For the purpose of river to river comparison, it is scaled to the river watershed area, express in specific flux in kgC km⁻² d⁻¹. According to the nutrient considered, either N-ICEP or P-ICEP can be defined, following relationships (5) or (6), respectively:

- If NFlux/PFlux < 16 (N limiting), where N:Si ad C:N are the Redfield ratio,

$$N - ICEP = \left[\left[\frac{NFlux}{14} - \frac{SiFlux}{28} \times \frac{N}{Si} \right] \times \frac{C}{N} \times 12 \right] \quad (5)$$

- If $NFlux/PFlux > 16$ (P limiting), where P:Si and C:P are the Redfield ratio,

$$P - ICEP = \left[\left[\frac{PFlux}{31} - \frac{SiFlux}{28} \times \frac{P}{Si} \right] \times \frac{C}{P} \times 12 \right] \quad (6)$$

Where $NFlux$, $PFlux$ and $SiFlux$ are the mean specific fluxes of Total N, Total P and DSi, delivered at the outlet of the river basin, expressed in $kgN\ km^{-2}\ d^{-1}$, $kgP\ km^{-2}\ d^{-1}$ and $kgSi\ km^{-2}\ d^{-1}$, respectively. A negative value of the N-ICEP or P-ICEP illustrates that silica is in excess over N or P; and would thus characterize limitation of eutrophication phenomenon, characterized by undesirable algae. Positive value indicates an excess of either N or P over the requirements for diatom growth, a condition for potentially harmful non-siliceous algal development such as cyanobacteria or dinoflagellates.

We observed that the Saigon River, under high human pressure, was characterized by N:P ratios above Redfield ratio ($N:P > 16$, Nguyen et al., 2019), which indicated an excess of nitrogen over phosphorus with respect to algal growth requirements. Thus P becomes a potential limitation of primary production. Further, the ICEP provided positive value for both Saigon River ($P-ICEP = 16.7\ kgC\ km^{-2}\ d^{-1}$) and Dongnai River ($P-ICEP = 8.5\ kgC\ km^{-2}\ d^{-1}$). The ICEP applied to the Saigon – Dongnai River provided positive values for P-ICEP ($9.8\ kgC\ km^{-2}\ d^{-1}$) (see Table 5-9). This indicated that nitrogen and phosphorus were presented in excess over silica requirement for algal growth. A positive ICEP also indicates high potential for eutrophication in the Saigon River with potential increase of cyanobacteria. Indeed, the Saigon River would really suffer from eutrophication phenomenon, which is confirmed from our measurements, Chlorophyll a being often higher than $25\ \mu g\ L^{-1}$ and up to $150\ \mu g\ L^{-1}$, far beyond good eutrophic value (Nguyen et al., 2019). Eutrophication did not yet occur in the downstream of the Saigon River – Soai Rap estuary; however, without an awareness of the situation, this coastal zone could rapidly face to eutrophication problem with the occurrence of undesirable algae that may threaten the aquaculture within the Can Gio mangrove area. With the planification of building new WWTPs, this situation could improve in the near future.

Table 5-9. Specific nutrient fluxes delivered by Saigon and Dongnai River basin and the indicator of coastal eutrophication potential (ICEP)

	Saigon River basin (4 717 km ²)	Dongnai River basin (26 499 km ²)	Saigon and Dongnai River basin (31 216 km ²)
N:P (FluxN:FluxP)	29	27	27
P specific flux (kgP km ⁻² d ⁻¹)	0.49	0.39	0.40
N specific flux (kgN km ⁻² d ⁻¹)	6.49	4.72	4.98
Si specific flux (kgSi km ⁻² d ⁻¹)	1.59	3.22	2.97
P-ICEP (kgC km ⁻² d ⁻¹)	16.7	8.5	9.8
N-ICEP (kgC km ⁻² d ⁻¹)	33.2	19.5	21.6

4.3. Future nutrients emissions from Ho Chi Minh City by 2025-2050 and recommendation for better management

In coming decades, gross nutrients inputs is expected to increase by two fold in 2040 and three fold in 2050, for both Total N and Total P fluxes. The expected construction and operation of ten new WWTPs should lead to a decrease in Total N and P fluxes from urban canals to rivers (e.g., a decrease from about 18 tonN yr⁻¹ in 2016 to 10 tonN yr⁻¹ in 2040 and from about 5 tonP yr⁻¹ in 2016 to 3 tonP yr⁻¹ in 2040, Table 5-7). The HCMC megacity should rise to 23 million inhabitants in 2050; consequently, the net nutrients fluxes from canals to rivers increase to about 30 tonN yr⁻¹

¹ and 8 tonP yr⁻¹. It means that the population of 17 million inhabitants in 2040 is the maximal value for treated capacity of ten WWTPs.

To evaluate the effect of the construction of ten WWTPs in future, we estimate nutrients ratio and ICEP index based on nutrients fluxes in 2025, 2040 and 2050. Nguyen et al. (2019) observed that dissolved silica dynamics were stable, thus for this estimation, we assume that silica fluxes marked at stability. As discussed above, we observed that the main source of nutrients fluxes in the Saigon River comes from urban canals. Indeed, about 58% of Total P flux and 48% of Total N flux in the Saigon River are originated from urban canals (Figure 5-8a and 5-9a). Therefore, river fluxes in future is a sum of river fluxes in present and nutrients inputs from urban canals to rivers. Based on this hypothesis, we observed a change in P limiting factor to N limiting factor for algal development in the Saigon River. While P becomes a potential limitation of primary production during period of 2012-2016 (N:P > 16, Nguyen et al., 2019); N:P ratio is below 16, which indicated an excess of phosphorus over nitrogen with respect to algal growth requirements. Further, the P-ICEP increased remarkable from 16.7 kgC km⁻² d⁻¹ during period of 2012-2016 to 369 kgC km⁻² d⁻¹ in 2050. It indicates extremely high potential for eutrophication in the Saigon River with potential increase of undesirable algae. Eutrophication will occur in the downstream of the Saigon River and the coastal zone could face to eutrophication problem with the occurrence of undesirable algae.

The results presented in this article highlight the challenges that the megacity of HCMC is now facing to ensure jointly good economic and ecological health. The estimation of future nutrients inputs underlines clearly the need to build the ten WWTPs, as planned in the spatial planning framework. HCMC is not unique and exemplified the trajectory that many coastal megacities are (or will) experiencing in Asia and Africa.

5. Conclusions

Based on dataset of water discharge, TSS, and nutrients (N and P) concentrations, we assessed that the Saigon – Dongnai rivers basin delivered 1.47×10^6 tonSS yr⁻¹, 36.4×10^3 tonN yr⁻¹ and 2.1×10^3 tonP yr⁻¹ into the estuary. Seasonal and spatial variations are closely linked with water discharge and hydrological patterns of Saigon and Dongnai watersheds. The fluxes of TSS and nutrients were found to be highly variable between the two rivers. The absence of any clear relationship between nutrient flux and sediment characteristics implies that there were other factors controlling

fate of nutrients. This first budget calculation highlights the worrying impact of untreated domestic wastewaters released by HCMC on to the estuarine system. Especially urban canals are likely responsible of high accumulation of TSS and P. The distinct shift of nutrient ratio in estuary is also evidenced by negative ICEP values within the Saigon – Dongnai Rivers system. These notable changes to the exports and ratios of nutrient variables and anthropogenic driven nutrient concerns are becoming a defining feature of the Saigon – Dongnai Rivers to the East Sea. Because of its morphology, the estuary in the downstream of HCMC enhances dilution and biogeochemical transformation of the highly polluted Saigon River waterbody. For this reason, the aquatic ecosystem located downstream of HCMC would not yet really suffer from eutrophication phenomenon. The Saigon River, which crosses the heart of the megacity has concentrations of N and P 7 times and 2 times higher than the Dongnai Rivers, respectively. This ongoing increase of nutrients release in the aquatic ecosystem has and will have some consequences on the biogeochemical cycles that need to be assessed and anticipated. Even if the financing plan is not yet consolidated, the local authorities have already planned to build ten new WWTP in the next ten years to reduce the direct release of untreated wastewaters and to recover good water quality, which is believed to be a mandatory to secure the water resource.

Acknowledgements

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Conclusion of Chapter 5

This first budget calculation highlights the evident impact of untreated wastewater of HCMC. It shows also the high contribution of HCMC to the total TSS and nutrient fluxes delivered to estuary. The TSS and nutrients fluxes were found to be highly variable within the Saigon – Dongnai River basin. Urban canal network is responsible of high accumulation of TSS and P. The notable change in ICEP index and ratios of nutrient variables and anthropogenic driven nutrient concerns are becoming a defining feature of the Saigon – Dongnai Rivers. Because of the dilution and biogeochemical transformation of the highly polluted Saigon River waterbody by the estuary in the downstream of HCMC, the aquatic ecosystem would not yet really suffer from eutrophication phenomenon.

GENERAL CONCLUSIONS AND PERSPECTIVES

The present study aims at better understanding the relationships between urbanization and water quality in one of the most exposed low elevation coastal zone of the world, i.e. South East Asia. The study focuses on Ho Chi Minh City and Saigon-Dongnai Rivers system, in South of Vietnam.

This thesis not only provides a critical analysis of the water quality, nutrient behavior and nutrient budget in the Saigon – Dongnai River system, a tropical river located in Southern Vietnam, but also assesses the impact of human activities (domestic, industrial and agricultural) on a tidally influenced ecosystems. Beside, this work has provided an overview of nutrient inputs from upstream part of the Saigon River to Can Gio Mangrove in the downstream of HCMC, as well as the transformation, retention and transfer of nutrients within the salinity gradient. The approach adopted relied first on a bi-monthly monitoring (from July 2015 to December 2017), secondly, on some surveys conducted during wet and dry seasons of 2017 and finally, on the analysis and post processing of a national monitoring database supported by the Center of Environmental Monitoring of the Ho Chi Minh City Department of Natural Resources and Environment. While previous studies have mainly focused on risk hazards (flood disasters), and water regulation issues (irrigation system, reservoir construction), our concern is on human activities and their impacts on aquatic ecosystems.

The outputs of this work can be summarize as follow:

1. The field studies carried out during 2.5 years showed that water quality in the study area is deteriorating, principally because of development of human activities related to economic and demographic booms in HCMC. The pollution in the heart of HCMC, the highest population-concentrated city in Vietnam (4 029 inhabitants km²) reached an alarming level, particularly in urban canals, where suspended sediment, nitrogen and phosphorus contamination, bad smell, dark color were observed.
2. Phosphorus peaks of concentration was observed in the heart of HCMC with values about three times higher than the ones measured upstream, in the fluvial section, and downstream, in the estuarine environment. Particulate inorganic P represented 80% of total particulate P. This thesis illustrated the strong influence of SS concentrations on the P adsorption onto sediment, e.g. P adsorption capacity increase when SS concentrations rose. SS

concentration revealed to be the main driver for adsorption capacity of P onto suspended sediments, far beyond salinity and turbulence factors. These results were obtained through innovative techniques of measurements of particles sedimentation and flocculation (SCAF instrument), coupled with experimental adsorption desorption in controlled laboratory conditions

3. Regarding the hydrology, available data – water discharge in the Saigon – Dongnai River were used for establishing a general map of nutrients budget. The pluri-annual database allowed assessing the Saigon and Dongnai Rivers' hydrology system, underlining that both rivers are strongly affected by tidal oscillations and, at a secondary level, by seasonality.
4. Based on this, the water budget was established for five years in order to quantify the main water flows in the Saigon – Dongnai River system. The results show that the mean annual discharge delivered to the estuary was around $50 \text{ m}^3 \text{ s}^{-1}$ from the Saigon River and around $630 \text{ m}^3 \text{ s}^{-1}$ from the Dongnai River. In total urban contribution to the budget was estimated at about $10 \text{ m}^3 \text{ s}^{-1}$, mainly from untreated wastewaters discharges. The efforts should be pursued to get a robust estimation of water discharge.
5. Upstream to downstream budgets have been assessed for suspended sediment and nutrients (N, P and Si). This first budget calculation highlights the evident impact of untreated wastewaters of HCMC. It shows also the contribution of HCMC to the total flux of nutrients and suspended sediment (through riverbank erosion) to coastal area. We demonstrated that the estuarine system, downstream of the megacity, has an important role in retaining/eliminating, phosphorus and nitrogen coming from the upstream part. The calculation of the indicator of coastal eutrophication potential (ICEP) showed a negative value, which indicates that silica is present in excess over the other nutrients and would thus characterize the absence of undesirable algal development, even during critical eutrophic states

Despite the difficulties due to alternations between France and Vietnam, bi-monthly surveys were conducted continuously thank to the Saigon River project, funded by the region AURA and EC2C2 Bioeffect Initiative project, and the collaborative work initiated between French and Vietnamese teams. Some recommendations can be addressed to improve scientific knowledge and create a dynamic bridge between knowledge and practical applications:

1. Additional hydrological surveys should be organized in order to get a comprehensive understanding of the tidal system and of the water flux income from urban canals.
2. Investigations to get a synchronic and comprehensive data set on water quality and pollution sources to rivers, especially at the scale of urban canals.
3. The modelling approach is needed to solve the spatio-temporal variability of nutrient inputs in the estuarine systems, by combining and reproducing the hydrodynamics and the biogeochemical processes. This model is on the way to be implemented in the Saigon River in order to test some scenario of water quality recovery (building of new WWTPs) or anticipation of new degradation (population increase and uncontrolled urbanization).
4. As discussed in chapter 3, the Saigon River episodically experienced extreme eutrophication episodes. Therefore, there are numerous problems and challenges with regards to water quality management in HCMC. The main question is how to revise, improve and implement the current legal system regarding the water sector in general, and wastewater management sector in particular? Appropriate quantification of the eutrophication status was a first step which scientists and policy makers need to discuss to now go through operational modelling tools: i.e. to go from observations to mitigation.
5. The application of a numerical model can help us understanding and assessing the link between human activities and water quality in tropical rivers and estuaries. A GIS database should be assembled at the scale of the whole Saigon – Dongnai basin, with layers documenting geomorphology, lithology, meteorology, land use and agriculture, population, industrial wastewater release for estimating the role of natural and anthropogenic factors on the water quality and biogeochemical functioning of the river system.

While the low elevation coastal zone (LECZ, at elevation below 10 meters above sea level) of Asia gathers most of the coastal population in the world (~75%), this study provides an interesting case study of the behavior of a tropical river under intense human pressure and may help stakeholders to anticipate risks and adopt appropriate strategies for the sustainable management of tropical rivers and estuaries.

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APPENDIX

Appendix 1. Phytoplankton dynamics

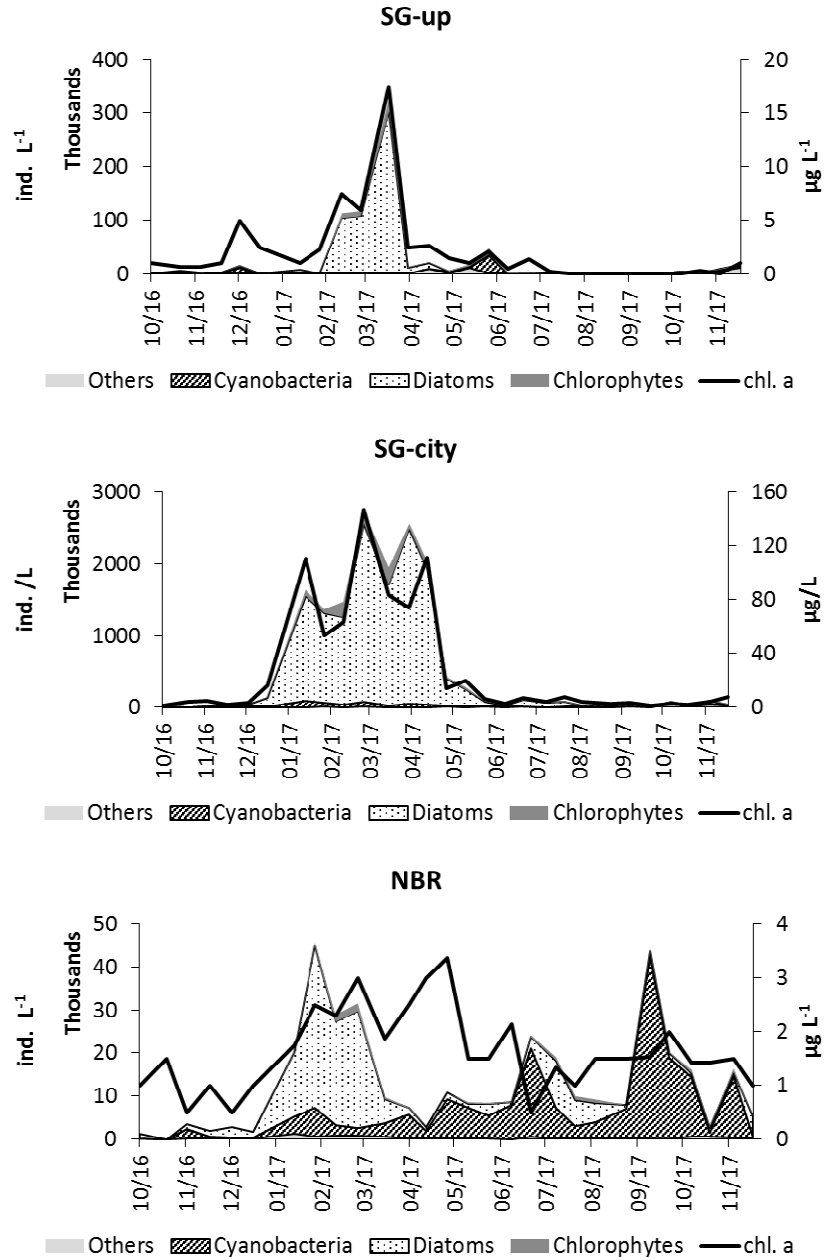


Figure A-1. Phytoplankton abundance along the Saigon from October 2016 to December 2017 (coll. Thanh-Son Dao, HCMUT).

Appendix 1. Phytoplankton dynamics

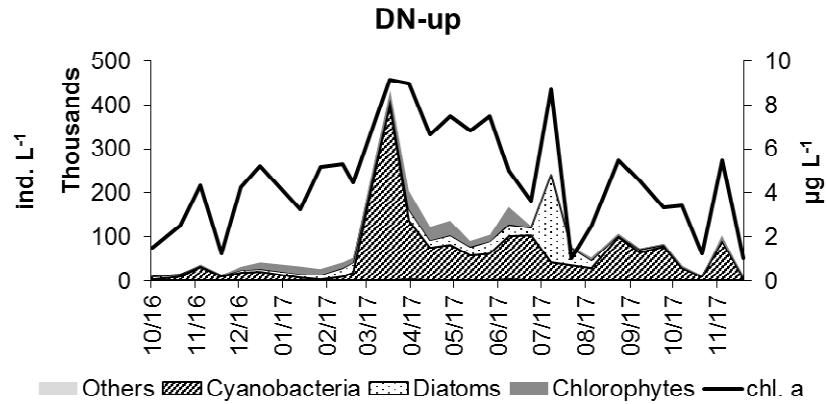


Figure A-2. Phytoplankton abundance in the Dongnai Rivers from October 2016 to December 2017 (coll. Thanh-Son Dao, HCMUT).

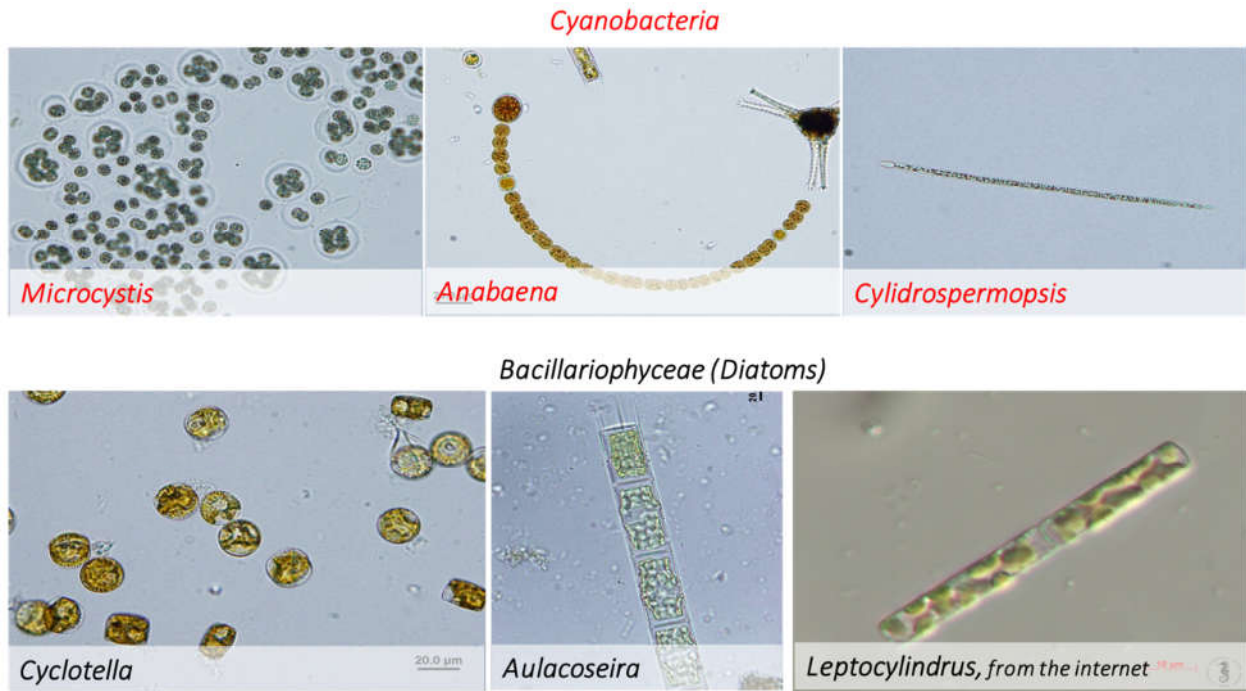


Figure A-3. Dominant species of phytoplankton in the studied area.

Appendix 1. Phytoplankton dynamics

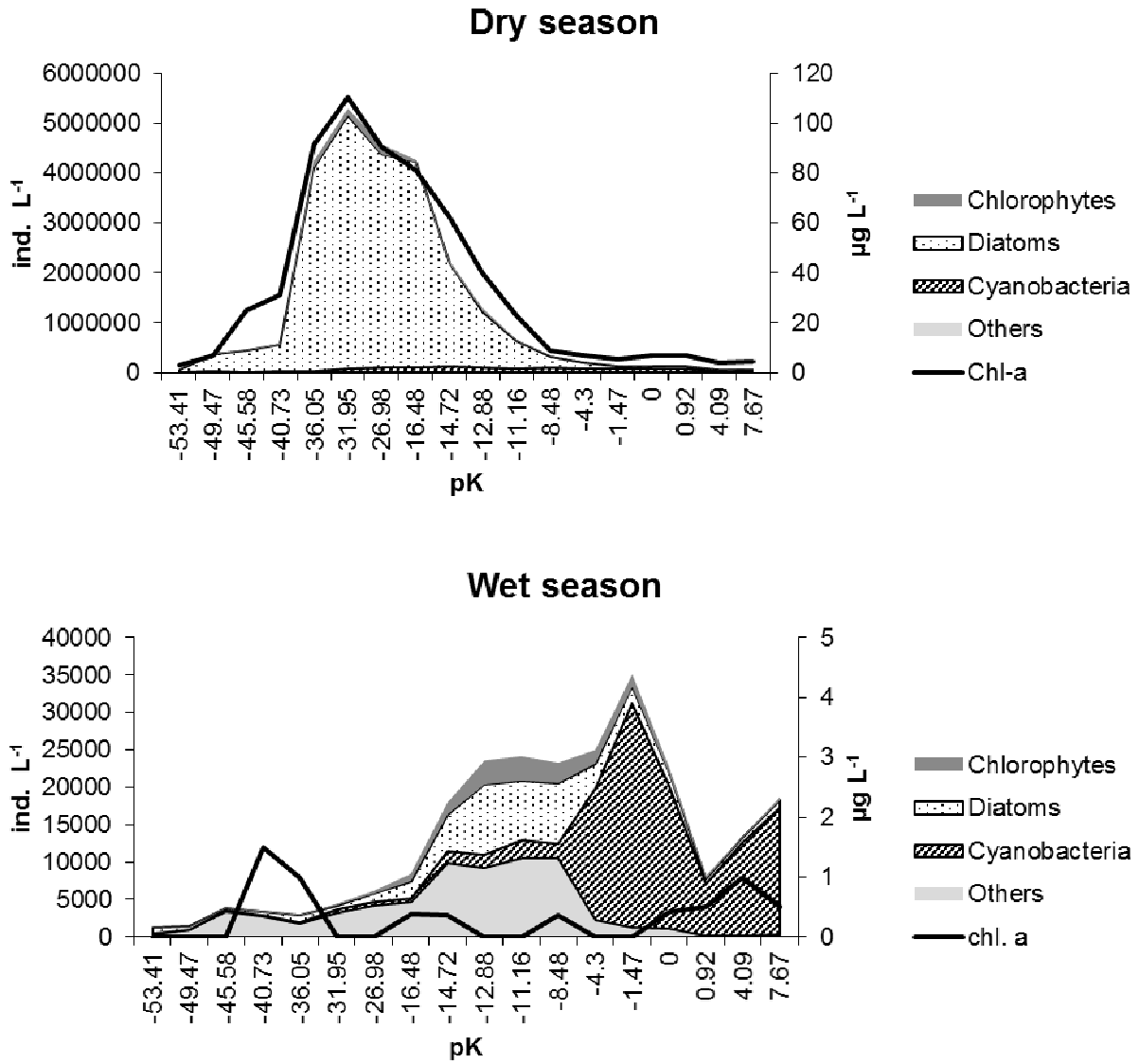


Figure A-4. The longitudinal profiles of phytoplankton along the Saigon River during the dry and wet season in 2017.

Appendix 1. Phytoplankton dynamics

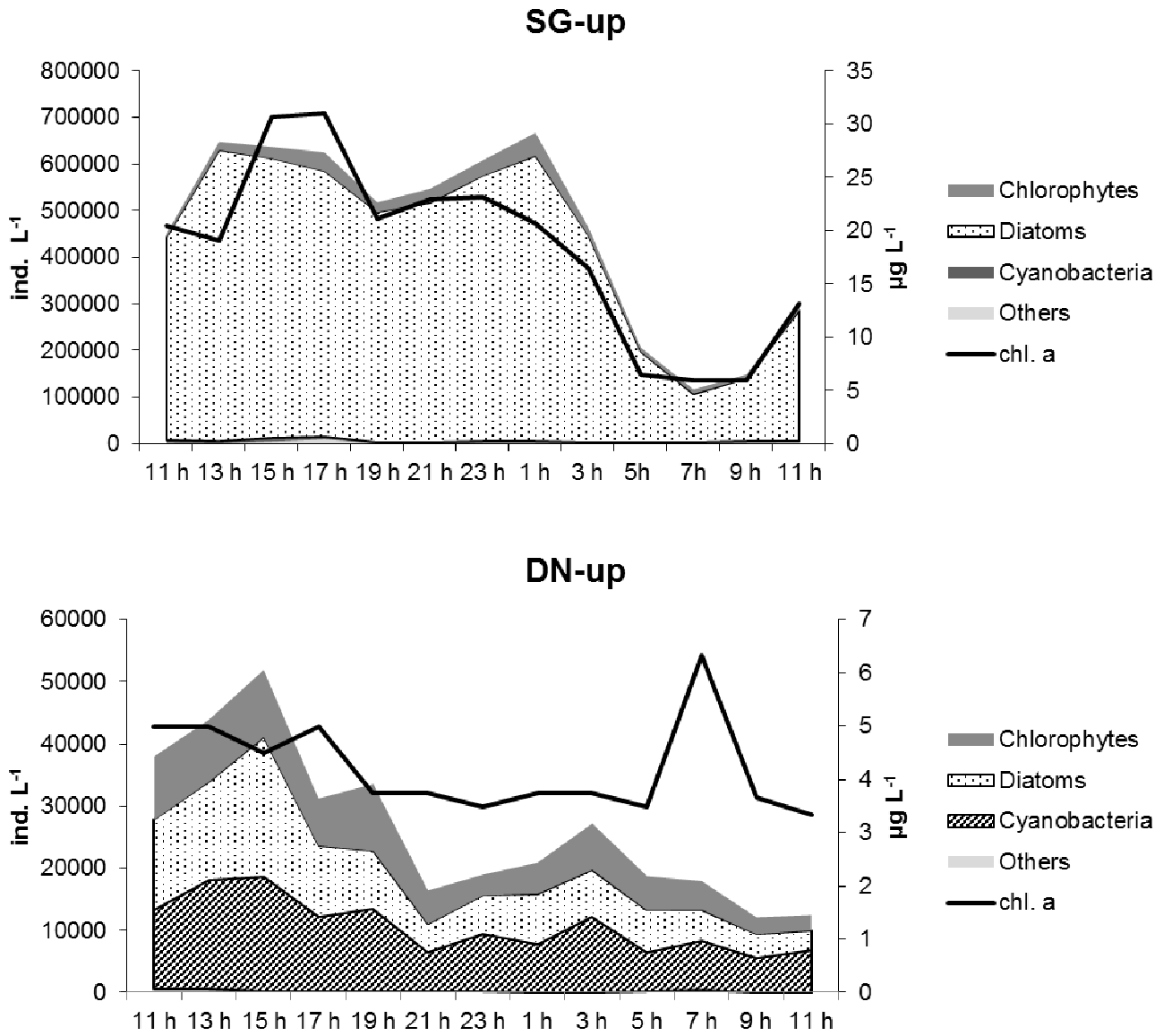


Figure A-5. The diurnal survey during the dry season (March 2017) at the upstream of the Saigon River (SG-up) and the Dongnai River (DN-up).

Appendix 1. Phytoplankton dynamics

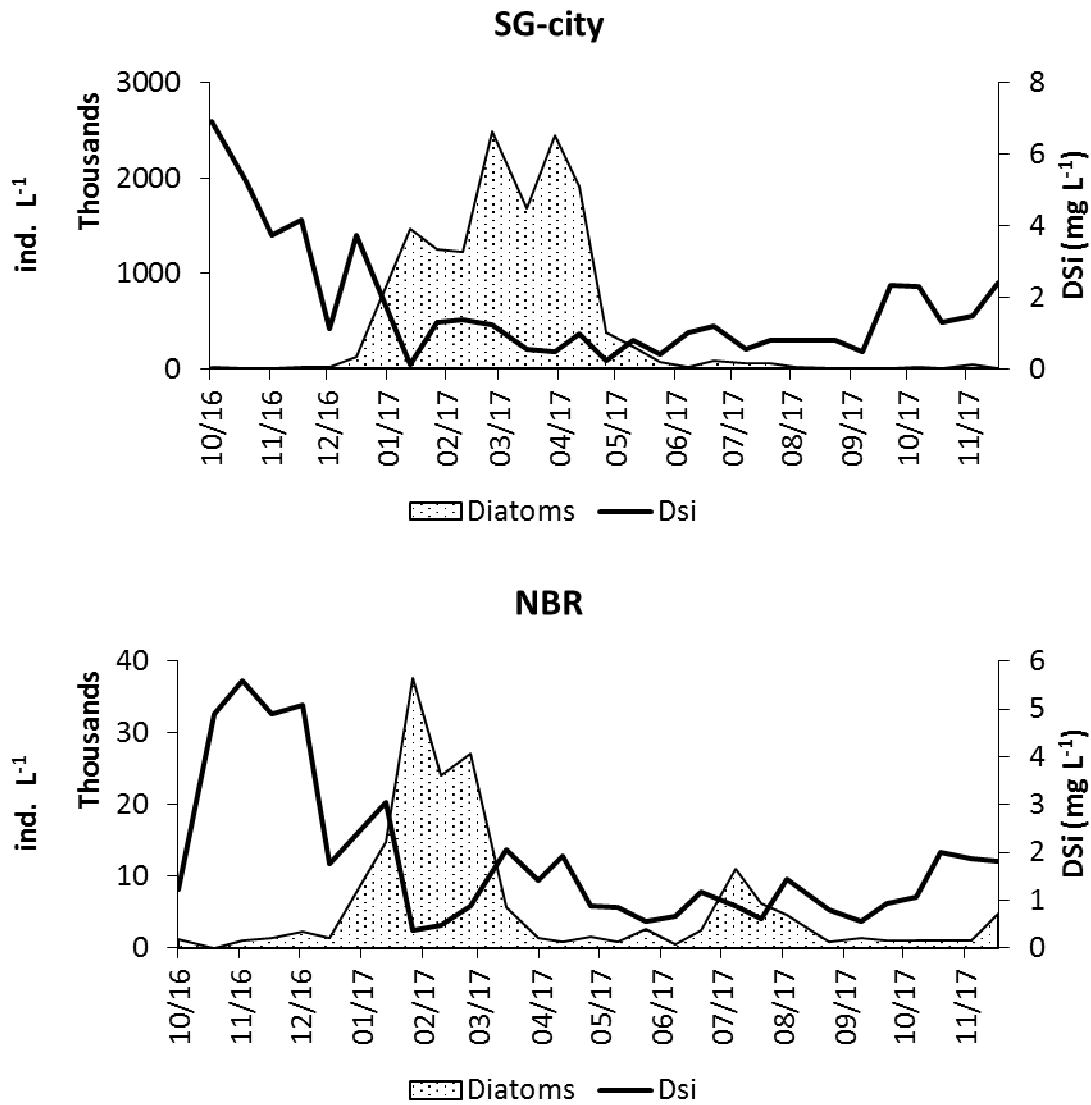


Figure A-6. Variability of diatom and DSI at SF-city and NBR from October 2016 to December 2017.

Appendix 2. Diurnal sampling surveys at Hoa An (DN-up, the Dongnai River)

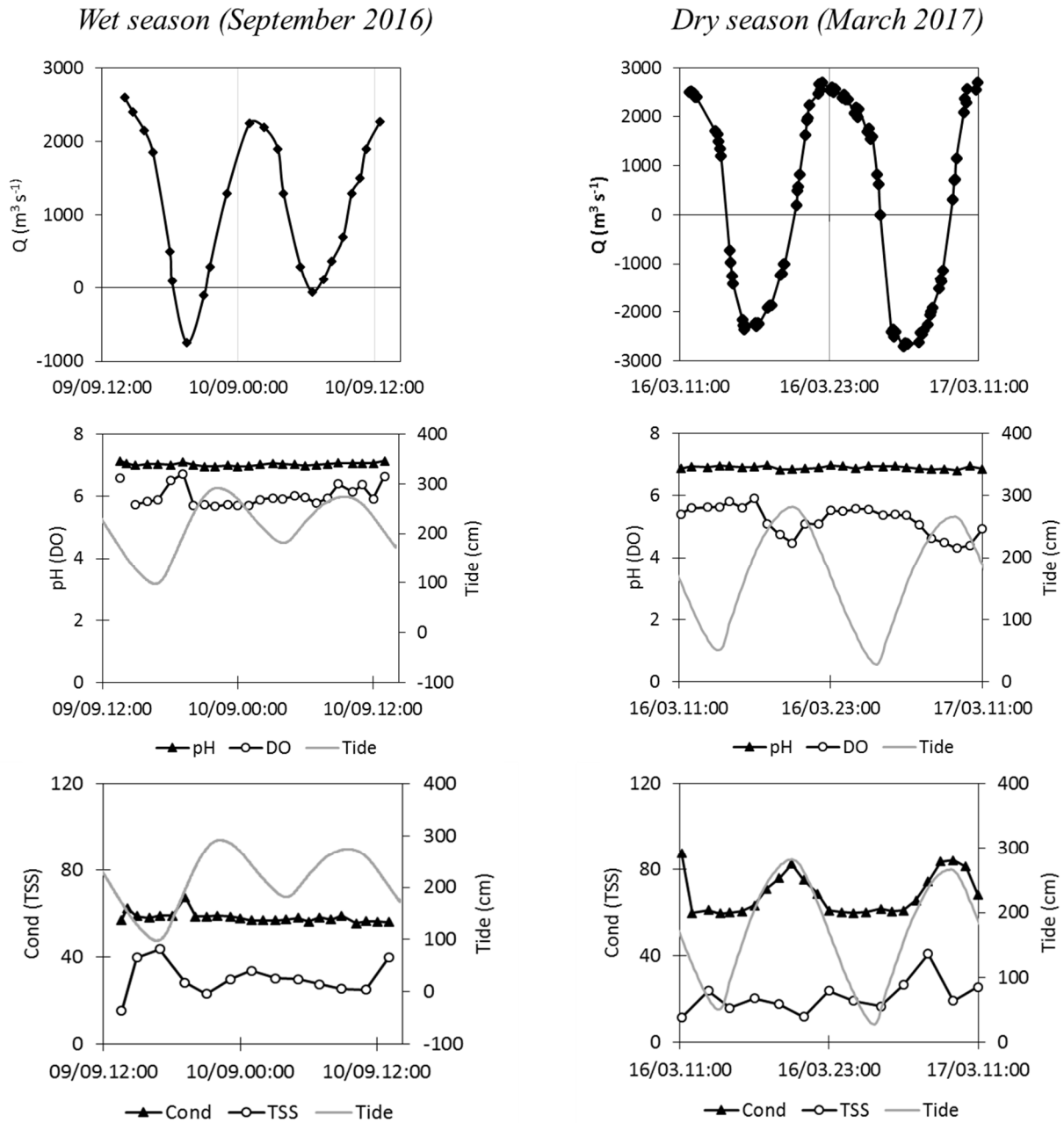


Figure A-7. Variation of water discharge and physico-chemical parameters at Hoa An (DN-up) during 24h cycle during the wet and dry season. DO: dissolved oxygen ($mg L^{-1}$), Cond: conductivity ($\mu S cm^{-1}$) and TSS: Total Suspended sediment ($mg L^{-1}$).

Appendix 2. Diurnal sampling surveys at Hoa An (DN-up, the Dongnai River)

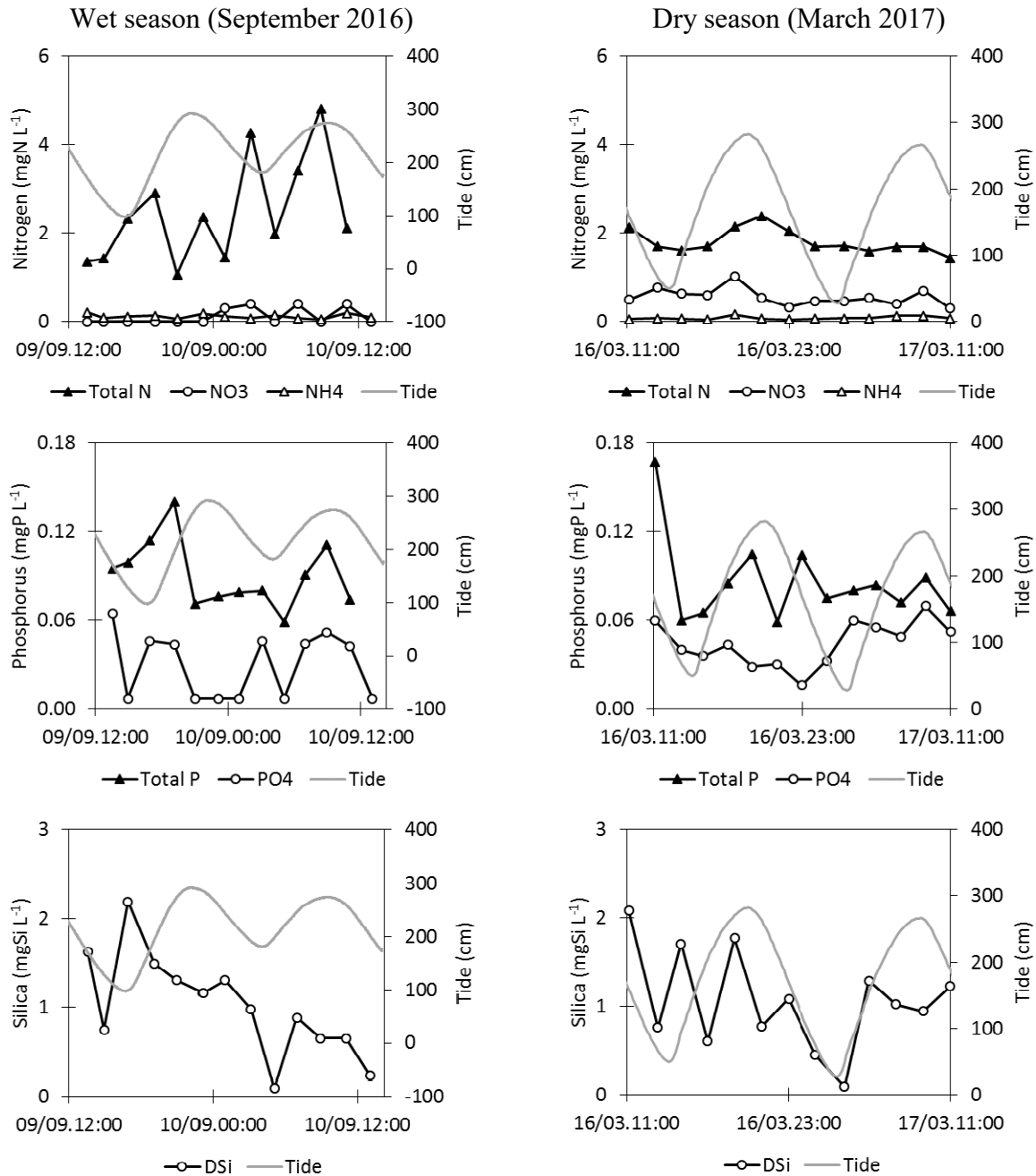


Figure A-8. Variation of nutrient concentration at Hoa An (DN-up) during 24h cycle in wet and dry season. Total N: Total nitrogen, NO₃: nitrate, NH₄: ammonium, Total P: Total phosphorus, PO₄: orthophosphate and DSi: dissolved silica.

Appendix 2. Diurnal sampling surveys at Hoa An (DN-up, the Dongnai River)

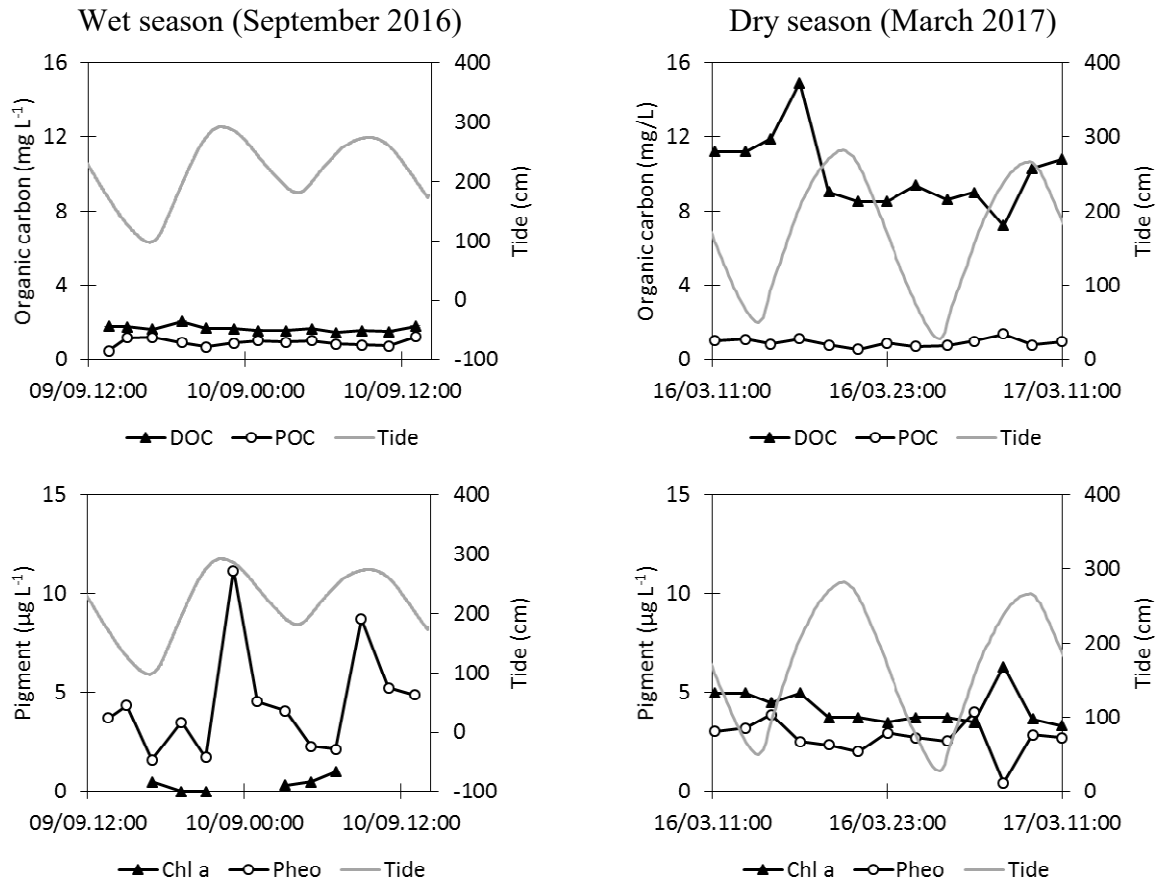


Figure A-9. Variation of organic carbon and Chlorophyll a concentration at Hoa An (DN-up) during 24h cycle in wet and dry season. DOC: dissolved organic carbon, POC: particulate organic carbon, Chl-a: Chlorophyll a and Pheo: Phaeopigments.

Appendix 2. Diurnal sampling surveys at Phu Cuong (SG-up, the Saigon River)

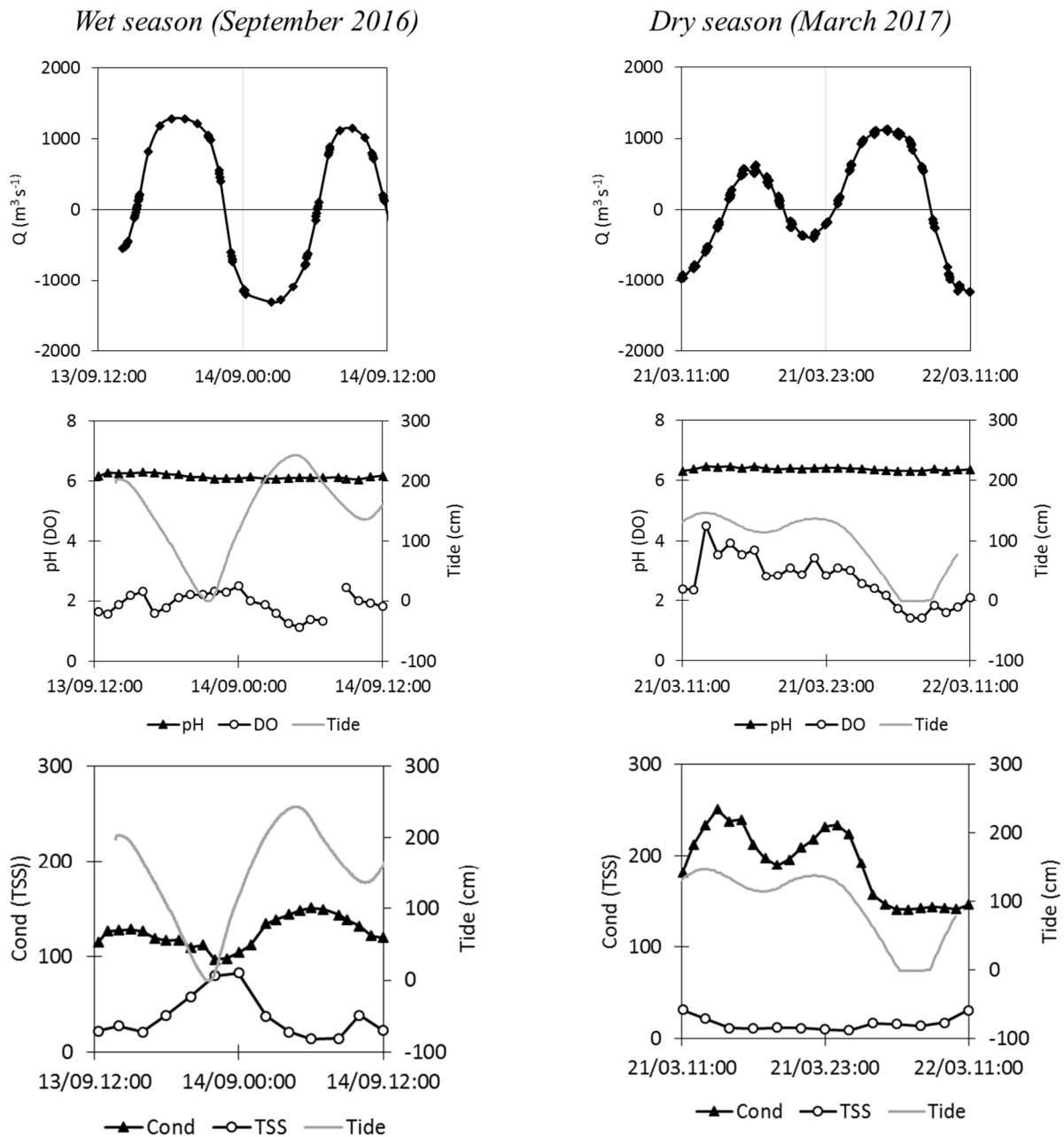


Figure A-10. Variation of physico-chemical parameters at Phu Cuong (SG-up) during 24h cycle in wet and dry season. DO: dissolved oxygen (mg L^{-1}), Cond: conductivity ($\mu\text{S cm}^{-1}$) and TSS: Total Suspended sediment (mg L^{-1}).

Appendix 2. Diurnal sampling surveys at Phu Cuong (SG-up, the Saigon River)

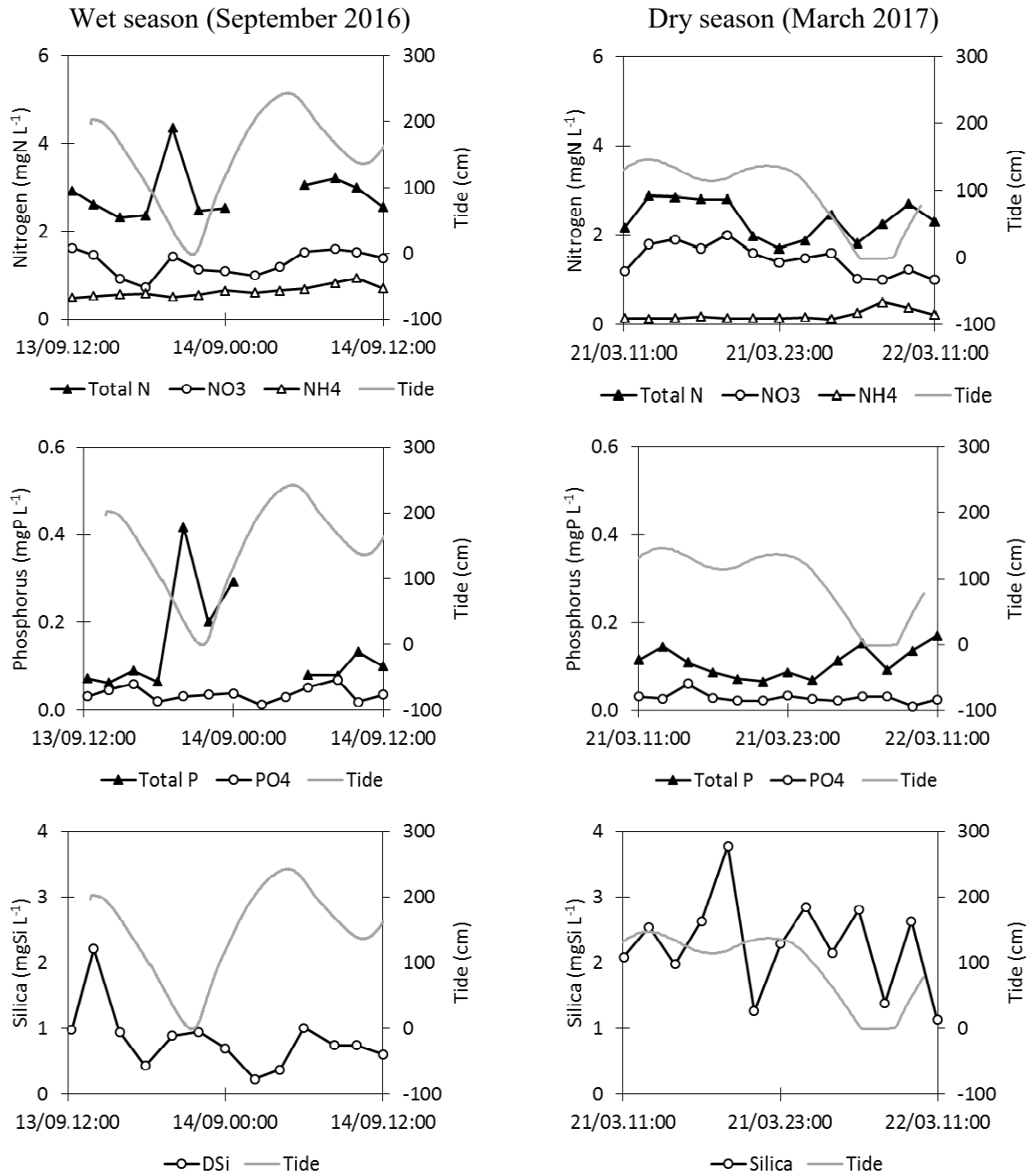


Figure A-11. Variation of nutrient concentration at Phu Cuong (SG-up) during 24h cycle in wet and dry season. Total N: Total nitrogen, NO3: nitrate, NH4: ammonium, Total P: Total phosphorus, PO4: orthophosphate and DSi: dissolved silica.

Appendix 2. Diurnal sampling surveys at Phu Cuong (SG-up, the Saigon River)

Wet season (September 2016)

Dry season (March 2017)

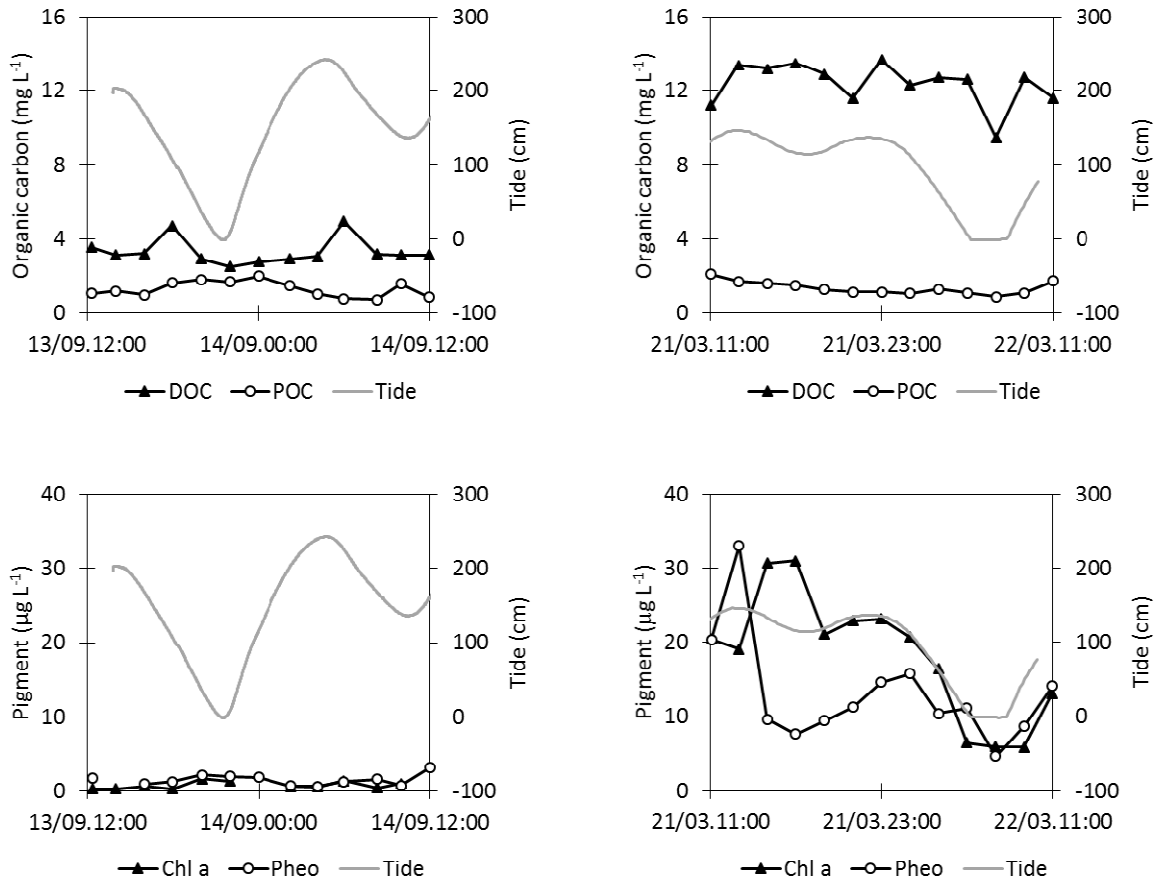


Figure A-12. Variation of organic carbon and Chlorophyll a concentration at Phu Cuong (SG-up) during 24h cycle in wet and dry season. DOC: dissolved organic carbon, POC: particulate organic carbon, Chl-a: Chlorophyll a and Pheo: Phaeopigments.

Appendix 3. Atmospheric deposition

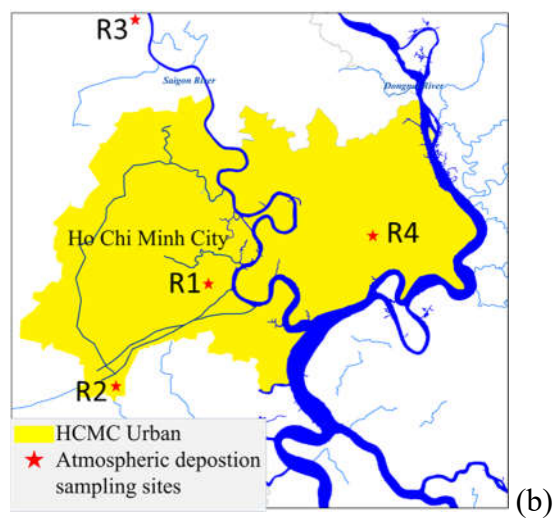


Figure A-13. Sampling sites of atmospheric deposition monitoring.

Appendix 3. Atmospheric deposition

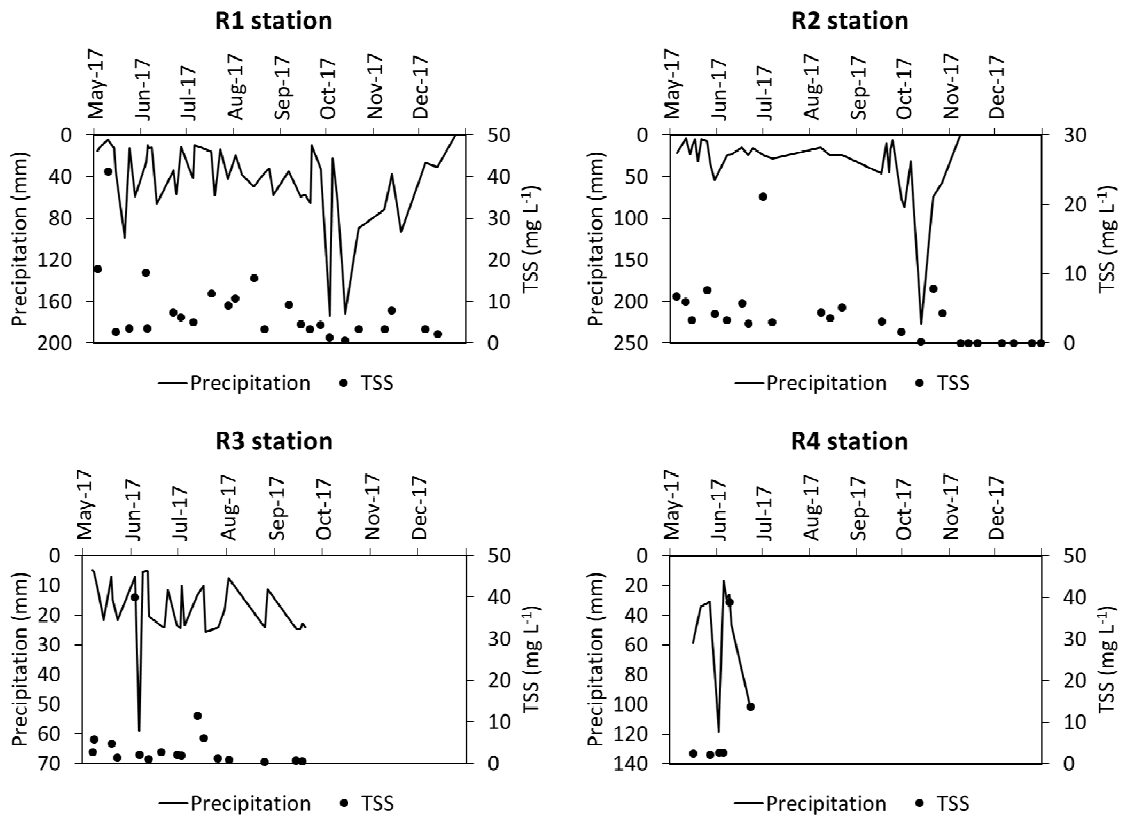


Figure A-14. Precipitation and TSS concentration variation during the wet season in 2017 at 4 sampling sites in HCMC.

Appendix 3. Atmospheric deposition

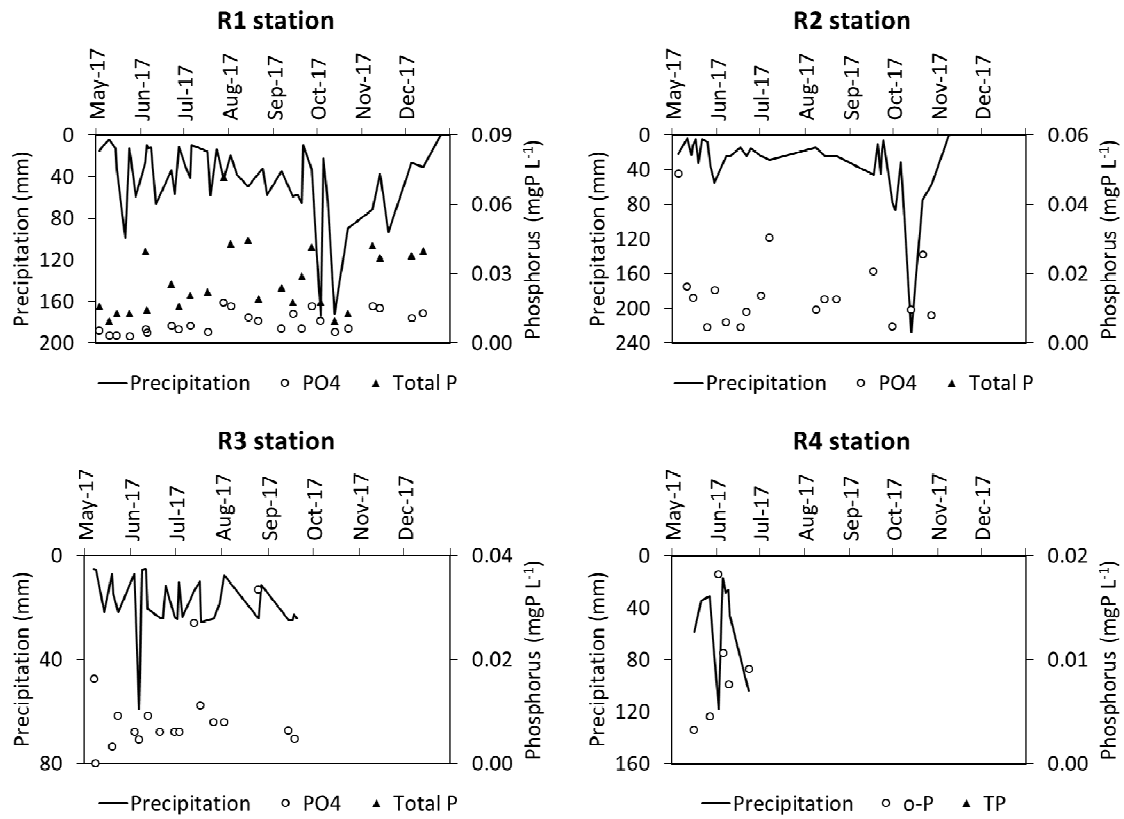


Figure A-15. Precipitation and Phosphorus concentration variation during the wet season in 2017 at 4 sampling sites in HCMC.

Appendix 3. Atmospheric deposition

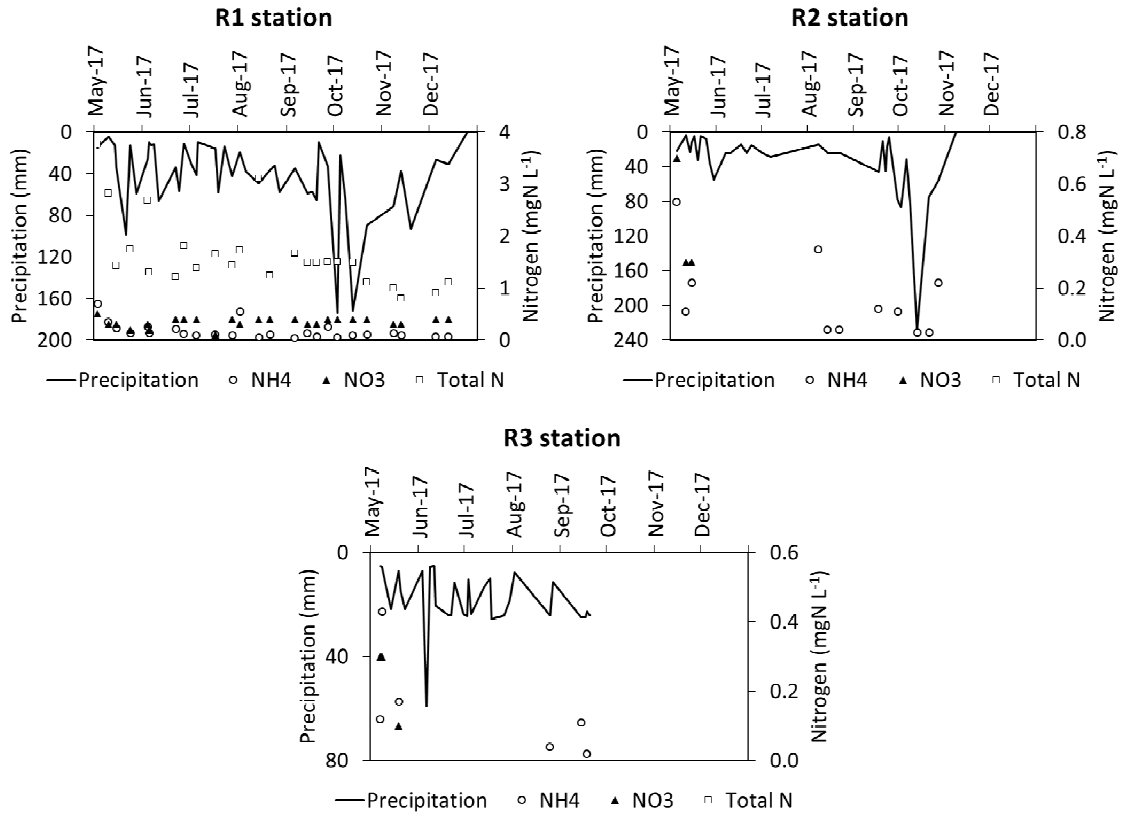


Figure A-16. Precipitation and Nitrogen concentration variation during the wet season in 2017 at 4 sampling sites in HCMC. Missing data for station R4.

Appendix 3. Atmospheric deposition

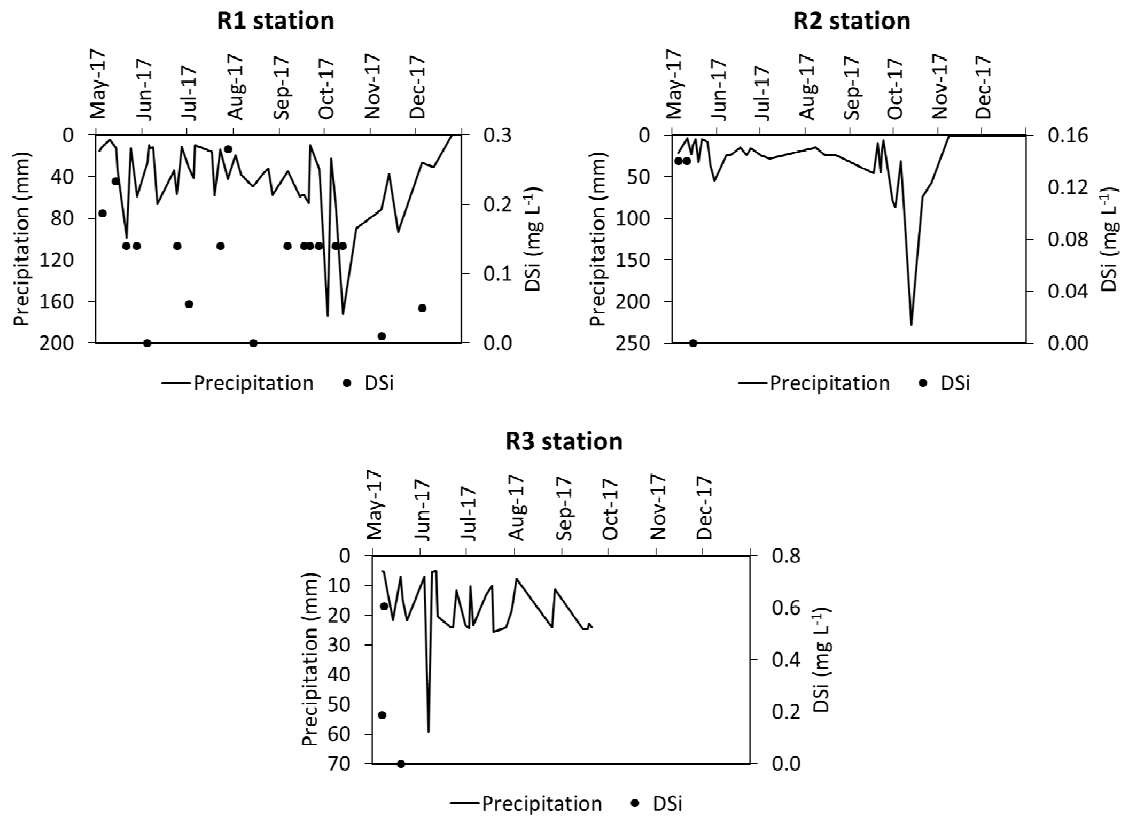


Figure A-17. Precipitation and Silica concentration variation during the wet season in 2017 at 4 sampling sites in HCMC. Missing data for station R4.

Appendix 4. Hydrodynamic: water discharge at 6 sites within the Saigon and Dongnai Rivers

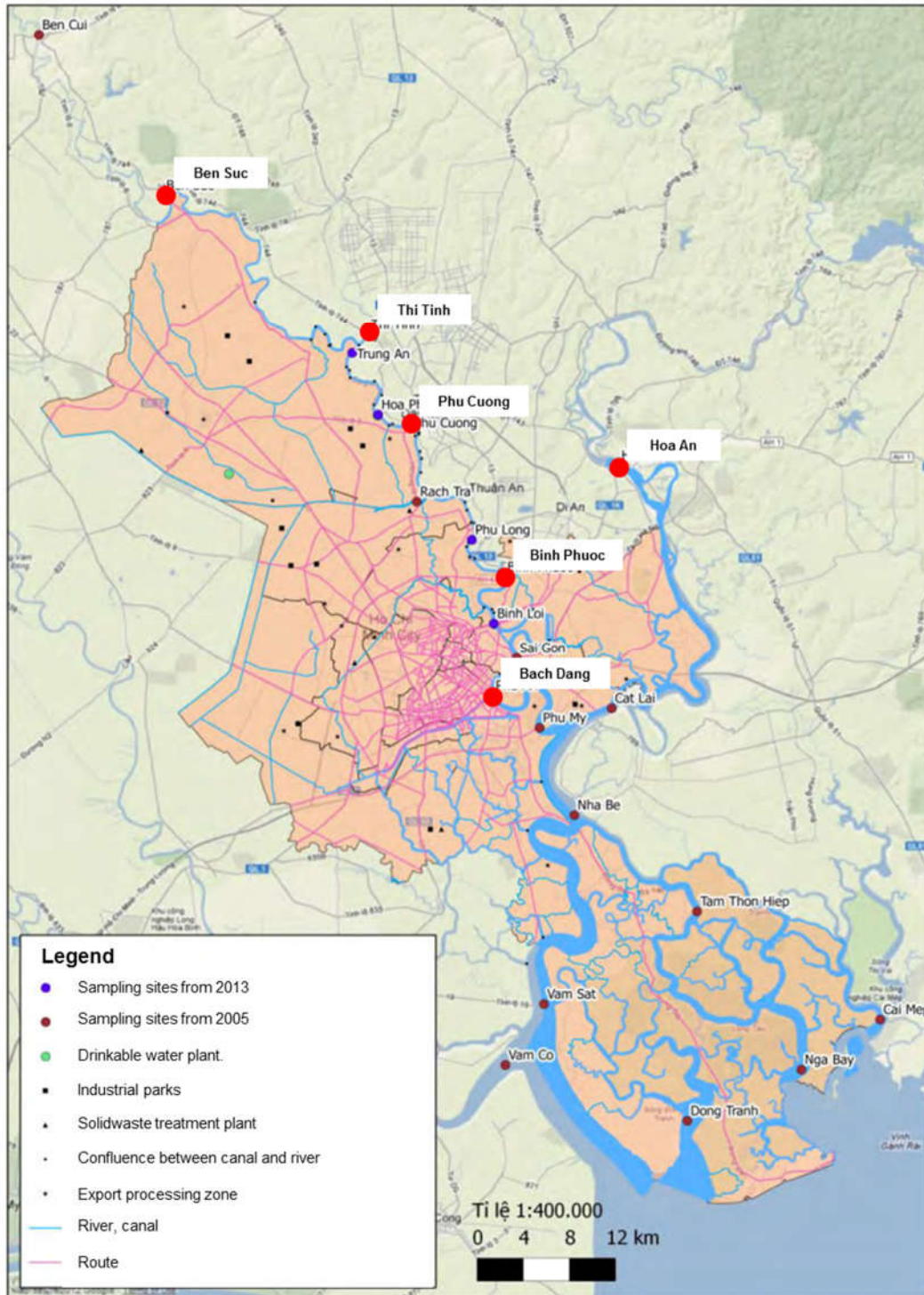


Figure A-18. Sampling sites for water discharge measurement within the Saigon – Dongnai River basin (source: CEM-DONRE).

Appendix 4. Hydrodynamic: water discharge at 6 sites within the Saigon and Dongnai Rivers

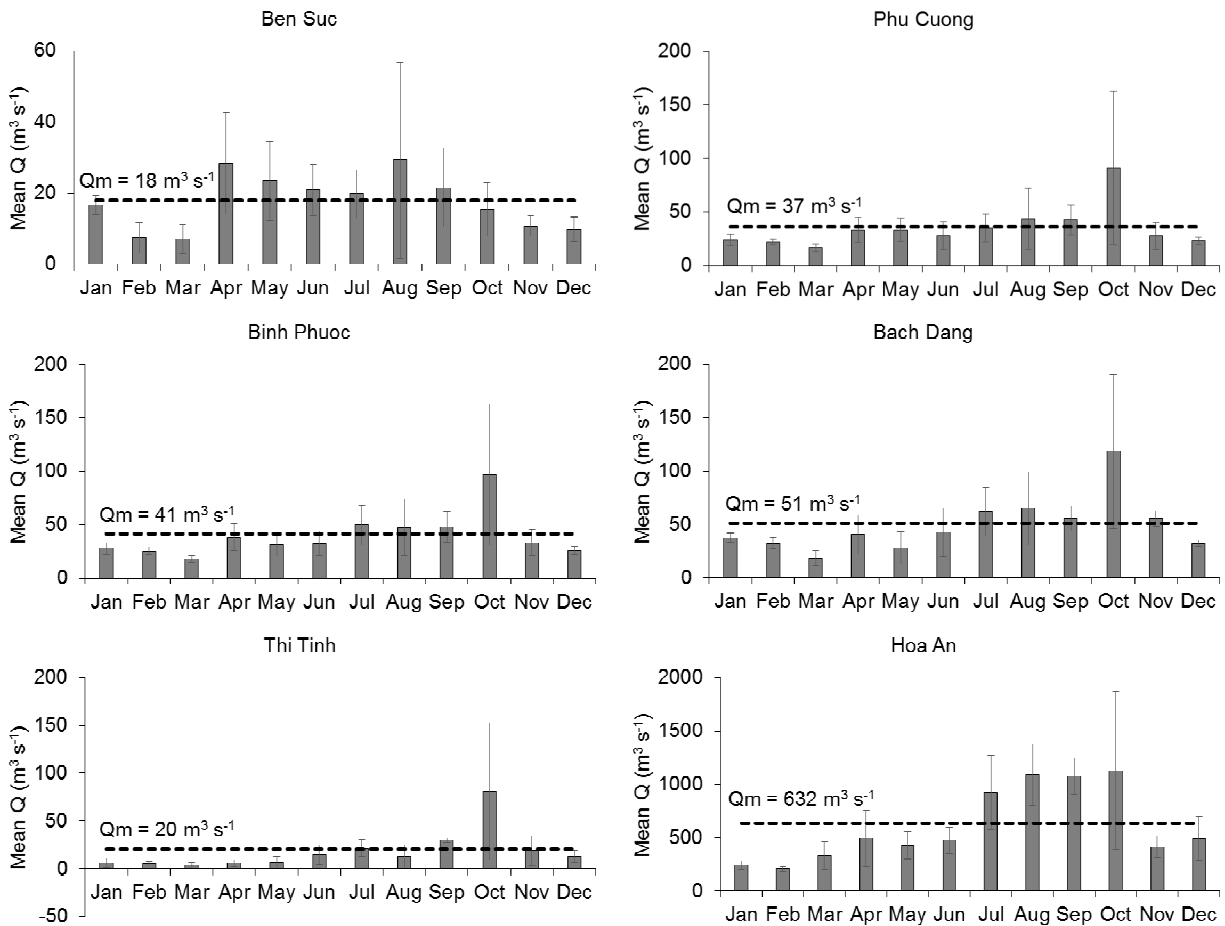


Figure A-19. Distribution of mean monthly discharges (period 2012-2016) in Saigon River and Dongnai River (data source: DONRE; see Figure 1 for the location).

Appendix 5. TSS, NH_4^+ and PO_4^{3-} concentrations in urban canals

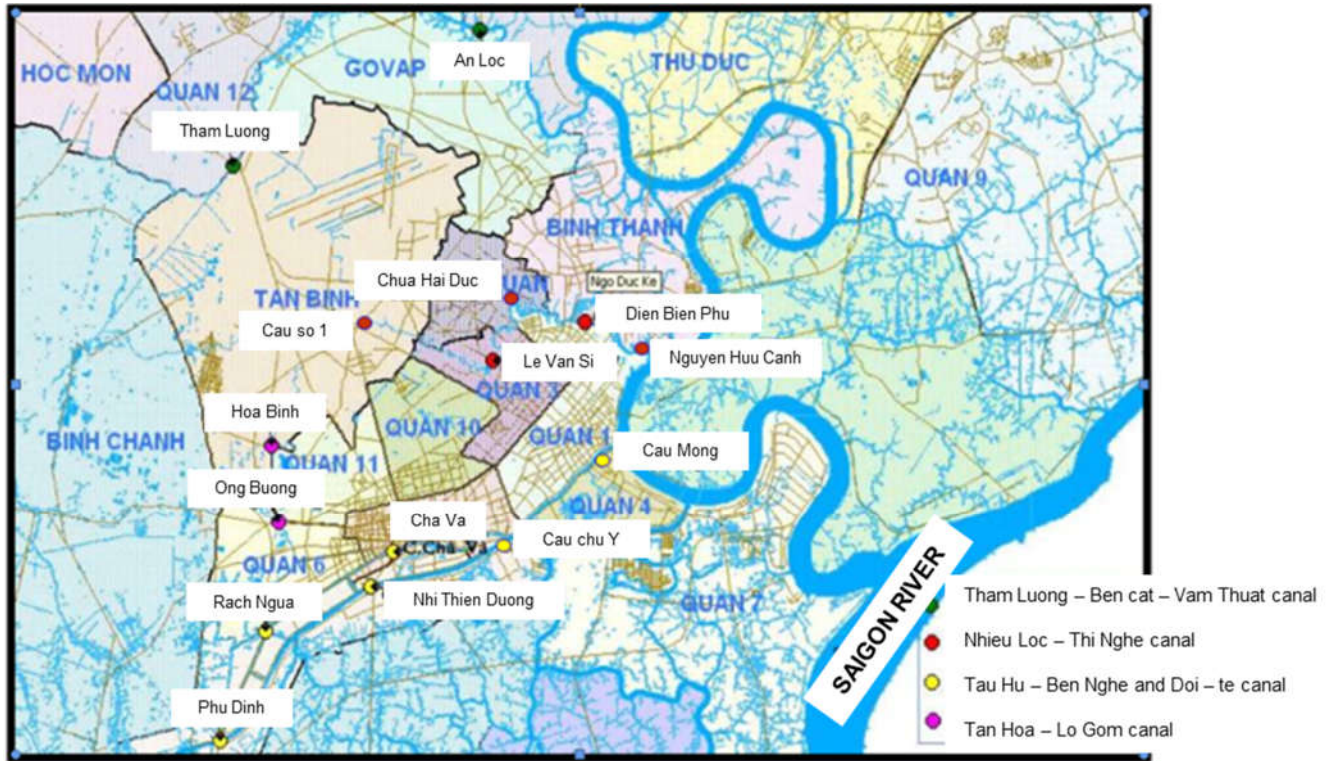


Figure A-20. Sampling sites for TSS, NH_4^+ and PO_4^{3-} measurement in urban canal network (source: CEM-DONRE).

Appendix 5. TSS, NH_4^+ and PO_4^{3-} concentrations in urban canals

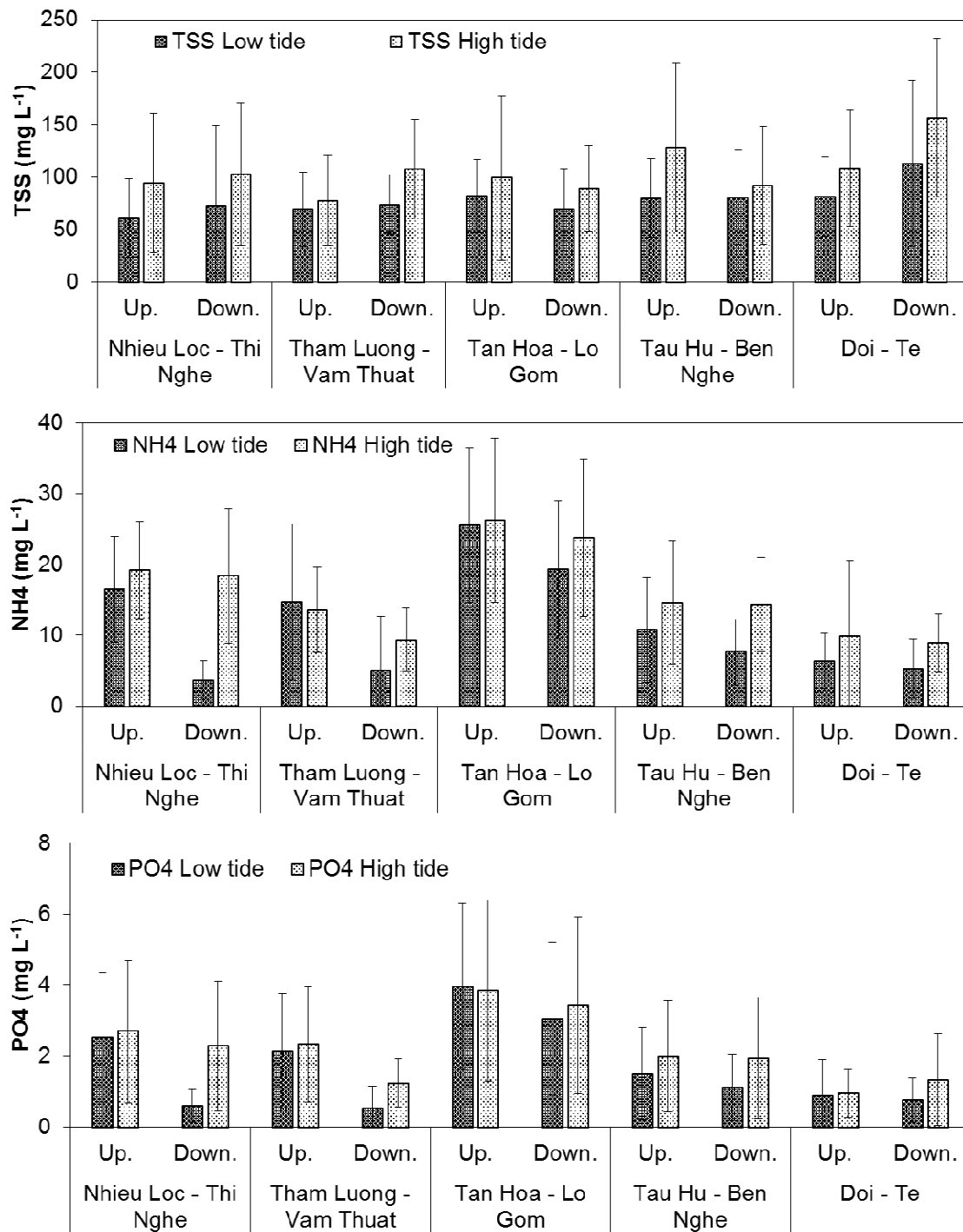


Figure A-21. Mean TSS, NH_4^+ and PO_4^{3-} concentrations in low tide and high tide over the period 2012-2016 in urban canals of HCMC (DONRE Database).