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Hydrodynamic and water quality modelling in Cartagena Bay, Colombia, as a method of target setting for policy on land-based discharges into the coastal zone

Modelación hidrodinámica y de calidad de agua en la Bahía Cartagena, Colombia, como método de determinar metas para la política de descargas terrestres en la zona costera

Ph. D. Thesis

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Abstract

This thesis presents research on coastal water processes and the development of tools for marine and coastal management. The research is applied to Cartagena Bay, Colombia, one of the Caribbean's hot-spots of coastal pollution, known to be impacted by land-based sources of pollution, including continental runoff, industrial effluents and domestic wastewater. Management of coastal water bodies such as Cartagena Bay requires tools to assess pollution impacts, land-based discharges and potential mitigation plans, while effective policy is needed to establish coastal water quality standards. Through four individual yet related manuscripts, the research presented in this thesis combines water and sediment quality monitoring with hydrodynamic and water quality modelling in order to conduct a diagnostic of pollution impacts, evaluate the effect of mitigation plans on coastal circulation, and develop novel methods for assessing land-based pollution sources and setting coastal water quality targets.

An exhaustive monthly monitoring program of 14 water quality parameters and an analysis of seven metals in sediments demonstrated the spatio-temporal extent and seasonal variability of water and sediment pollution in Cartagena Bay. Potential pollution impacts were evident as nearly all of the measured parameters were found at inadequate levels in comparison to recommended threshold values for marine conservation and recreational adequacy. These included total suspended solids, turbidity, biological oxygen demand, nitrate, phosphate, total phosphorus, phenol, faecal coliforms and enterococci, which were all highest in waters during the rainy season. The transitional season yielded higher temperatures and lower dissolved oxygen concentrations. Increased chlorophyll-a concentrations in the water column during dry season, when water transparency improves, suggest that primary productivity in this eutrophic system is limited by light rather than nutrients. Concentrations of mercury, cadmium, chromium, copper and nickel found in the bay's sediments were indicative of potential impacts on marine life. These findings are important to coastal management, as they justify mitigation actions and support policy development.

The task of identification and prioritization of pollution sources is essential but complicated when there are a variety of pollution sources and data is limited. This study presents a methodology for the integrated assessment of anthropogenic pollution sources in the coastal zone by estimating pollutant loads and comparing their relative contributions to receiving coastal waters. This approach combines various methods of load estimation based on monitoring, GIS analyses and previous research, while emphasizing the importance of calculating confidence intervals for each load value. Results showed that continental runoff from the Dique Canal is the principal source of sediments and phosphorus, nearby domestic wastewater is the principal source of coliforms, while significant loads of nitrogen and organic matter are contributed by continental runoff, domestic wastewater as well as nearby industrial wastewater.

Mitigation of pollution sources to date has focussed on freshwater flow from the Dique Canal. This flow has been projected to increase in future years, but there are plans to reduce it by constructing upstream hydraulic doors. Given the influence of freshwater discharge on coastal water renewal processes, a calibrated 3D hydrodynamic model (MOHID) and a coupled Lagrangian transport model were applied to assess how these upstream changes will affect the bay's hydrodynamic processes. Mean residence times of 3-6 days and flushing times of 10-20 days were

estimated for water from the Dique Canal in the bay, while mean residence times of 23-33 days and flushing times of 70-99 days were calculated for the bay's complete water volume. The variability of these water renewal times depends primarily on the canal's discharge level and secondarily on prevailing winds. An assessment of future scenarios showed that increases in freshwater runoff would result in faster water renewal in the bay, while plans to decrease freshwater discharge by constructing hydraulic doors would result in slower water renewal. It is therefore imperative that any plans for reducing fluvial fluxes into the bay be accompanied by the control of local pollution sources, which are abundant and could worsen the bay's water quality should water renewal times become longer.

Effective management of pollution sources requires policy on coastal water quality that considers ecologically-relevant thresholds and has a scientific foundation linking land-based discharges with marine water quality. This study demonstrates a practical method for setting local-scale coastal water quality targets for end-of-river suspended sediment loads in order to mitigate offshore coral reef turbidity impacts. The approach considers reef thresholds for suspended solid concentration and uses monitoring data to calibrate and apply a coupled 3D hydrodynamic-water quality model (MOHID) to link the marine thresholds to fluvial loads. Results showed that ecosystem suspended solids thresholds could be maintained within the extent of Cartagena Bay by reducing current suspended sediment loads in the Dique Canal by approximately 80-90% to target loads of 500-700 t/d. The substantial reductions that are needed reflect ongoing issues in the Magdalena watershed which has experienced severe erosional conditions and intense deforestation over the past four decades.

Though these studies are demonstrated in Cartagena Bay, the presented methods would be practical for application in other developing countries that similarly lack long-term datasets. The Wider Caribbean Region is strongly dependent on its natural coastal resources though pollution issues common throughout the region place these resources at risk. This thesis' approach to assessing pollution impacts, land-based sources, potential mitigation plans and developing targets for coastal water quality standards thus makes a valuable contribution to the development of coastal management tools that are applicable to similar polluted waters in the region.

Resumen

En esta tesis, se presenta una investigación sobre procesos de aguas costeras y el desarrollo de herramientas para la gestión marina y costera. La investigación es aplicada a la bahía de Cartagena, Colombia, uno de los "hot-spots" de contaminación costera en el Caribe, conocida por tener impactos de contaminación de fuentes terrestres incluyendo la escorrentía continental, efluentes industriales y aguas residuales domésticas. El manejo de cuerpos de aguas costeras como la bahía de Cartagena requiere de herramientas para evaluar los impactos de contaminación, las descargas de fuentes terrestres y planes potenciales de mitigación, mientras una política efectiva es necesaria para establecer estándares de la calidad de agua costera. A través de cuatro manuscritos, separados aunque relacionados, la investigación presentada en esta tesis combina el monitoreo de la calidad de aguas y sedimentos con la modelación de la hidrodinámica y la calidad de agua con el fin de realizar un diagnóstico de impactos de contaminación, evaluar el efecto de planes de mitigación sobre la circulación costera, y desarrollar métodos novedosos para la evaluación de fuentes terrestres de contaminación y la determinación de metas de la calidad de agua costera.

Un programa exhaustivo de monitoreo de 14 parámetros de calidad de agua y un análisis de siete metales en muestras de sedimento demostraron la extensión y variabilidad espacio-temporal de la contaminación de aguas y sedimentos en la bahía de Cartagena. Impactos potenciales de contaminación son evidentes ya que casi todos los parámetros medidos se encontraron en niveles inadecuados en comparación con los valores umbrales recomendados para la conservación marina y la adecuación recreacional. Estos incluyeron sólidos suspendidos totales, turbidez, demanda biológica de oxígeno, nitrato, fosfato, fósforo total, fenol, coliformes fecales y enterococos, los cuales presentaron niveles más altos en aguas durante la época lluviosa. La época de transición produjo temperaturas elevadas y concentraciones reducidas de oxígeno disuelto. Durante la época seca, la transparencia se mejoró y la concentración de clorofila-a aumentó en la columna de agua, sugiriendo que la productividad primaria en este sistema eutrófico es limitada por la luz en lugar de los nutrientes. En los sedimentos de la bahía, se hallaron concentraciones de mercurio, cadmio, cromo, cobre y níquel indicativas de impactos potenciales sobre la vida marina. Estos hallazgos son importantes para el manejo costero, dado que justifican acciones de mitigación y apoyan el desarrollo de política.

La tarea de identificar y priorizar las fuentes de contaminación es esencial pero complicada al haber una variedad de fuentes de contaminación e información limitada. Este estudio presenta una metodología para la evaluación integrada de fuentes de contaminación antropogénica en la zona costera a través de la aproximación de sus cargas y la comparación de sus aportes relativos a las aguas costeras receptoras. Esta estrategia combina varios métodos de la aproximación de cargas basados en monitoreo, análisis SIG e investigación previa, mientras se enfatiza la importancia de calcular intervalos de confianza para cada valor de carga. Los resultados muestran que la escorrentía continental del Canal del Dique es la fuente principal de sedimentos y fósforo, las aguas residuales domésticas son la fuente principal de coliformes, mientras que cargas significativas de nitrógeno y materia orgánica son atribuidos a la escorrentía continental, las aguas residuales domésticas y también a las aguas residuales industriales.

La mitigación de fuentes de contaminación hasta ahora se ha enfocado en el flujo de aguas dulces del Canal del Dique. Se proyecta que este flujo incremente en años futuros, pero hay planes para

reducirlo a través de la construcción de puertas hidráulicas en aguas arriba. Dada la influencia de descargas de aguas dulces sobre los procesos de renovación de aguas costeras, un modelo hidrodinámico 3D (MOHID) calibrado y asociado a un modelo de transporte lagrangiano fueron aplicados para evaluar como estos cambios en aguas arriba afectarán los procesos hidrodinámicos de la bahía. Tiempos medianos de residencia de 3-6 días y tiempos de recambio de 10-20 días fueron estimados para el agua del Canal del Dique en la bahía, mientras tiempos medianos de residencia de 23-33 días y tiempos de recambio de 70-99 días fueron calculados para el volumen completo de la bahía. La variabilidad de estos tiempos de renovación depende primariamente del nivel de descarga del canal y secundariamente de los vientos prevalentes. Una evaluación de escenarios futuros mostró que aumentos de la escorrentía de agua dulce resultarían en una renovación más rápida de las aguas de la bahía, mientras que planes para reducir la descarga de agua dulce a través de puertas hidráulicas resultarían en una renovación más lenta. Es imperativo entonces que cualquier plan para reducir flujos fluviales a la bahía se acompañe del control de fuentes locales de contaminación, que son abundantes y podrían empeorar la problemática de la calidad de agua en la bahía si los tiempos de renovación de agua se extendiesen.

El manejo efectivo de fuentes de contaminación requiere de una política para la calidad de aguas costeras que considere umbrales relevantes para el ecosistema y tenga una base científica que vincule las descargas terrestres con la calidad de agua marina. Este estudio demuestra un método práctico para determinar objetivos de calidad de agua costera a la escala local para las cargas fluviales de sedimentos suspendidos con el fin de mitigar los impactos de turbidez en los arrecifes coralinos en mar afuera. Esta estrategia considera umbrales para concentraciones de sólidos suspendidos en arrecifes y utiliza datos de monitoreo para calibrar y aplicar un modelo 3D acoplado de hidrodinámica y calidad de agua (MOHID) con el fin de vincular los umbrales marinos con las cargas fluviales. Los resultados muestran que los umbrales ecosistémicos de sólidos suspendidos podrían ser mantenidos dentro de la extensión de la bahía de Cartagena si se redujesen las cargas actuales de sedimento suspendido en el Canal del Dique en aproximadamente 80-90% hasta cargas de 500-700 t/d. Las reducciones substanciales que se necesitan reflejan una problemática en curso en la cuenca del Magdalena que ha experimentado condiciones severas de erosión y deforestación intensa a lo largo de las últimas cuatro décadas.

Aunque estos estudios son demostrados en la bahía de Cartagena, los métodos presentados serían prácticos para aplicar en otros países en desarrollo que similarmente carezcan de datos a largo plazo. La Región Caribe tiene una dependencia fuerte sobre sus recursos costeros naturales, pero problemas de contaminación son comunes a lo largo de la región, lo cual pone dichos recursos en riesgo. Los métodos de esta tesis para la evaluación de impactos de contaminación, fuentes terrestres de contaminación, planes potenciales de mitigación y el desarrollo de objetivos para estándares de calidad de agua costera hacen así un aporte valorable hacia el desarrollo de herramientas para el manejo costero que son aplicables a aguas similarmente contaminadas en la región.

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List of Acronyms and Abbreviations

3D: Three Dimensional
AcuaCar: Aguas de Cartagena
AE: Average Error
ANOVA: Analysis of Variance
BASIC: Basin Sea Interactions with Communities Project
BOD: Biological Oxygen Demand
Cardique: Corporación Autónoma Regional del Canal del Dique
CFU: Colony Forming Units
Chl-a: Chlorophyll-a
D.L.: Detection Limit
DIN: Dissolved Inorganic Nitrogen
DMA-80: Direct Mercury Analysis
DO: Dissolved Oxygen
EPA: U.S. Environmental Protection Agency
ETC: Enterococcus
EUR: European Commission
FC: Fecal Coliforms
GBR: Great Barrier Reef
GDP: Gross Domestic Product
GFS: Global Forecast System
GIS: Geographical Information Systems
GOTM: General Ocean Turbulence Model
IDRC: International Development Research Centre of Canada
L: Pollutant Load
LBS Protocol: Protocol Concerning Pollution from Land-Based Sources and Activities
MAE: Mean Absolute Error
MLLW: Mean Lower Low Water
MOHID: The MOHID Water Modelling System
MPA: Marine Protected Area
MPN: Most Probable Number
NO_3-N: Nitrate-Nitrogen
NO_x-N: Nitrate-Nitrite-Nitrogen
NOAA: National Oceanic and Atmospheric Administration
NTU: Nephelometric Turbidity Units
O_2Sat: Oxygen Saturation
PEL: Probable Effects Level
Phen: Phenols
PO_4-P: Orthophosphates
RE: Relative Error
RMSE: Root Mean Squared Error
S.D.: Sample Standard Deviation
S.M.: Standard Methods
Sal: Salinity
SOCAR: State of the Caribbean Report
TEL: Threshold Effects Levels
Temp: Temperature
TMDL: Total Maximum Daily Loads
TP: Total Phosphorus
TSS: Total Suspended Solids
Tur: Turbidity
U.S.: United States of America
WCR: Wider Caribbean Region

CHAPTER I

General Introduction

1.1 Prologue

In the year 1501 when the Spanish explorer Rodrigo de Bastidas discovered the bay of Cartagena de Indias, as it is now known, he found a natural harbour with all of the characteristics of protection, anchorage and terrestrial access needed to make it one of the Caribbean's principal ports in the centuries to come. Below the water, he would have found a pristine paradise of coral reefs, seagrass meadows and mangrove forests extending from the bay itself to the offshore horizon. What he did not know about this environment, wild and untouched in the eyes of the conquistador, was that these ecosystems had been coexisting for various millennia with some of the most ancient indigenous civilizations of the Americas (Reichel-Dolmatoff, 1997). The coexistence between humans and the bay's marine ecosystems would not continue so well in subsequent centuries. The seagrass meadows and various shallow reefs were eradicated between the years 1940-1960 with the hydraulic constructions of the Dique Canal and the ascent of the industrial era (Díaz & Gómez, 2003; Restrepo, 2008). Today, Cartagena Bay is characterized as an estuary and confluence of great quantities of freshwater, sediments, nutrients and multiple other substances which result in excess pollution.

1.2 Background and Justification

Pollution issues in Cartagena Bay are both complex and multifactorial. The bay's waters and sediments have been found to be contaminated by large contributions of freshwater, along with the sediments, nutrients, organic matter, pesticides and metals carried along with it. This is compounded by coastal discharges of domestic and industrial wastewater, among other local sources of pollution, which further contaminate the waters with pathogenic bacteria and hydrocarbons. This contributes to the degradation of marine ecosystems, not only within the bay itself but offshore as well in the Rosario Islands Marine Protected Area, and results in socioeconomic impacts on communities of people living in the coastal zone as well as Cartagena's touristic sector.

Cartagena is not only a hot-spot of coastal pollution, but also of tourism, representing Colombia's primary touristic destination, as well as economic development with a population of one million people and a growing industrial sector, the largest of its kind along the country's 3200 km coastline. Cartagena is thus a confluence of numerous stakeholders and interests which are described by four of the components of the Millenium Development Goals of the United Nations: economic prosperity, social inclusion, environmental sustainability, and adequate governance by stakeholders, including government and industry. In these terms, the justification to mitigate the bay's pollution issues thus relies on the recognition of ecosystem impacts and the multiple stakeholders impacted by these issues. These impacted parties include the marine environment, coastal human communities, artisanal fisheries and the tourist sector.

To address these issues of pollution impacts, the multidisciplinary project *Basin Sea Interactions with Communities (BASIC)* was developed and implemented between the years 2014-2017. The author of the present thesis led the preparation of the project proposal, which received a grant (#107756-001) from the International Development Research Centre (IDRC) of Canada, and coordinated the implementation of the project's administrative and operational activities in the role of project manager. The BASIC project involved five Colombian institutions (U.EAFIT, U.Andes, U.Cartagena, Cardique, Fundación HEO) that carried out research in six diverse components: 1) Research on water and sediment quality, hydrodynamic modelling and related policy in the coastal zone of Cartagena; 2) Hydrological studies of the Magdalena watershed and Dique Canal, which comprise one of the bay's principal sources of pollution; 3) Research of fish biology and ecotoxicology, to assess the impact of pollution on artisanal fisheries in nearby coastal communities; 4) An evaluation of the socio-economic impacts of pollution and fishery degradation on fishing communities; 5) Medical research of the health impacts of pollution and limited water services on coastal communities; and 6) An assessment of current management plans in Cartagena's coastal zone. While the BASIC project involved over 20 researchers, including various students (1 BSc, 2 MSc, 3 PhD, 1 Post-doc), the present PhD thesis comprises solely the work done on the project's first research component focused on water and sediment quality, hydrodynamic modelling and related policy in the coastal zone of Cartagena.

The confluence of so many forms of pollution in one place presents a formidable challenge for environmental management. An important first step in coastal management is a diagnostic of the current state of the water body in question (Cartagena Bay). Secondly, mitigation of these pollution issues clearly requires the reduction of contaminated discharges, though this entails the identification of primary pollution sources in order to orient management efforts and priorities. To be able to assimilate such a vast pollution load, the bay would require an enormous amount of circulation to drive its renewal with seawater, though water renewal time scales and their variability are not yet known, while the impact of a given mitigation plan on these time scales is important to consider. Finally, pollution control requires adequate standards for land-based discharges and receiving coastal waters, though the national policy instruments in Colombia are inadequate due to a lack of ecosystem-relevant thresholds and science-based methods of determination. These knowledge gaps present needs for effective marine and coastal management, which the present thesis resolves through scientific analyses, numerical modeling and the development of novel methodologies.

1.2.1 Environmental Characterization

Coastal and marine waters can be impacted by various types of pollution, such as sediment plumes, eutrophication, hypoxia, and unsafe swimming waters, among others. Eutrophication is a process that occurs due to the input of excess nutrients in a water body enhancing the growth of phytoplankton. Increased primary productivity in a water body has the effect of both decreasing water transparency and depleting dissolved oxygen (Welch et al., 2004). The decomposition of large algal blooms demands great amounts of oxygen, thereby reducing the water's oxygen to low (hypoxic) or almost null (anoxic) concentrations, and possibly the death of fish and other aquatic organisms (Correll 1998).

Similarly, sediment plumes also cause decreased water transparency, reducing the amount of light reaching the marine substrate, which is known to inhibit growth in coral reef ecosystems (Carricart-Ganivet & Merino, 2001). Sediments also have the added harm of smothering corals and creating soft substrate on which coral larvae cannot settle (Aller & Dodge, 1974; Cortes & Risk, 1985; Gilmour, 1999). Recurrent sedimentation favours long branching corals which are not smothered, but rather are susceptible to harm during large storm surges (Rogers, 1990). Excess nutrients may also disrupt the symbiosis between corals and their symbiont algae, zooxanthellae, and reduce reproduction (Ferrier-Pages et al., 2001). Chronic sedimentation and eutrophication typically results in an ecosystem shift from coral reefs to macroalgal dominance. In nutrient enriched waters, these algae grow taller and faster than corals, shading and out-competing them for light. Macroalgae also have an advantage over corals in regions of high sedimentation as macroalgae do not require hard substrate to settle (Fabricius, 2005).

To mitigate marine pollution impacts, such as eutrophication and various other types, a characterization of marine water and sediment quality is required. While many such studies have been carried out in Cartagena Bay, the lack of information characterizing land-based discharges and the absence of continuous coastal water and sediment quality monitoring have resulted in wide speculation regarding the pollution sources and processes causing this marine ecosystem degradation. Given the importance that this coastal zone's environmental quality holds on Cartagena's tourism, fisheries and marine life, it is essential that local management understands the current state of the marine environment.

Water quality characterizations are often applied to assess a water body's adequacy for a certain use as defined by relevant policies, such as recreational bathing or ecosystem protection (UN-Water, 2015). Such assessments require monitoring programs with frequencies relevant to the medium's variability; the high variability of water quality requires at least a frequency of monthly sampling (Karydis & Kitsiou, 2013), while the lesser variability of sediment quality may be assessed with a seasonal frequency (bi- or tri-annual). Many studies have undertaken monthly monitoring of seawater quality (e.g. Tomascik & Sander, 1985; Arhonditsis et al., 2000; Dikou & Van Woesik, 2006). Such studies may include a wide spectrum of parameters or may focus on specific parameters depending on the study area. If the priority is on marine ecosystem conservation, a great variety of parameters could be included, such as nutrients (nitrogen and phosphorus) and chlorophyll-a for the study of eutrophication (Lapointe, 1997; Newton et al. 2003) along with dissolved oxygen and biological oxygen demand (BOD) for studies of hypoxia (Diaz & Rosenberg, 1995). These parameters, along with temperature (Hoegh-Guldberg, 1999) and suspended sediments (Rogers, 1990), are of particular importance to studies of coral reefs due to their high sensitivity to environmental conditions. If the management priority is on beach recreation, microbiological parameters such as fecal coliforms and enterococcus serve as indicators of the water's sanitary conditions for bathers (WHO, 2003; Tosic et al., 2013). Similarly, sediment quality may be assessed for a wide variety of parameters as well, among which the analysis of heavy metals is particularly important for measurement in the ecosystem (Cesar et al. 2007; Casado-Martinez et al. 2009) and the potential for associated health risks to consumers of seafood (Buchman, 2008).

Previous characterizations of the water and sediment quality of Cartagena Bay have been done on numerous occasions. A biannual, surface seawater quality monitoring program has been carried out since the year 2000 by the local environmental authority Cardique, showing high levels of coliforms, nutrients and turbidity in the canal and bay (INVEMAR, 2001-2015). Hypoxic conditions at the bottom of the bay have documented (Garay & Giraldo, 1997; Tuchkovenko & Lonin, 2003; Cañon et al.; 2007). High concentrations of biological oxygen demand, nutrients and chlorophyll-a have also been found in the upper layer of the bay (Tuchkovenko & Lonin, 2003; Cañon et al., 2007).

The bay's surface sediments have also been shown to contain high concentrations of various metals (Parra et al., 2011a). The most studied metal in this coastal area is undoubtedly mercury, which has been found at high concentrations in the sediments (FAO & CCO, 1978; Guerrero et al., 1995; Alonso et al., 2000; Cogua et al., 2012; Parra et al., 2011a, 2011b), marine organisms (Alonso et al., 2000; Olivero-Verbel et al., 2008, 2009; Cogua et al., 2011), coastal birds (Olivero-Verbel et al., 2013) and nearby human populations (Olivero-Verbel et al., 2008). Furthermore, high levels of hydrocarbons have also been reported in the sediments and marine organisms (Garay, 1983: Parga-Lozano et al., 2002; Johnson-Restrepo et al., 2008), while pesticides have been found in the waters, sediments, and biota of Cartagena Bay (Castro, 1997; INVEMAR, 2009; Jaramillo-Colorado et al., 2015).

These impacts have been associated with the deterioration of marine ecosystems in the bay and in the adjacent Rosario Islands. The bay's benthic ecosystem was previously composed of seagrass beds that drastically declined decades ago (Díaz & Gómez, 2003; Restrepo et al., 2006). In the adjacent Rosario Islands, seagrass beds, coral reefs and benthic suspension feeding invertebrates have all declined in association with turbid sediment plumes (Restrepo et al., 2006, 2016b). Meanwhile the bay's previously reported hypoxic conditions likely inhibit benthic organisms and fish populations, which have also been reduced drastically in recent decades (personal communication with local fishermen).

Despite multiple studies of these wide-scale impacts, previous characterizations of the bay's water and sediment quality have not adequately informed or motivated coastal zone management towards mitigation solutions. This may be due to the uncertainty of previous characterizations or the complexity of the bay's multi-factorial pollution sources and impacts. Regardless, there is a need for an updated assessment of the bay's water and sediment quality, with a wide-range of parameters covering the various types of pollution and a sampling scheme that adequately accounts for spatio-temporal variability.

1.2.2 Pollution Sources

While a diagnostic of marine water and sediment quality is required to understand the types of prevalent pollution impacting the marine ecosystem, there are many sources that could contribute each pollutant type. For example, nutrient enrichment of freshwater and marine environments has been directly linked to land-based sources such as urban and agricultural areas (Lapointe & Clark, 1992; Lapointe & Matzie, 1996; Cloern, 2001; Lapointe et al., 2004). These anthropogenic areas include both point-source and non-point-sources of pollution, requiring multi-faceted pollution control (Lamb, 1985). Increased turbidity could be caused, on the one hand, by algal blooms due to excess nutrient input, or on the other hand, by fluvial sediment discharge coming from the upstream watershed (Boyd, 2000).

Depending on its size, a watershed can contain a myriad of anthropogenic activities that affect downstream water quality. Surface runoff is chemically altered by surface materials, while subsurface drainage water is affected by the composition of soils and substratum. Agricultural lands involve the addition of commercial fertilizers and manure to the soil, cultivation of the soil itself, and the presence of animal waste in animal farms or pastures. Without adequate land management practices, agricultural runoff has the potential to deliver large quantities of sediments and nutrients to downstream receiving waters (Moody, 1990; Brookes et al., 1996). High-density grazing lands can also be problematic if the animals have access to waterways (Brodie & Mitchell, 2005). Urban areas generate high amounts of runoff due to their impermeable surfaces while wastewater from these areas often carry the risks of harmful bacteria and nutrients (Sansalone & Buchberger, 1997). On a global scale, nitrogen levels in rivers are related to their watershed's population density (Peierls et al., 1991; Howarth et al., 1996). Industrial wastewater varies widely in its effect on water quality, as effluents depend on the specific type of industrial process. Land cover change, such as deforestation, has a large impact on runoff water quality as it increases runoff rates (Leitch & Harbor, 1999), while it uproots vegetation, loosens and exposes soil along with any contained nutrients which are then very susceptible to transport via runoff (Fredriksen, 1971). Mining activities not only result in land cover change, but also involve chemical processes, such as mercury amalgamation, which can further contaminate the downstream waters.

In the watershed draining to Cartagena Bay, a multitude of pollution sources can be found. These can be generally divided into four types of pollution sources: nearby domestic wastewater, nearby industrial wastewater, continental runoff, and marine sources of pollution. Domestic sources of pollution consist of wastewater discharge from the city of Cartagena and surrounding local communities, along with some commercial activities in the city of Cartagena which also discharge to the city's sewage system. Industrial sources of pollution consist of discharges from the city's industrial zone, called the "Mamonal", which includes at least 50 different industries. Runoff from the Dique Canal consists of freshwater discharged by the Magdalena River, which carries with it a wide-range of potential pollutants from domestic, industrial, agricultural and mining activities found throughout the Magdalena River watershed. Finally, marine sources of pollution consist of the impacts caused by intense maritime activity in Cartagena Bay, such as ballast water discharge or oil spills (both accidental and operational).

An assessment of pollution sources affecting a water body may be carried out with a variety of techniques, depending on the types of pollutions sources and the quantity and quality of available data. Non-point sources of pollution, such as those distributed throughout a watershed, may be characterized using land-cover maps and quantified with runoff export models (Arhonditsis, 2002), with advanced distributed watershed models, and also including ground water processes (Kim et al., 2008). However, in the coastal zone, a watershed's runoff may be treated as a single point-source characterized by the pollutant load discharged at the watershed's outlet. The runoff load may be quantified through measurements of discharge and pollutant concentrations (McPherson, et al., 2002; Restrepo et al., 2006; Tosic et al., 2009; Joo et al., 2012). Pollutant loads discharged from domestic and industrial wastewater system outlets can be similarly assessed with discharge and pollutant concentration data. In the case of domestic wastewater, pollutant loads can also be calculated based on population by using approximations of load per habitant and respective reductions dependent on wastewater treatment (Taebi & Droste, 2004; Tsuzuki, 2006). Such assessments of domestic and industrial pollutant loads have been carried out on a large scale by

UNEP's Regional Seas Program, such as in the Wider Caribbean Region (UNEP-UCR/CEP, 2010) where it was found that the highest loads-per-drainage-area of organic material, suspended solids and nutrients flow from the Southern Caribbean region, in large part due to the discharges from Colombia's Magdalena River.

These methods may be adequate to quantify the pollution load of each source and such assessments have previously been carried out for Cartagena Bay (Garay & Giraldo, 1997; Tuchkovenko & Lonin, 2003). However, as the methods and data quality used to assess each pollution source vary, there is a high amount of uncertainty in the comparison and interpretation of such results. This uncertainty can limit an analyst from reaching a clear conclusion and likewise dissuade a coastal zone management from confidentially making a decision. This may be the case in Cartagena Bay, as there is no clear consensus among different authorities regarding the primary pollution sources. As such, there is a need for approaches to pollution load assessment that include the quantification of uncertainty or variability involved in load calculations, in order to provide valid comparisons and interpretations of results.

1.2.3 Hydrodynamic Modelling

In general, a numerical model could be described as a set of equations used to resolve variables which characterize the system being studied. As these variables cannot feasibly be measured continuously over time and space, numerical models based on differential equations, with the corresponding boundary conditions, permit calculation of the time-space distribution of these variables. A wide range of such models has been developed to study and describe coastal zone hydrodynamics, which are founded on the physical principles of the movement of water (James, 2002; Lonin, 2009). These mathematical models can be used to study a bay's circulation patterns (Lonin et al., 2004), its water renewal time scales (Braunschweig et al., 2003) or other physical phenomena such as the interaction between wind, waves and tides (Kagan et al., 2001, 2003a, b). A transport model can also be used to study the movement of some dissolved pollutants that behave as passive tracers (e.g. inert, dissolved substances), though most pollutants have physical, chemical or biological properties that generate distinct processes and thus require their own set of equations in addition to the initial hydrodynamic movement. In order to integrate these multiple processes, some hydrodynamic models are coupled with physical-chemical and ecosystem models to describe more complex water quality processes such as sediment dispersion, oil spills, metal pollution and eutrophication (James, 2002; Lonin, 2009).

One such example of an integrated model is the MOHID water modelling system. It is an integrated modelling system with a number of coupled modules that describe water properties, hydrodynamics, geometry, advection-diffusion processes, atmosphere and benthos, amongst others (Mateus & Neves, 2013). The MOHID model is built on a broad foundation of scientific experience, using open-source code that permits the continuous inclusion of new developments, and has been applied by researchers and professionals in dozens of studies. Among its applications, MOHID has been used to study hydrocarbon dispersion (Janeiro et al., 2008) and to simulate scenarios of nutrient loading impacts on coastal estuaries where water circulation and contaminant transport dictate the water body's capacity to receive pollution (Saraiva et al., 2007).

In Cartagena Bay, water circulation and contaminant transport have previously been studied for an improved understanding of pollution issues (Pagliardini et al., 1982; Andrade et al., 1988; Urbano et al., 1992). When combined synergistically with hydrodynamic and transport models, such observational-based studies provide important information about a system's dynamics (James, 2002). The 3-dimensional hydrodynamic model CODEGO was developed and used to study various processes in Cartagena Bay: hydrodynamics and sediment transport (Lonin, 1997a; Lonin & Tuchkovenko, 1998; Lonin et al., 2004); eutrophication and oxygen regimes (Lonin & Tuchkovenko, 1998; Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003); oil spills (Lonin, 1997b, 1999; Lonin & Parra, 2005); and pollution due to phenols, fecal coliforms, grease and fats (Lonin, 2009). Modelling research in this bay has also studied water exchange mechanisms (Molares & Mestres, 2012a; Grisales et al., 2014), water levels (Molares & Mestres, 2012b; Andrade et al., 2013, 2017), tidal dynamics (Palacio et al. 2010; Rueda et al., 2013) and sediment distribution (Restrepo JC et al., 2016).

These modelling studies succeeded in characterizing many of the bay's hydrodynamic processes. It was found that strong thermohaline stratification is generated seasonally by freshwater inflow from the Dique Canal and surface heating, which results in weak vertical exchange and reduced turbulent diffusion below the pycnocline (0-4 m depths) trapping fine sediments in the water's surface layer (Lonin et al., 2004). The horizontal advection of water through the bay's two seaward straits is likewise stratified, as the accumulation of freshwater discharged from the Dique Canal forces water to flow out to sea along the surface, while tide-driven seawater renews the bay's bottom waters mostly through the Bocachica navigation canal but also as a thin layer of underflow atop the Bocagrande seawall (Tuchkovenko & Lonin, 2003; Lonin et al., 2004; Molares & Mestres, 2012a). Flood tides tend to increase the sub-surface inflow of seawater and reduce the surface outflow, ebb tides conversely increase surface outflow and decrease sub-surface inflow, while there is a net inflow through Bocachica, a net outflow through Bocagrande, and a general increase in all flows during the high-discharge season (Grisales et al., 2014). Weak circulation in the southwest lobe of the bay results in a greater deposition of sediments in this area during the dry season when vertical mixing increases due to reduced stratification and northerly winds restrict plume dispersion (Tuchkovenko et al., 2000; Lonin et al., 2004; Restrepo JC et al., 2016). The principal factor affecting plume dispersion is discharge from the Dique Canal, due to the influence of freshwater in determining the pycnocline, the mixing layer and currents generated by elevated water surface height around its outlet (Lonin et al., 2004). The persistence of turbid plumes during most of the year causes primary production in the bay to be light-dependent, rather than nutrient-limited, resulting in a lack of productivity below the pycnocline until the dry season when the plumes subside and algal blooms occur (Lonin, 1997a; Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003). These seasonal blooms, in combination with the bay's stratification (weak vertical mixing) and morphology (small straits, deep bay), result in oxygen depletion in the bottom waters (Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003).

However, no previous study has quantified time scales of water renewal in Cartagena Bay, though some authors have recommended it (Gómez et al., 2009; Grisales et al., 2014). Water renewal time scales can effectively reflect a water body's capacity to receive pollution and provide useful information for coastal monitoring and management (Karydis & Kitsiou, 2013). Semi-enclosed coastal water bodies, such as Cartagena Bay, are also particularly vulnerable to changes in water renewal, because these water bodies play the important role of buffering fluxes of water, sediments, nutrients and organisms between the land and sea, and so changes to water renewal rates can alter the ecosystem's composition, sensitivity to eutrophication and ultimately its dissolved oxygen concentrations (Dettmann, 2001; Anthony et al., 2009; Newton et al., 2014).

1.2.4 Water Quality Policy

From a management point of view, one of the most important questions regarding pollution is: what is the receiving water body's capacity to receive pollution? To answer this question one needs not only an understanding of pollution inputs, but also of the receiving water body's ambient conditions, ecosystem thresholds and water renewal times. With this knowledge, policy can be developed to adequately establish marine water quality standards and land-based discharge limits or targets (UN-Water, 2015).

Threshold levels have been identified for various uses of coastal water resources, such as recreational adequacy and ecosystem protection (e.g. WHO, 2003; GBRMPA, 2010; UN-Water, 2015). The identification of these threshold levels in coastal waters is essential to determining a potential risk in the coastal environment. However, in order to reduce this risk, there is a need to control land-based discharges which are governed by distinct reference values known as maximum discharge limits or end-of-river targets. The development of these two types of reference values for public policy is often undertaken separately without knowledge of the local conditions of hydrodynamics and dispersion, which determine a water body's capacity to receive land-based discharges. Therefore, in order to ensure compliance with both marine water quality standards and maximum discharge limits, hydrodynamic and transport models are effective tools that could be incorporated in public policies to define targets for land-based discharges in the coastal zone.

Pollution control is often dictated by the policies which oblige monitoring, reporting and compliance with water quality standards for discharges and receiving waters. The method by which policy-makers determine the threshold values of water quality standards will inevitably vary. A common method used for the determination of standards is that of assembling an ad-hoc group of experts and stakeholders which convene in order to determine these values through technical discussions and negotiations. This method has previously been applied in Colombia (INVEMAR-MADS, 2011) and in the Wider Caribbean Region (UNEP-UCR/CEP, 2010), and it is considered acceptable because in many cases there is a lack of available scientific data on which to base this important determination. However, this method of an ad-hoc expert group lacks scientific transparency, which weakens the justification and effectiveness of the standards themselves. For this reason, a process for determining standards based on scientifically justified targets would be preferable if credible data is available.

Scientifically justified target setting of water quality standards could be achieved by different methods. For example, maximum discharge limits could be established based on the Best Available Technology Economically Achievable (EPA, 2018) for wastewater treatment that is economically achievable for a specific industry. This method is used in the USA and in Spain, though it has been criticized because it has no guarantee for the protection of receiving waters (Karr & Yoder, 2004). Conversely, discharge targets could also be determined based on the hydrodynamics and transport processes in the receiving water environment. Such strategies have been implemented in Australia, where established seawater quality standards of chlorophyll-a have been linked to end-of-river nitrogen load targets and to farm level management practice targets by using coastal and watershed water quality models (Brodie et al., 2009). Other studies in the Great Barrier Reef have simply assumed direct proportionality between end-of-river loads and observed average seawater turbidity in order to calculate the proportional improvement needed in seawater quality and thus the proportional reduction target for river loads (Kroon, 2012). The MOHID model has also been used to evaluate different nutrient loading scenarios and their effect on a coastal estuary's primary production (Saraiva et al., 2007).

However, large-scale studies, such those on the Great Barrier Reef, will expectedly produce results that are somewhat robust without knowledge of the nearshore mixing processes in a specific coastal zone. Modelling applications at the local scale would be particularly useful for target setting as they account for local hydrodynamic conditions. As most water quality policies are defined at the national level, there exists the risk that local dispersion processes will not allow for compliance with both the maximum discharge limits and receiving water quality standards. Furthermore, practical applications of hydrodynamic modelling would also consider mixing zones to determine at what distance from the discharge the receiving water quality standard would eventually be met. For these reasons, further research is needed on modelling for local-scale target setting of coastal water quality policy.

1.3 Hypotheses & Objectives

The thesis' **general objective** is to conduct research on the coastal water processes of Cartagena Bay towards the development of practical methodologies for the integrated assessment of land-based sources of pollution and target setting of coastal water quality standards. The **general hypothesis** of the thesis, as originally stated in its approved research proposal (2015), was as follows: Knowledge on the coastal dispersion processes of land-based discharges can be used for an improved harmonization of targets defined by policy on coastal water quality; where water quality may be defined by a spectrum of appropriate parameters (pollutants), selected by specific criteria depending on the study area. While this general hypothesis adequately encompasses the overall goal achieved in chapter 5 of the thesis, each of the following four chapters have their own specific aims, objectives, hypotheses and research questions, described as follows.

The second chapter's **aim** is to carry out a diagnostic of pollution impacts on the waters and sediments of Cartagena Bay. The **specific objective** is to assess the bay's present state of water and sediment quality, in terms of spatio-temporal distribution and in the context of relevant national and international policy. This study addresses the following **research question**: What is the current extent of water and sediment pollution in Cartagena Bay and which factors control its seasonal variability? The **hypothesis** is that the bay's water quality, defined by a spectrum of parameters (pollutants) and selected criteria (reference values), has a marked seasonal variability controlled by different environmental factors.

In chapter 3, the **aim** is to demonstrate an integrated approach for pollution load assessment with a focus on data uncertainty and variability. The **specific objective** is to develop a practical methodology for the integrated assessment of anthropogenic pollution sources affecting a coastal zone. Applied to the coastal zone of Cartagena, the study addresses the **research question** of: Which land-based sources of pollution are responsible for the issues of hypoxia, turbidity and unsanitary conditions in the waters of Cartagena Bay? The **hypothesis** is that this approach can confidently inform decision-makers on the primary pollution sources affecting a coastal zone.

The fourth chapter's **aim** is to demonstrate the effectiveness of applying a hydrodynamic modelling approach as a tool for the integrated management of coastal waters in a tropical bay. The **specific objective** of this approach is to use field monitored data and a calibrated hydrodynamic model to characterize the hydrodynamic processes of water renewal and vertical exchange in Cartagena Bay. The **hypothesis** is that hydrodynamic modelling can generate knowledge on water renewal time scales that inform decision-makers on the effectives of potential mitigation strategies. The model was applied to evaluate possible future scenarios in order to answer the following **research question**: How can upstream anthropogenic impacts on freshwater runoff affect the bay's hydrodynamic processes?

In chapter 5, the **aim** is to demonstrate a practical method for setting local-scale coastal water quality targets. The method is applied to the example of Cartagena Bay, Colombia, by setting targets for end-of-river suspended sediment loads in order to mitigate offshore reef turbidity impacts. The chapter's **specific objective** is to present a method that is: i) appropriate at the local-scale, ii) science-based, iii) ecosystem-relevant, and iv) applicable to the coastal zones of developing countries where historical datasets are unlikely to be available. The chapter's specific **hypothesis** is that marine suspended sediment concentrations in the outer limits of Cartagena Bay can be maintained below coral reef ecosystem thresholds by reducing fluvial suspended sediment loads in the bay's principle freshwater source, the Dique Canal. The specific **research question** thus asks: What fluvial suspended sediment load is needed to effectively ensure that the selected marine ecosystem target is not exceeded?

1.4 Critical Analysis of the Background

1.4.1 Water and Sediment Quality Assessments

While there have been various previous studies of water and sediment quality in Cartagena Bay, these studies have either been focused only on specific individual parameters, or have only had a limited spatial distribution of sampling stations and lower monitoring frequencies. The local environmental authority Cardique has conducted a biannual surface seawater quality monitoring program since the year 2000, showing high levels of coliforms, nutrients and turbidity in the canal and bay (Cardique, 2000-2012). While this monitoring program does include a wide range of parameters, the low sampling frequency of twice a year is insufficient to adequately reflect seasonal conditions or perform statistical analyses of temporal variability. Garay & Giraldo (1997), Tuchkovenko & Lonin (2003), and Cañón et al. (2007) previously documented hypoxic conditions at the bottom of the bay, along with high levels of biological oxygen demand, nutrients, and chlorophyll-a. However, these studies were either focused on specific parts of the bay or were limited by a small number of measurements. While mercury contamination has been a recurrent topic of study in the bay's sediments (Alonso et al. 2000; Cogua et al., 2011; Parra et al., 2011), the latter study of Parra et al (2011) has been the only study that analyzed various different types of heavy metals other than mercury, though it was based on a single sampling session which cannot account for temporal variability.

As a result, environmental authorities in the coastal zone of Cartagena have not had access to reliable data in order to understand the spatio-temporal extents of pollution impacts and identify potential sources of pollution (Restrepo, 2008). Furthermore, consideration of appropriate reference values is essential for an adequate assessment of water and sediment quality, though previous studies in Cartagena Bay have limited such comparisons to the reference values of Colombia's national legislation (MinSalud, 1984), which is both outdated and includes very few parameters (oxygen, pH and fecal coliforms). Additionally, coastal ecosystem processes in the Colombian Caribbean are poorly understood due to insufficient data on pre-disturbance water quality or habitat status, a lack of data from undisturbed sites and inadequacies in the measurement of water quality parameters (Restrepo JD et al., 2016). The aforementioned limitations have made it difficult for environmental managers in Cartagena to understand the overall pollution issues in the bay, which are both multi-factorial and exhibit pronounced spatio-temporal variation, as demonstrated in the following chapter.

1.4.2 Pollution Source Assessments

While it is accepted by local environmental managers, academia and the public that Cartagena Bay's pollution issues are due to land-based sources, there are so many different sources with such limited data that the identification of the principal sources remains a topic governed by speculation and controversy. These pollution sources include domestic and industrial wastewater along with continental runoff from the Magdalena Watershed discharged via the Dique Canal and marine sources of pollution. Unfortunately, reliable knowledge on the relative pollutant loads of each source is non-existent, in large part due to a lack of data and methods that include estimates of uncertainty and variability needed to compare pollutant loads, which is a common limitation in the Caribbean Region (UNEP-UCR/CEP, 2010).

There have been studies quantifying the suspended sediment load discharged from the Dique Canal (Restrepo et al. 2006) though in terms of other parameters, such as organic material, coliforms and nutrients, the only information available is that of the local environmental authority's biannual water quality monitoring program (Cardique, 2000-2012) and other few unpublished studies (e.g. UniNorte & Cormagdalena, 2004). There is also little information regarding domestic discharges, though unofficial reports state that between 80-95% of the city's urban area is currently serviced by the sewage system. Adding to that a prominent industrial zone with at least 50 different industries on which there is little to no available data, and the "hot-spot" nature of Cartagena Bay's pollution issues become both evident and complex.

There have been very few attempts to quantitatively assess the pollution loads entering Cartagena Bay (Garay & Giraldo, 1997; Tuchkovenko & Lonin, 2003; Ramírez et al., 2006). However, the comparison and interpretation of their results are limited as these studies did not report estimates of the uncertainty or variability involved in their load calculations, without which a decision-maker cannot confidently reach a conclusion. Chapter 3 of this thesis proposes an integrated approach for pollution load assessments with a focus on data uncertainty and variability, through which environmental managers can confidently identify the coastal zone's primary pollution sources.

1.4.3 Hydrodynamic Modelling

As described in section 1.2.3, there has been a large mass of research that has developed many aspects of Cartagena Bay's hydrodynamics and water quality modelling. However, there have been no studies of the bay's water renewal times which are important when considering mitigation strategies that could alter a water body's hydrodynamics (Lee & Park, 2003). An ongoing hydraulic intervention in Colombia plans to construct hydraulic doors along the Dique Canal and thus reduce flows of freshwater and pollution into Cartagena Bay (Fondo Adaptación, 2018). While this intervention has been in discussion for at least 20 years and has motivated numerous modelling studies (Lonin & Tuchkovenko, 1998; Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003), none have focused on the important question of assessing the effect that reduced discharge could potentially have on the bay's water renewal time.

Furthermore, no previous study has modelled scenarios in the bay of increased freshwater discharge from the Dique Canal. Studies of the Magdalena watershed show significant increases in streamflow and sediment load of 24% and 33%, respectively, during the 2000-2010 period compared to the pre-2000 period (Restrepo et al., 2018). Based on these trends, hydrological modelling predictions show that water discharge and sediment flux from the Canal del Dique will increase by ~164% and ~260%, respectively, by the year 2020 when compared with the average discharge during the 2000-2010 period. Further increases in freshwater discharge could also be expected due to ongoing deforestation, future precipitation increases due to climate change and the intensification of the El Niño-Southern Oscillation (IPCC, 2014; Paeth et al., 2008; Restrepo et al., 2015, 2018). Therefore, these future projections should be incorporated into any assessment of mitigation strategies.

1.4.4 Water Quality Standards

Current policy in Colombia does not include targets for river outlets nor does it include marine ambient water quality standards for many important parameters, such as suspended sediments or nutrients (MinSalud, 1984). Though policy has recently been developed for point-source discharges in Colombia's coastal zone (MADS, 2018) the discharge limits of this regulation were determined without established marine water quality thresholds or consideration of nearshore dispersion processes. Therefore, the policy lacks the science-based foundation needed to ensure marine ecosystem relevance.

In other parts of the world, various methods have been applied to determine land-based discharge targets. In the USA, the Clean Water Act entails the establishment of Total Maximum Daily Loads (TMDL) which should be designed to meet a water quality standard in an impaired receiving water body (Karr & Yoder, 2004). Yet effluent limits in the USA are determined by methods, such as the Best Available Technology Economically Achievable (EPA, 2018), which are actually focused on the economic feasibility of implementing wastewater treatment rather than on the assimilative capacity of the receiving environment.

In Australia, previous methods of target setting for land-based discharges were based on pre-industrial loads (Brodie et al., 2001) or the feasibility of improved agricultural practices (Brodie et al., 2012). However, these targets did not consider the coastal zone's hydrodynamic characteristics and marine ecological thresholds, and so there is no guarantee of their effectiveness in protecting marine ecosystems (Kroon, 2012). More recent research in the Great Barrier Reef has used advanced approaches to link end-of-river and ecosystem-relevant marine water targets by incorporating ecosystem thresholds determined by long-term ecological assessments (Brodie et al., 2017). The authors established marine thresholds of photic depth for seagrass, and then used a linear relationship between marine photic depth and river phosphorus loads to determine river sediment load targets. They also established marine thresholds of chlorophyll-a for coral reefs, and then used linear relationships between marine chlorophyll-a, coastal salinity and river dissolved inorganic nitrogen loads to determine river nitrogen load targets (Wooldridge et al., 2006, 2015).

Research in the Baltic Sea coupled a river basin flux model with a marine ecosystem model to set target concentrations of nitrogen, phosphorus and chlorophyll-a in river outlets and marine waters (Schernewski et al., 2015). By modelling pre-industrial conditions, this approach established a marine water reference as 150% of pre-industrial conditions. The authors then targeted the reductions needed in river nitrogen load to comply with this marine reference.

However, while the examples mentioned in the Baltic Sea or Great Barrier Reef successfully linked water quality objectives for land-based discharges and marine waters through a scientific approach, the methods these studies used may only be appropriate for large-scale areas. More detailed methods are needed for smaller local-scales, where nearshore hydrodynamics and dispersion processes could deviate from the generalized relationships of seasonally averaged, spatially integrated approaches. Furthermore, developing countries such as Colombia and others in the Caribbean are unlikely to have long-term datasets of monthly monitoring data, such as those of the aforementioned studies.

On the other hand, some previous studies at the local-scale have quantified the link between land-based loads and receiving waters by applying hydrodynamic and water quality models. In China, modelling methods have been applied to relate river nutrients to marine chlorophyll-a (Deng et al., 2010; Han et al., 2011). These studies modelled the receiving water's response to a unit of pollution input and then used linear programming to calculate the maximum load that would maintain seawater quality below the given criteria. However, this engineering approach did not consider ecosystem relevance as it sought to maximize loads and seawater quality targets were extremely relaxed. Furthermore, the vagueness of the reported methods also makes these studies unreproducible.

In Canada, a study of Hamilton Harbour in Lake Ontario successfully calibrated a water quality model to relate land-based phosphorus loads to the harbour's water quality criteria for phosphorus and chlorophyll-a (Ramin et al., 2011). This method proved successful and incorporated sophisticated techniques to analyze model uncertainty. However, the study was also based on seasonal average values which do not consider the temporal variability that in reality could result in the frequent exceedance of the water quality criteria.

Chapter 5 of this thesis proposes a practical method for setting local-scale coastal water quality targets that resolves the aforementioned limitations. The proposed method is thus designed to be: i) appropriate at the local-scale, ii) science-based, iii) ecosystem-relevant, and iv) applicable to the coastal zones of developing countries where historical datasets are unlikely to be available.

1.5 Thesis Structure

The format of this thesis is manuscript-based, in which each of the following four chapters correspond to a different manuscript. Though each of the four studies are closely related and share some of the data used, the topics of each chapter are sufficiently different to merit individual manuscripts. Each chapter's full citation is listed below. The manuscripts are presented identically to the publications with slight changes in formatting as well as the addition of final sections describing the connection of one chapter to the next. All bibliographic references made throughout the thesis are summarized into a single bibliography section at the end.

> **1st manuscript (Chapter II)**: Tosic M, JD Restrepo, S Lonin, A Izquierdo, F Martins. 2017. Water and sediment quality in Cartagena Bay, Colombia: Seasonal variability and potential impacts of pollution. Estuarine Coastal and Self Science. Accepted 9 Aug. 2017. https://doi.org/10.1016/j.ecss.2017.08.013
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> **2nd manuscript (Chapter III)**: Tosic M, JD Restrepo, A Izquierdo, S Lonin, F Martins, R Escobar. 2018. An integrated approach for the assessment of land-based pollution loads in the coastal zone demonstrated in Cartagena Bay, Colombia. Estuarine Coastal and Self Science 211: 217-226. http://dx.doi.org/10.1016/j.ecss.2017.08.035
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> **3rd manuscript (Chapter IV)**: Tosic M, F Martins, S Lonin, A Izquierdo, JD Restrepo. 2018. Hydrodynamic modelling of a polluted tropical bay: Assessment of anthropogenic impacts on freshwater runoff and estuarine water renewal. Manuscript submitted for publication to Journal of Environmental Management, 28-Aug.-2018.
>
> **4th manuscript (Chapter V)**: Tosic M, F Martins, S Lonin, A Izquierdo, JD Restrepo. 2018. A practical method for setting coastal water quality targets: Harmonization of land-based discharge limits with marine ecosystem thresholds. Manuscript submitted for publication to Marine Policy, 4-Oct-2018.

The thesis prologue was previously published in Spanish as part of a book chapter (Tosic, 2017) which summarizes findings from the first two manuscripts. Research conducted during the thesis' literature review was also contributed to Restrepo et al. (2018) and Newton et al. (2018). Full citations are listed below:

> Tosic M. 2017. La Bahía de Cartagena: Un destino final de la contaminación en Colombia. In: Restrepo JD (ed) Arrastrando la Montaña hacia el Mar. Editorial Agenda del Mar Comunicaciones, Medellín, Colombia, pp. 56-65. ISBN 978-958-57860-8-0
>
> Restrepo JD, R Escobar, M Tosic. 2018. Fluvial fluxes from the Magdalena River into Cartagena Bay, Caribbean Colombia: Trends, future scenarios and connections with

upstream human impacts. Geomorphology 302: 92-105. http://dx.doi.org/10.1016/j.geomorph.2016.11.007

Newton A, A Brito, …, M Tosic, …, B Béjaoui. 2018. 2018. Assessing, quantifying and valuing the ecosystem services of coastal lagoons. Journal for Nature Conservation 44: 50-65. https://doi.org/10.1016/j.jnc.2018.02.009

Each of the aforementioned chapters (II-V) may be summarized as follows. In chapter 2, results are presented of following one-year of water and sediment quality monitoring in Cartagena Bay. The monitoring program integrates monthly measurements of 14 physical, chemical, and biological parameters analysed in water samples as well as the analysis of metals in sediment samples. This study presents a first look at an integrated diagnostic of the bay that covers a wide range of parameters monitored at a sufficient frequency and spatial extent to adequately reflect seasonal and spatial variability.

Chapter 3 presents an integrated approach for pollution load assessment with a focus on data uncertainty and variability. The approach combines different methods of load estimation, including effluent monitoring, spatial analyses with geographical information systems (GIS), and previously published results. In order to permit a comparison of the estimates calculated by differing methods, a novel approach is proposed for the approximation of confidence intervals for each load, considering the uncertainty and variability inherent in each value used for load calculation. The application of this approach is demonstrated in the study area of Cartagena in order to determine which land-based sources of pollution are responsible for the issues of hypoxia, turbidity and unsanitary conditions in the waters of Cartagena Bay. Using this approach, ultimately the decision-makers can confidently identify the primary pollution sources.

In chapter 4, a hydrodynamic modelling approach is applied by using field monitored data and the calibrated MOHID model to characterize the hydrodynamic processes of water renewal and vertical exchange in Cartagena Bay. The model is then applied to evaluate possible future scenarios of i) increased freshwater runoff due to ongoing tendencies in the watershed, and ii) reduced runoff due to a proposed mitigation project. By basing its modelling on an extensive monitoring dataset, this study provides local environmental authorities with reliable knowledge on the bay's hydrodynamics and water renewal processes along with a novel tool used to manage these coastal water resources.

Chapter 5 presents a practical method for setting local-scale coastal water quality targets. The method is applied to the example of Cartagena Bay, by setting targets for end-of-river suspended sediment loads in order to mitigate offshore reef turbidity impacts. By considering marine ecological thresholds and applying the coupled 3D MOHID hydrodynamic-water quality model to link the marine thresholds to fluvial loads, the presented approach is both science-based and ecosystem-relevant. The model's fine resolution accurately captures local hydrodynamic and dispersion processes, making it appropriate at the local scale, while the two years of monthly data used in this study could feasibly be collected by a developing country devoid of historical datasets. The study shows that marine suspended sediment concentrations in the outer limits of Cartagena Bay can be maintained below coral reef ecosystem thresholds by reducing fluvial suspended sediment loads in the bay's principle freshwater source, the Dique Canal, and quantifies the target loads under different seasonal conditions.

CHAPTER II

Water and sediment quality in Cartagena Bay, Colombia: Seasonal variability and potential impacts of pollution

"Whence do you have it that the terrestrial globe is so heavy? For my part, either I do not know what heaviness is, or the terrestrial globe is neither heavy nor light, as likewise all other globes of the universe. Heaviness to me (and I believe to Nature) is that innate tendency by which a body resists being moved from its natural place and by which, when forcibly removed therefrom, it spontaneously returns there. Thus a bucketful of water raised on high and set free, returns to the sea; but who will say that the same water remains heavy in the sea, when being set free there, does not move?"

- Galileo Galilei, 1624

2.1 Abstract

Cartagena Bay, one of the Caribbean's hot spots of pollution, is an estuarine system connected to the Caribbean Sea by two straits. Large freshwater discharges from the Dique Canal into the south of the bay produce estuarine conditions strongly related to the seasonal variability of runoff from the Magdalena River watershed. The bay's seasonal conditions may be characterized by three seasons: strong winds/low runoff (Jan.-April), weak winds/intermediate runoff (May-Aug.), and weak winds/high runoff (Sept.-Dec.). This coastal zone is known to be impacted by land-based sources of pollution, including continental runoff, industrial effluents and domestic wastewater. However, previous studies have not sufficiently ascertained the spatio-temporal extent of this pollution. This study addresses the following research question: What is the current extent of water and sediment pollution in Cartagena Bay and which factors control its seasonal variability? Monthly seawater samples (Sept.2014-Aug.2015) were taken from surface and bottom depths at 16 stations in and around Cartagena Bay and analyzed for physical, chemical, and biological parameters. Surface sediments were sampled from the bay's bottom every three months and analyzed for various trace metals. Seasonal variability was observed in nearly all of the water quality parameters, with higher concentrations usually coinciding with the high runoff season. Potential pollution impacts are shown by wet-season averages of total suspended solids (45.0±89.5 mg/l), turbidity (26.1±59.7 NTU), biological oxygen demand (1.20±0.91 mg/l), chlorophyll-*a* (2.47±2.17 μg/l), nitrate (171.1±112.6 μg/l), phosphate (43.1±63.5 μg/l), total phosphorus (85.3±77.2 μg/l), phenol (2.9±17.4 mg/l), faecal coliforms (798±714 MPN/100ml) and enterococci (32±30 CFU/100ml) in excess of recommended threshold values for marine conservation and recreational adequacy. The bay's hypoxic conditions are evident with low dissolved oxygen concentrations (<4 mg/l) found at bottom depths during the wet season, moderate concentrations in the windy season, and low concentrations approaching surface waters during the transitional season, showing a seasonality related to the variability of water circulation and vertical stratification. Lower chlorophyll-*a* levels found in the water column during the wet season suggest that primary productivity in this eutrophic system is not limited by nutrients, which are abundant due to land-based effluents, but rather by water transparency which is significantly reduced during the wet season due to large sediment loads discharged from the Dique Canal. Sediments from the bay's bottom were found to have concentrations of mercury, cadmium, chromium, copper and nickel in excess of the Threshold Effects Levels (TEL) used as an indicator of potential impacts on marine life.

2.2 Introduction

Achieving harmony between a sustainable natural environment and economical development presents a difficult challenge common to any growing human population. Anthropogenic development typically results in changes to the environment's water resources due to the introduction of foreign substances or excess natural substances, such as sediments and nutrients, into receiving water bodies. The capacity of a water body to assimilate these inputs is limited, which presents the risk of pollution impacts on the water resources, the aquatic ecosystem and the economies dependent on ecosystem services. This is particularly the case in the coastal zone of Cartagena, Colombia, where impacts on coastal water quality have the potential to disrupt the local economy for tourism and artisanal fisheries, and to degrade the ecosystem of the adjacent marine protected area (MPA) of the Rosario and San Bernardo Islands.

The coastal city of Cartagena, Colombia, has approximately one million inhabitants (DANE, 2017a), a large number of prominent ports and shipping operations, and the country's largest coastal industrial sector that currently includes at least 50 industries (Cardique & AGD, 2006). The historic city, its nearby beaches and MPA also represent Colombia's principal touristic destination while the surrounding coastal communities have traditionally depended on artisanal fisheries. The juxtaposition of these anthropogenic activities, some of which depend on ecosystem services while others degrade ecosystem function, is the essence of the conflicting use of natural resources in Cartagena Bay. Pollution issues in this bay are further compounded by runoff from the Dique Canal which drains 7% of Colombia's largest river, the Magdalena (Fig. 2.1; Restrepo et al., 2014), about a third of which ultimately discharges into Cartagena Bay (Restrepo et al., 2018).

Impacts on the water and sediment quality of Cartagena Bay have been documented for decades. The local environmental authority Cardique has conducted a biannual, surface seawater quality monitoring program since the year 2000, showing high levels of coliforms, nutrients and turbidity in the canal and bay (INVEMAR, 2001-2015). Garay & Giraldo (1997), Tuchkovenko & Lonin (2003), and Cañon et al. (2007) have documented hypoxic conditions at the bottom of the bay. These studies also showed high levels of biological oxygen demand and nutrients (Cañon et al., 2007), as well as high concentrations of chlorophyll-*a* in the upper layer of the bay (Tuchkovenko & Lonin, 2003; Cañon et al., 2007).

Various metals have been found in high concentrations in the bay's surface sediments (Parra et al., 2011a). Mercury is one of the metals of most interest that has been found at high levels in the sediments (FAO & CCO, 1978; Guerrero et al., 1995; Alonso et al., 2000; Cogua et al., 2012; Parra et al., 2011a, 2011b), in marine organisms (Alonso et al., 2000; Olivero-Verbel et al., 2008, 2009; Cogua et al., 2011), as well as in coastal birds (Olivero-Verbel et al., 2013) and in human populations in the surrounding fishing communities of Caño del Oro and Bocachica (Olivero-Verbel et al., 2008). Furthermore, previous research has shown high levels of hydrocarbons in sediments and marine organisms (Garay, 1983: Parga-Lozano et al., 2002; Johnson-Restrepo et al., 2008) and the presence of pesticides in the waters, sediments, and biota of Cartagena Bay (Castro, 1997; INVEMAR, 2009; Jaramillo-Colorado et al., 2015).

These impacts on the water and sediment quality of the bay are undoubtedly related to the observed deterioration of the marine ecosystems in the bay and in the adjacent Rosario Islands. Historically, the bay's benthic ecosystem used to be composed of seagrass beds which were drastically depleted decades earlier (Díaz & Gómez, 2003; Restrepo et al., 2006). The degradation of seagrass beds, coral reefs and benthic suspension feeding invertebrates in the adjacent Rosario Islands has been shown to be related with the turbid waters coming from Cartagena Bay (Restrepo et al., 2006, 2016b). Meanwhile the hypoxic conditions found at the bottom of the bay (Garay & Giraldo, 1997; Tuchkovenko & Lonin, 2003; Cañon et al., 2007) inhibit the recovery of benthic organisms in the bay, which is likely related to drastic reductions in artisanal fisheries observed in recent decades by the communities of Ararca and Caño del Oro (personal communication with fishermen; Fig. 2.1).

There are not many scientific interpretations available for the effects of fluvial, industrial and domestic inputs into the Caribbean coastal systems of Colombia. To the best of our knowledge, most studies and assessments lack sufficient sampling for an integrated analysis of water quality. Even more concerning, environmental authorities do not have access to reliable data to identify sources of pollution and the spatio-temporal extents of impacts (Restrepo, 2008). Additionally, coastal ecosystem processes under the influence of fluxes from the Andean rivers of Colombia are poorly understood due to insufficient data on pre-disturbance water quality or habitat status, a lack of data from undisturbed sites and inadequacies in the measurement of water quality parameters (Restrepo JD et al., 2016).

In this study, the following research question is addressed: What is the current extent of water and sediment pollution in Cartagena Bay and which factors control its seasonal variability? Results are presented of a one-year monitoring program integrating the monthly monitoring of 14 physical, chemical, and biological parameters analyzed in water samples as well as the analysis of metals in sediment samples. Previous studies have either been focused only on specific individual parameters, or have only had a limited spatial distribution of sampling stations and lower monitoring frequencies. The present study thus presents a first look at an integrated approach that covers a wide range of parameters monitored at a sufficient frequency and spatial extent to adequately reflect seasonal and spatial variability in Cartagena Bay.

2.3 Study Area

Cartagena Bay, located on the north coast of Colombia in the Caribbean Sea, is one of the receiving estuaries of fluvial fluxes from the main Andean catchment of Colombia, the Magdalena River basin (principal panel in Fig. 2.1). The Magdalena River is the main contributor of fluvial fluxes to the Caribbean Sea (Restrepo & Kjerfve, 2000; Restrepo, 2008) and flows to Cartagena Bay via the Dique Canal, a 114-km-long man-made distributary channel that diverges from the Magdalena River at Calamar (lower panel in Fig. 2.1). The canal has been dredged since the late 1920s and because of increased sedimentation in Cartagena Bay during the 1940s, new channels were constructed from El Dique to Barbacoas Bay. Since then, the suspended sediment load has reached and impacted the coastal ecosystems of Cartagena and Barbacoas bays, as well as the coral reefs of the Rosario Islands (Mogollón, 2013; Restrepo et al., 2006) which form Colombia's major continental coral reef system and a national marine protected area.

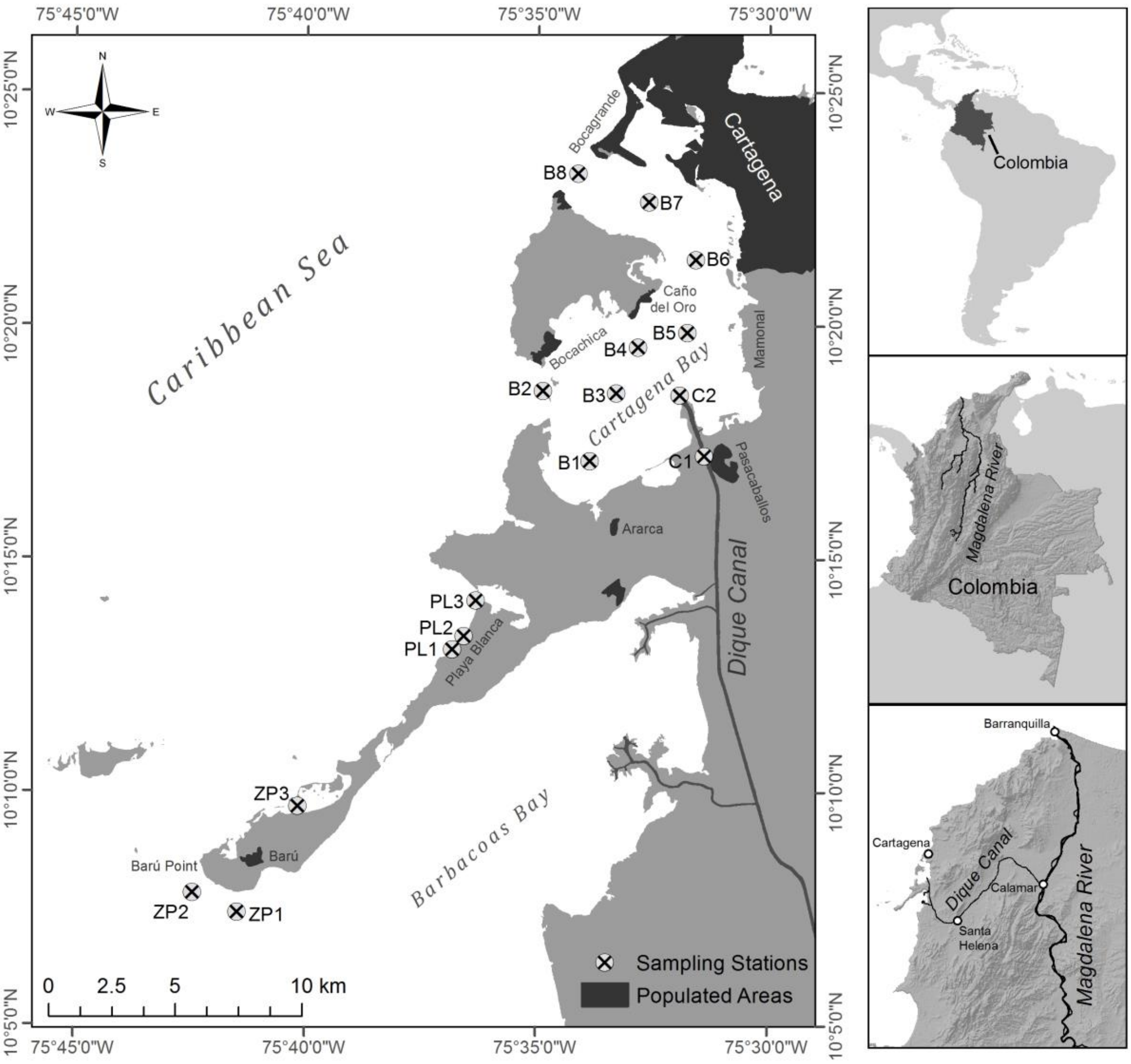


Figure 2.1. Principal panel: study area showing water and sediment quality sampling stations in three zones (see section 3.1 of text for station descriptions); Cartagena Bay, "Playa Blanca" Beach, and the fishing zone of Barú Point. Secondary panels: location of Colombia (upper panel); location of the Magdalena River (middle panel); flow of the Magdalena into the Caribbean Sea and along the Dique Canal into Cartagena Bay (lower panel).

Cartagena Bay has a surface area of approximately 84 km², an average depth of 16 m, a maximum depth of 32 m, a maximum meridian length of 16 km (N-S), and a latitudinal length of 9 km (E-W). The bay is connected to the Caribbean Sea by two straits, Bocachica to the south and Bocagrande to the north, and has a small internal bay, situated to the north and adjacent to the city center (Tuchkovenko & Lonin, 2003). Water exchange in the bay is governed by wind-driven circulation and tidal movement through its two seaward straits and the influent discharge of freshwater from the Dique Canal to the south (Molares & Mestres, 2012a). The tides in the bay are mixed, mainly diurnal with a micro-tidal range varying between 20 and 50 cm. Freshwater discharge from the Dique Canal produces estuarine conditions in the bay characterized by a highly stratified upper water column with low salinity and high turbidity at the surface.

The Dique Canal discharges approximately 55-250 m^3/s of freshwater into the bay (Tuchkovenko & Lonin, 2003), the variability of which is strongly related to the seasonality of runoff from the Magdalena River watershed. The mean annual sediment load transported by the Dique Canal between 1984 and 2010 is 6.7 Mt/y, of which 1.9 Mt/y is ultimately delivered to Cartagena Bay and studies show this load is increasing (Restrepo et al., 2018). During 26 years of monitoring, the Dique Canal has discharged approximately 177 Mt of sediment to the coast, 52 Mt of which was discharged into Cartagena Bay (Restrepo et al., 2018).

The bay's seasonal conditions may be categorized according to the variability of winds and freshwater discharge. Figure 2.2 shows multi-year average wind and discharge data from the local airport and canal gauging station, respectively. Based on daily data from 1998-2008, the canal gauging station at Santa Helena (approximately 35 km upstream of the bay) shows that the highest discharge levels occur from October to December and that the lowest levels occur from February to April.

Hourly METAR data from 1997-2015 of the Rafael Núñez International Airport (approximately 10 km north of the bay) show that from January to April the winds are strongest and predominantly northerly. This period of strong trade winds coincides with the strengthening of the southern Caribbean upwelling system which contributes to cooler water temperatures (Andrade & Barton, 2005; Rueda-Roa & Muller-Karger, 2013). From August to November, breezy conditions are observed when weaker winds come from a large range of directions, with a prominent westerly component in September and October. For the purposes of this study, months were grouped into three distinct seasons: the rainy season from Sept.-Dec., the dry/windy season from Jan.-April, and the transitional season from May-Aug.

A multitude of domestic, industrial, continental and maritime pollution sources are found in Cartagena Bay. Domestic sources of pollution include wastewater from parts of the city population not yet connected to the sewage system and about 32,500 people in the surrounding communities which discharge directly into the Dique Canal or into subterranean wells that can be susceptible to seepage or overflow during storm events. Previously, about 40% of the sewerage system (~48,000 m^3/day) was discharged directly to Cartagena Bay without treatment via an 800 m submarine outfall (UNDP-UNOPS, 1999; Tuchkovenko & Lonin, 2003), though since 2013 the city's sewage system has been routed to an outfall sufficiently far north of the city to not affect the bay. The city's industrial zone, called the "Mamonal", extends along the east coast of the bay and includes at least 50 industries which discharge directly to the bay or indirectly via small canals that ultimately discharge in the bay (UNDP-UNOPS, 1999). These activities include chemical plants, electric plants, petrochemical factories, a petroleum refinery, cement factories, aquaculture, pharmaceutical complexes, production of plastics and food processing industries (INVEMAR-MADS, 2011).

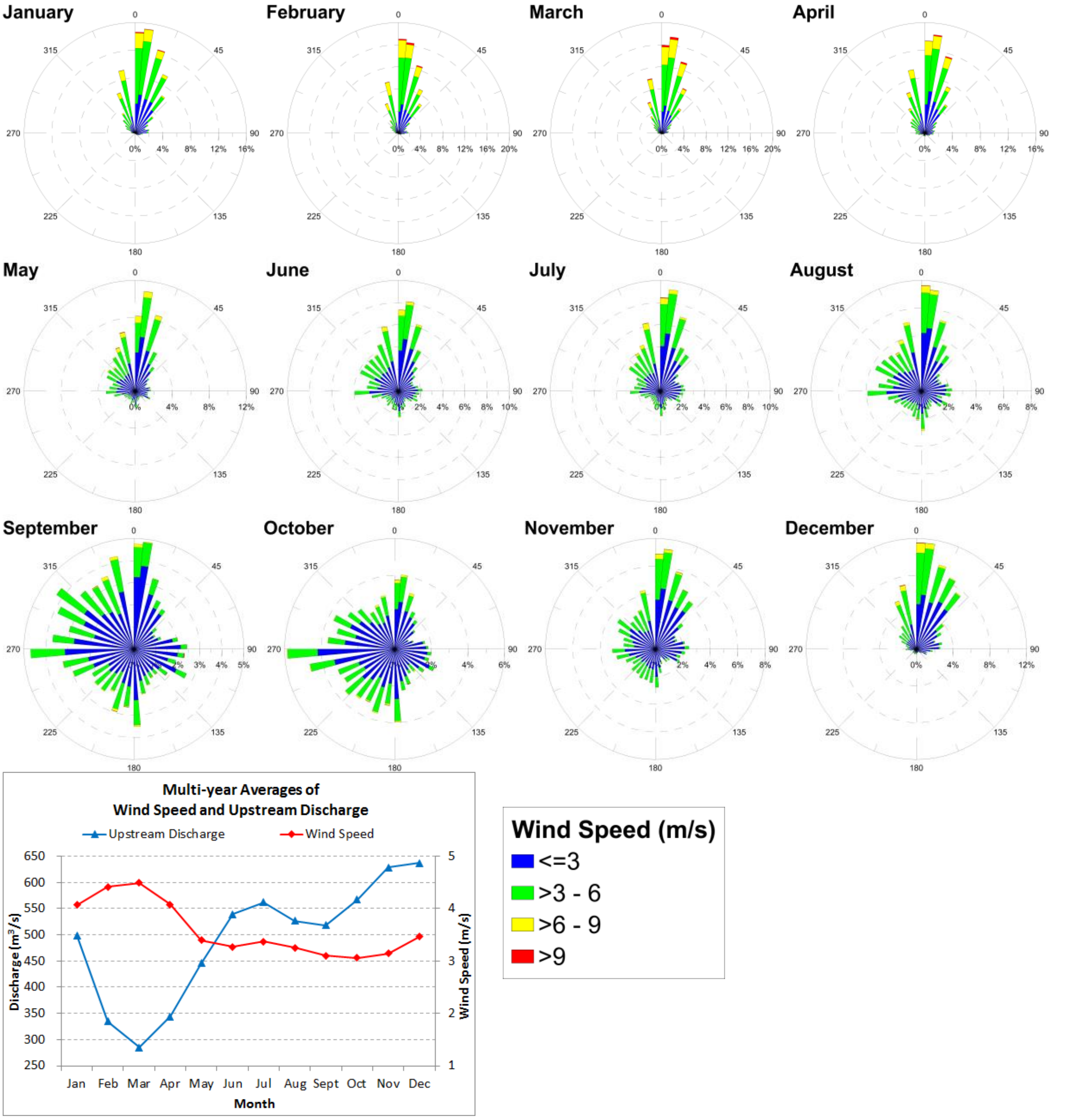


Figure 2.2: Seasonality of winds and upstream discharge. Wind speed and direction data obtained from Metar station SKCG at Rafael Núñez International Airport in Cartagena for the period 1997-2015. Discharge data obtained from IDEAM gauging station at Santa Helena (see Fig. 2.1) approximately 35 km upstream of Cartagena Bay for the period 1998-2008.

Runoff from the Dique Canal consists of turbid freshwater drained from the Magdalena River watershed. This watershed has an area of 260,000 km^2, covers approximately 25% of the country's land area, and includes approximately 80% of the national population along with a large number of industrial, agricultural and mining areas. Thus, the canal waters carry many potential pollutants along with a significant sediment load. Finally, marine sources of pollution are attributed to the bay's intense maritime activity. Previous reports cite more than 57 recreational and industrial docking areas and 5,000 dockings per year pertaining to cargo, services, and cruise ships (Garay & Giraldo, 1997; UNDP-UNOPS, 1999).

2.4 Material and Methods

2.4.1 Field Sampling

A water quality monitoring program was undertaken in and around Cartagena Bay from Sept. 2014 to Aug. 2015. Sampling was carried out between the hours of 9:00-12:00 on a monthly basis (Karydis & Kitsiou, 2013). Samples were taken from 16 stations (principal panel, Fig. 2.1), including a station in the Dique Canal (C1), another at its outlet (C2), eight stations in Cartagena Bay (B1-8), three stations at the beach "Playa Blanca" (PL1-3), and three stations in the fishing areas of Barú Point (ZP1-3). Grab samples were taken from surface waters while deeper waters were sampled with a Niskin bottle. Sediment samples were obtained with a grab sampler from the same stations in March, June, Sept. and Dec. of 2015. Additional sediment samples were also taken in Nov. 2014 for mercury analysis.

In situ analyses along the water column were done with a CTD Castaway for salinity and temperature and with a YSI Pro1020 for oxygen. Water and sediment samples were delivered to three different laboratories for same-day processing. The various water and sediment quality parameters analyzed in each sample are shown in Table 2.1 along with the laboratories, their analytical methods and detection limits. Also, shown in Table 2.1 are the depths at which measurements and samples were taken along the water column. Bottom depth was defined as 22 m for all stations except B8 (5 m) and ZP1-2 (15 m), while at shallower stations (C1, C2, B2, ZP3, PL1-3) only surface water samples were taken. To optimize monitoring of the beach stations, only *in situ* parameters, turbidity, chlorophyll-*a,* and microbiological parameters were analyzed (Tosic et al., 2013).

2.4.2 Data Analysis

Results were analyzed in comparison to two types of references values, including threshold values and background values. Threshold values consist of water and sediment quality standards defined by the Colombian legislation, international norms and scientific studies. These values indicate levels at which a parameter may pose a certain type of risk to the environment or to recreational use of the waters, and are thus used to decide whether the water or sediment presents adequate conditions for the water body's use. Background reference values consist of results found in previous studies in the area.

Statistical analyses were conducted on the water quality dataset of the bay to assess the spatio-temporal effects of the stations, seasons, and depths of samples. Only the data of three parameters displayed a normal distribution (dissolved oxygen, salinity, and temperature). A log-transformation was applied to the data of the other parameters in order to satisfy the conditions for the statistical tests. Spatio-temporal analyses of station, season and station-season interaction effects were conducted using an Analysis of Variance (ANOVA) for repeated measures in which monthly results were grouped by season: Rainy season (Sept.-Dec.), dry/windy season (Jan.-Apr.), and transitional season (May-Aug.). The effect of sampling depth was assessed using two-tailed T-tests. The results of the respective ANOVA F-tests and T-tests were compared to the 95% probability level ($p<0.05$).

Table 2.1: Water and sediment quality parameters measured in the coastal monitoring program, including analytical methods, detection limits and depths at which samples were taken in the water column. *S.M.* indicates Standard Methods; *EPA* indicates the US Environmental Protection Agency; *EUR* indicates the European Commission.

Medium	Parameter	Code	Unit	Method	Lab Methodology	Method Reference	Detection Limit	Sample Depth
Water	Temperature	Temp	°C	CTD Castaway	In Situ	-	0.01	Multiple (every 0.3m)
	Salinity	Sal				-	0.01	
	Dissolved Oxygen	DO	mg/l	YSI Pro 1020	In Situ	-	0.01	0.3, 3, 5, 10, 15, 22m
	Oxygen Saturation	O_2sat	%				0.1	
	Turbidity	Tur	NTU	Lab Cardique	Turbidometry	S.M. 2130-B	0.07	0.3m & 22m
	Total Suspended Solids	TSS	mg/l		Filtration	S.M. 2540-D	4.21	
	Biological Oxygen Demand	BOD_5	mg/l		Membrane electrode	S.M. 5210-B, 4500-O-G	0.46	
	Nitrate-Nitrogen	NO_3-N	µg/l		Colorimetry - Cd reduction	S.M. 4500-NO3-E	10.4	
	Orthophosphates	PO_4-P	µg/l		Colorimetry - ascorbic acid	S.M. 4500-PO4	26	
	Total Phosphorus	TP	µg/l		Acid digestion, Colorimetry - ascorbic acid	S.M. 4500-P-E	32	
	Chlorophyll-*a*	Chl-a	µg/l		Spectrophotometry	S.M. 10200-H	0.25	
	Phenols	Phen	mg/l		Direct photometric method	S.M 5530 D	-	Surface
	Fecal Coliforms	FC	MPN/100ml	Lab AcuaCar	Multiple tube fermentation	S.M. 9221-B	1.8	
	Enterococcus	ETC	CFU/100ml		Membrane filtration	S.M. 9230-C	1	
Sediment	Cadmium	Cd	µg/kg	Lab Universidad de Cordoba	Graphite furnace atomic absorption spectrometry	EPA 3051-A - GFAAS	25	Surface Sediments
	Chromium	Cr	mg/kg		Graphite furnace atomic absorption spectrometry	EPA 3051-A - GFAAS	0.1	
	Copper	Cu	mg/kg		Flame atomic absorption spectrometry	EPA 3051-A - FLAAS	1	
	Mercury	Hg	µg/kg		Direct Mercury Analysis (DMA-80)	EPA 7473	0.1	
	Methyl Mercury	MeHg	µg/kg		Thermal decomposition, amalgamation, and atomic absorption spectrophotometry	EUR 25830 EN - 2013	2	
	Nickel	Ni	mg/kg		Flame atomic absorption spectrometry	EPA 3051-A - FLAAS	5	
	Lead	Pb	mg/kg		Graphite furnace atomic absorption spectrometry	EPA 3051-A - GFAAS	0.1	

For visualization purposes, maps were created with water quality results that were averaged over each season and spatially interpolated. Different interpolation methods were attempted to assess the most adequate method for each dataset. For horizontal maps, data were interpolated using the method of Inverse Distance Weighting (Xu et al., 2001) with a power of three and a search radius of three. For vertical oxygen profiles, a Kriging interpolation was applied (Buzzelli et al., 2002) based on a linear model with a slope of one.

2.5 Results

The monitoring program's results are summarized as annual averages in Table 2.2 with the results of statistical analyses presented in Table 2.3. Surface water salinity in the bay ranged from 9.7 to 35.8 (average: 25.0 ± 5.3) and was significantly different from bottom waters (Table 2.3) which ranged from 32.3 to 36.3 (average: 35.7 ± 0.6). A significant seasonal effect was found in both the bay's surface and bottom waters with lower salinity occurring in the rainy season (Fig. 2.3) due to the freshwater discharge from the canal (Fig. 2.4). Surface water salinity also showed significant spatial variability with the lowest salinity found at station B5 north of the canal. Salinity inside the bay was always lower than outside the bay at Barú Point but all stations had their highest salinity values during the dry season.

Surface water temperature was significantly different from bottom waters in the bay (Table 2.3) as average surface temperature was 30.0 ± 4.9 °C while average bottom temperature was 27.8 ± 0.7 °C. Seasonal variability was significant in both surface and bottom waters. Temperature was cooler at the surface and bottom of all stations during the dry/windy season (Fig. 2.3). Spatial variation was not observed in surface nor bottom waters.

Concentrations of total suspended solids (TSS) in the bay had a range of 6-72 mg/l with an average of 20.1 ± 12.8 mg/l. A significant difference was not detected between surface waters and bottom waters (Table 2.3). TSS concentrations in the canal had a range of 60-658 mg/l, an average of 242 ± 155 mg/l, and peaks during the months of October and April coinciding with the onset of periods of increased runoff (Fig. 2.4). Seasonal variability in the bay was significant in both surface and bottom waters with higher concentrations found during the rainy season ($p<0.05$, Table 2.3). A significant spatial effect was also detected in surface waters showing higher concentrations at stations B4, B5 and B6 north of the canal's outlet which peaked at 30-72 mg/l during the rainy season (Fig. 2.3). Outside the bay, on average the stations at Barú Point had similar bottom water concentrations (20.9 ± 12.5 mg/l) and lower surface water concentrations (15.6 ± 8.2 mg/l) compared to the bay. In Figure 2.5, spatio-temporal variability in the bay can be observed as greater TSS levels are shown to occur during the rainy season to the north of the outlet, while high levels can also be found to the west of the outlet during the dry season when strong northerly winds are prevalent.

Table 2.2: Summary of environmental variables measured at different depths of the water column and in surface sediments at each monitoring station. Results are presented as annual means with standard deviation in parentheses. N_s indicates number of samples collected at each station; < D.L. indicates below detection limit. Station locations are shown in Figure 2.1; Parameter abbreviations are shown in Table 2.1.

Parameter	N_s	Station and Sample Depth (m)																								
		B1		B2	B3		B4		B5		B6		B7		B8		C1	C2	ZP1		ZP2		ZP3	PL1	PL2	PL3
		0.3	22	0.3	0.3	22	0.3	22	0.3	22	0.3	22	0.3	22	0.3	5	0.3	0.3	0.3	15	0.3	15	0.3	0.3	0.3	0.3
Water Samples																										
Temp (°C)	*12*	30.0	27.9	29.8	29.9	28.0	30.0	27.8	30.5	27.9	30.2	27.8	29.7	27.8	29.5	29.5	30.2	29.9	29.1	30.0	29.2	28.8	30.1	29.7	29.7	29.7
		(1.0)	(0.7)	(1.3)	(1.4)	(1.0)	(1.2)	(0.7)	(1.1)	(0.7)	(1.0)	(0.8)	(1.0)	(0.8)	(1.0)	(3.3)	(0.7)	(1.0)	(2.8)	(3.1)	(1.2)	(1.2)	(1.1)	(1.1)	(1.1)	(1.2)
Sal	*12*	25.3	35.2	25.6	24.6	35.1	23.9	35.3	21.0	35.9	23.7	35.9	29.1	35.8	28.5	33.3	0.1	6.6	31.4	33.9	33.6	35.4	33.6	34.5	34.5	34.6
		(5.2)	(2.3)	(5.0)	(3.9)	(2.9)	(6.1)	(2.0)	(6.0)	(0.2)	(4.0)	(0.2)	(3.7)	(0.3)	(6.3)	(3.7)	(0.1)	(6.7)	(3.4)	(2.9)	(2.1)	(0.9)	(2.1)	(1.4)	(1.4)	(1.5)
DO (mg/l)	*10*	6.4	2.7	6.7	6.9	4.5	6.8	4.6	6.6	4.3	6.7	3.8	6.4	2.9	6.6	5.6	4.8	4.8	6.1	5.6	5.9	5.7	5.8	5.8	5.9	6.0
		(1.5)	(1.1)	(1.6)	(1.9)	(1.3)	(1.7)	(1.3)	(1.6)	(1.7)	(1.2)	(1.7)	(1.3)	(1.2)	(1.5)	(1.3)	(1.0)	(1.1)	(1.7)	(1.5)	(1.4)	(1.4)	(1.2)	(1.4)	(1.4)	(1.6)
O_2sat (%)	*10*	98.0	39.4	100.6	103.9	65.0	102.4	66.4	100.1	62.4	102.7	54.8	96.3	43.8	97.7	82.43	72.7	72.3	91.3	81.0	87.4	83.9	87.4	86.8	88.2	89.2
		(13.1)	(11.0)	(10.9)	(13.9)	(10.1)	(12.1)	(10.0)	(11.6)	(20.2)	(10.8)	(19.7)	(6.6)	(18.0)	(12.7)	(8.5)	(5.9)	(9.9)	(14.3)	(6.9)	(5.6)	(5.3)	(5.1)	(9.9)	(7.2)	(10.8)
Tur (NTU)	*12*	4.8	4.2	5.0	5.7	3.5	22.6	3.0	28.9	5.5	11.4	6.7	5.4	2.6	2.8	2.5	211.2	213.7	2.8	4.1	1.3	3.8	2.4	1.7	1.9	1.3
		(1.2)	(1.8)	(1.9)	(3.6)	(1.2)	(16.8)	(0.8)	(26.0)	(4.3)	(7.1)	(9.2)	(6.0)	(1.5)	(2.2)	(1.3)	(62.2)	(105.1)	(2.9)	(3.5)	(1.0)	(4.4)	(1.5)	(1.5)	(1.2)	(0.5)
TSS (mg/l)	*12*	14.9	21.5	17.3	16.4	19.9	26.0	20.6	29.1	23.5	23.4	23.3	16.1	18.5	14.0	16.4	319.1	276.7	13.7	22.3	15.6	19.6	17.5			
		(7.9)	(18.7)	(13.9)	(7.3)	(12.8)	(15.0)	(13.5)	(16.2)	(13.8)	(18.3)	(12.8)	(7.8)	(7.1)	(7.8)	(4.9)	(406.5)	(197.3)	(8.6)	(12.3)	(8.7)	(13.1)	(7.3)			
BOD_5 (mg/l)	*12*	1.6	1.0	1.3	1.4	0.8	1.0	0.7	0.9	1.3	1.1	1.1	1.0	1.1	1.3	1.4	0.9	0.9	0.7	1.0	0.5	0.9	0.5			
		(1.4)	(0.9)	(0.8)	(1.1)	(0.6)	(0.9)	(0.6)	(0.7)	(1.0)	(0.8)	(0.8)	(0.7)	(1.0)	(0.9)	(1.1)	(0.6)	(0.8)	(0.6)	(0.9)	(0.6)	(1.0)	(0.4)			
Chl-a (µg/l)	*12*	2.7	0.8	3.0	2.7	0.9	3.1	0.8	4.2	1.1	3.8	1.2	3.2	0.9	3.3	1.6	7.5	7.2	1.9	0.8	0.9	0.8	1.0	0.6	0.6	0.6
		(1.2)	(0.3)	(1.2)	(0.9)	(0.4)	(1.1)	(0.3)	(2.1)	(0.9)	(1.9)	(1.1)	(1.3)	(0.7)	(2.2)	(1.0)	(1.3)	(1.8)	(1.9)	(0.4)	(0.7)	(0.6)	(0.4)	(0.4)	(0.4)	(0.3)
NO_3-N (µg/l)	*12*	46.2	82.1	59.2	52.6	63.9	126.4	53.0	122.4	74.9	134.6	84.2	90.8	99.3	71.4	28.1	265.9	230.6	18.9	20.5	7.5	16.4	9.8			
		(63.7)	(90.3)	(83.0)	(88.9)	(74.1)	(130.8)	(57.8)	(125.3)	(75.2)	(130.3)	(95.3)	(96.4)	(101.6)	(102.2)	(33.5)	(220.9)	(199.1)	(31.5)	(18.5)	(9.0)	(17.4)	(10.2)			
PO_4-P (µg/l)	*12*	16.6	25.0	17.4	18.3	18.5	22.8	22.8	24.1	25.9	23.6	30.0	18.8	38.4	17.4	23.9	57.5	42.5	23.6	27.8	20.5	36.7	16.6			
		(14.8)	(19.3)	(16.4)	(16.8)	(19.7)	(20.3)	(18.9)	(24.6)	(20.2)	(20.0)	(25.2)	(16.2)	(30.7)	(15.8)	(22.3)	(47.8)	(38.6)	(21.7)	(31.2)	(18.3)	(41.2)	(14.8)			
TP (µg/l)	*12*	32.2	40.0	39.7	35.7	36.4	58.4	55.0	71.7	47.5	44.2	45.0	34.2	52.5	41.3	28.7	206.8	189.2	38.8	44.7	22.8	43.5	26.2			
		(27.5)	(35.6)	(39.3)	(36.2)	(31.9)	(49.5)	(47.0)	(64.3)	(39.5)	(37.3)	(39.6)	(27.8)	(41.1)	(38.1)	(26.0)	(157.5)	(162.7)	(35.7)	(50.4)	(23.9)	(55.2)	(29.7)			
Phen (mg/l)	*12*	0.4		0.5	0.4		0.5		0.5		0.5		0.4		0.3		1.6	1.3	0.2		0.3		0.2			
		(0.4)		(0.6)	(0.3)		(0.4)		(0.8)		(0.4)		(0.4)		(0.3)		(1.9)	(1.1)	(0.1)		(0.3)		(0.4)			
FC (MPN/100ml)	*11*	43.7		126.4	193.5		191.1		573.8		419.5		229.6		236.6		1540.7	2650.6	83.2		21.5		19.6	19.5	25.6	20.6
		(68.9)		(324.3)	(379.7)		(292.5)		(869.0)		(823.0)		(456.4)		(479.9)		(1975.4)	(5167.8)	(235.6)		(39.3)		(50.1)	(25.1)	(49.0)	(49.7)
ETC (CFU/100ml)	*11*	24.2		25.6	20.1		29.0		23.5		23.0		15.5		16.3		37.0	52.5	21.6		27.5		14.5	12.9	24.7	17.8
		(29.5)		(34.3)	(27.9)		(30.2)		(23.2)		(28.2)		(20.1)		(26.5)		(41.5)	(61.9)	(21.8)		(33.3)		(13.4)	(11.5)	(31.0)	(22.9)
Sediment Samples																										
Cd (µg/kg)	*4*	191.1		43.9	331.0		400.1		540.4		272.2		149.4		21.8		1251.0	1282.6	40.0		29.3		87.4		19.5	
		(189.6)		(26.8)	(288.6)		(306.8)		(344.6)		(233.0)		(132.6)		(8.9)		(891.8)	(787.9)	(35.0)		(20.8)		(140.4)		(7.9)	
Cr (mg/kg)	*4*	44.5		5.9	42.0		50.9		54.2		59.8		52.2		17.2		38.4	42.3	33.8		24.6		8.9		3.1	
		(2.0)		(4.4)	(7.4)		(9.7)		(9.9)		(9.2)		(7.8)		(8.8)		(1.4)	(10.9)	(4.0)		(13.9)		(6.6)		(3.8)	
Cu (mg/kg)	*4*	27.4		3.1	26.0		31.0		32.5		38.6		29.4		3.4		29.9	30.3	10.3		9.2		1.9		< D.L.	
		(3.9)		(3.2)	(3.1)		(7.7)		(4.7)		(11.7)		(6.8)		(3.4)		(5.1)	(8.9)	(7.0)		(6.5)		(1.7)			
Hg (µg/kg)	*5*	117.8		24.8	136.1		116.7		351.3		141.1		164.1		20.6		97.6	83.7	52.7		41.6		32.1		5.9	
		(49.9)		(19.4)	(80.7)		(49.2)		(552.7)		(49.8)		(78.7)		(9.6)		(70.5)	(44.9)	(33.6)		(37.7)		(34.7)		(2.9)	
MeHg (µg/kg)	*4*	8.7			8.5		6.4		10.3		7.2		9.7				9.2	8.8	3.8							
		(8.9)			(9.0)		(4.6)		(9.6)		(7.3)		(6.2)				(6.9)	(8.5)	(4.2)							
Ni (mg/kg)	*4*	27.5		< D.L.	25.7		31.6		32.7		30.8		24.6		10.0		29.5	29.2	19.5		13.6		< D.L.		< D.L.	
		(4.5)			(5.4)		(7.9)		(5.6)		(5.5)		(5.1)		(3.3)		(5.7)	(6.0)	(5.6)		(7.4)					
Pb (mg/kg)	*4*	10.9		1.6	14.3		12.9		13.0		14.6		10.7		2.8		10.2	11.2	8.0		3.3		1.2		0.3	
		(7.0)		(1.2)	(2.7)		(5.2)		(5.6)		(6.5)		(6.8)		(2.6)		(6.3)	(6.6)	(4.9)		(3.7)		(0.6)		(0.2)	

Table 2.3: Results of the repeated measures ANOVA *F*-tests in terms of probabilities of significance for spatial (stations), seasonal (wet, dry, transition) and interaction (seasonal*spatial) effects in surface and bottom waters. Also shown are results of *T*-tests in terms of probabilities of significance for the effect of depth (surface vs bottom waters). Values in bold indicate significance ($p < 0.05$). Parameter abbreviations are shown in Table 2.1.

Parameter	Surface Waters			Bottom Waters			Effect of Depth (Surface vs Bottom)
	Spatial	Seasonal	Seasonal*Spatial	Spatial	Seasonal	Seasonal*Spatial	
Temp (°C)	0.2165	**< 0.0001**	0.9652	0.8310	**< 0.0001**	0.9543	**< 0.0001**
Sal	**< 0.0001**	**< 0.0001**	0.7481	0.8996	**0.0004**	0.9950	**< 0.0001**
DO (mg/l)	0.7576	**< 0.0001**	0.9340	**< 0.0001**	**< 0.0001**	0.4148	**< 0.0001**
Tur (NTU)	**< 0.0001**	**< 0.0001**	**0.0041**	**0.0011**	**< 0.0001**	0.1471	**< 0.0001**
TSS (mg/l)	**0.0042**	**0.0075**	0.0564	0.8455	**0.0002**	0.8567	0.1752
BOD_5 (mg/l)	0.6943	**0.0162**	0.3294	0.1327	**< 0.0001**	0.0862	0.1805
Chl-a (µg/l)	0.2732	0.6907	0.7891	0.5969	**0.0004**	0.7234	**< 0.0001**
NO_3-N (µg/l)	**0.0026**	**< 0.0001**	0.6164	0.1232	**< 0.0001**	0.5048	0.6524
PO_4-P (µg/l)	0.4647	**< 0.0001**	0.9714	**0.0025**	**0.0019**	0.6977	**0.0037**
TP (µg/l)	0.1120	**0.0384**	0.1066	0.8333	**0.0234**	0.5699	0.6089
Phen (mg/l)	0.9912	0.1712	0.9977	-	-	-	-
FC (MPN/100ml)	**0.0159**	**< 0.0001**	0.2466	-	-	-	-
ETC (CFU/100ml)	0.7493	**0.0476**	0.8299	-	-	-	-

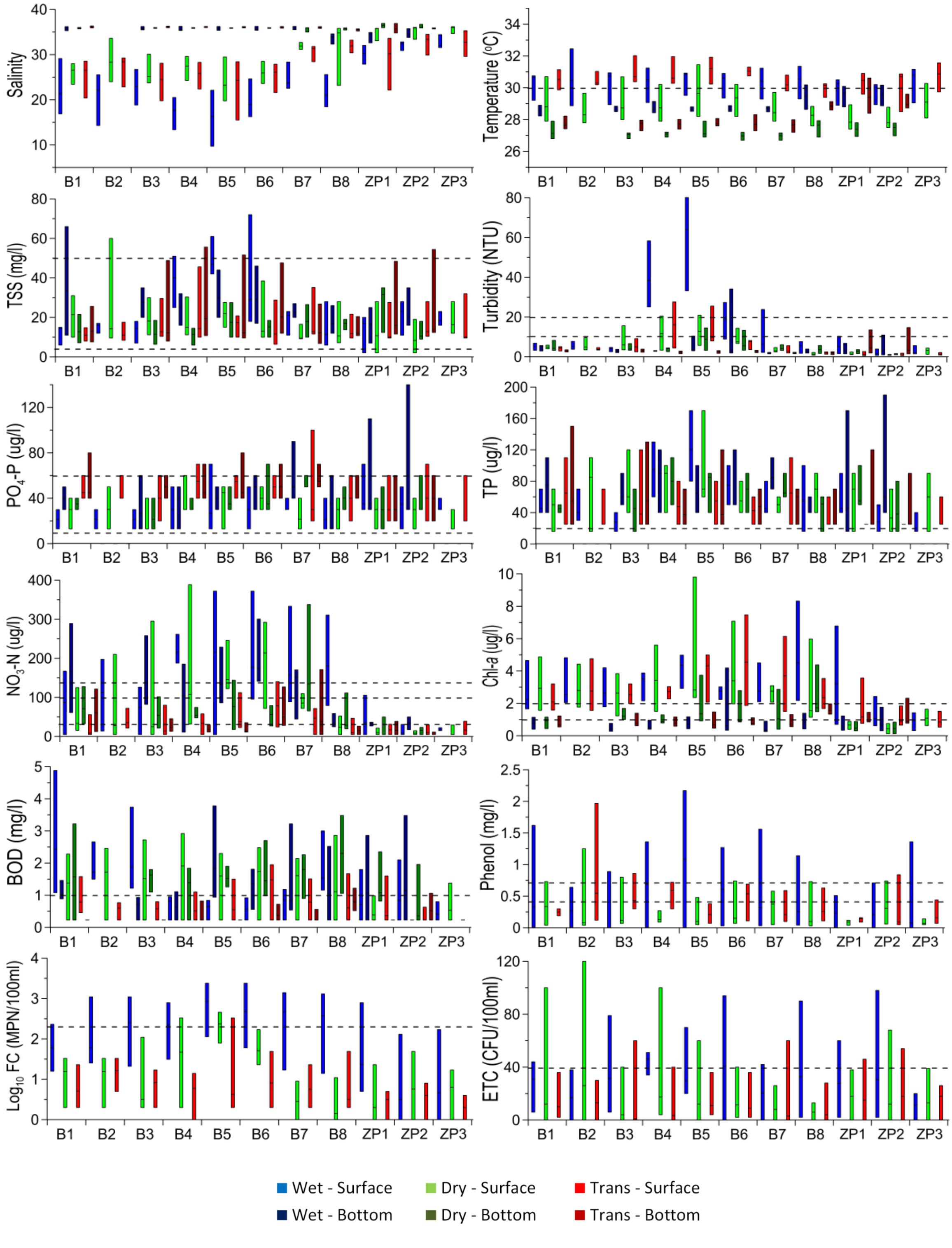


Figure 2.3: Box plots of water quality parameters measured at Cartagena Bay and Barú Point. Colours differentiate the wet, dry and transitional seasons as well as surface and bottom waters (see legend). The top of each box is at the maximum value, the bottom of each box is at the minimum value, and a line is drawn across each box at the median value. Dashed lines drawn across the plots represent threshold values cited in the text. Station locations are shown in Figure 2.1; Parameter abbreviations are shown in Table 2.1.

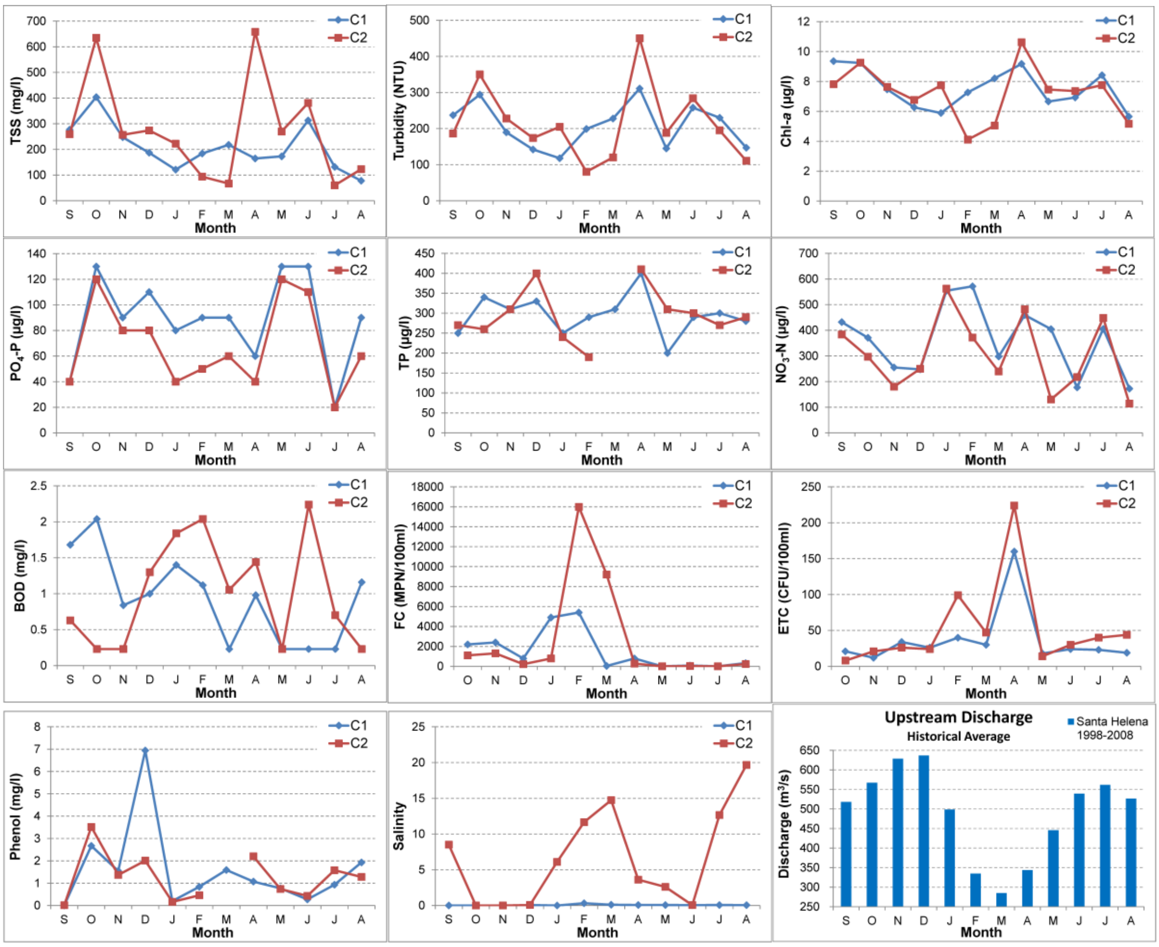


Figure 2.4: Time series plots of water quality parameters measured monthly at stations C1 and C2 of the Dique Canal. Station locations are shown in Figure 2.1; Parameter abbreviations are shown in Table 2.1. Also shown are average discharge data obtained from IDEAM gauging station at Santa Helena (see Fig. 2.1) approximately 35 km upstream of Cartagena Bay for the period 1998-2008.

The bay's turbidity levels showed much variability depending on season, station and depth. Surface waters were significantly different from bottom waters (Table 2.3) with average levels of 10.9 ± 14.4 NTU at the surface and 4.2 ± 4.4 NTU at the bottom. The canal's turbidity levels had a range of 80-450 NTU, an average of 211 ± 84 NTU, and peaks during the months of October and April coinciding with peaks of TSS and the onset of periods of increased runoff (Fig. 2.4). Significant seasonal variability was observed in both the bay's surface and bottom waters (Table 2.3). In surface waters, greater turbidity was found during the rainy season but a significant seasonal-spatial interaction effect indicates that the difference was not constant across all stations (Fig. 2.3), as stations west of the canal's outlet (B1-B3) were less affected by the rainy season, highlighting the northward trajectory of the canal's plume during this season (Fig. 2.5). In contrast, the bottom waters showed higher turbidity during the dry/windy season. Significant spatial variation was also detected in surface and bottom waters with higher surface turbidity at stations B4, B5 and B6 and higher bottom turbidity at station B5. Average turbidity outside the bay at Barú Point was similar for bottom waters (4.0 ± 3.9 NTU) and lower in surface waters (2.2 ± 2.0 NTU) compared to the bay.

Nitrate-nitrogen concentrations (NO_3-N) in the bay had a very large range of 5-389 μg/l with an average of 93 ± 88 μg/l and no significant difference between surface and bottom waters (Table 2.3). NO_3-N concentrations in the canal also had a large range of 114-572 μg/l and an average of 334 ± 139 μg/l (Fig. 2.4). A significant seasonal effect was found both in the bay's surface and bottom waters (Table 2.3) with higher NO_3-N during the rainy season and lower NO_3-N during the transitional season. Surface waters showed a significant spatial effect as well with greater NO_3-N found in the central part of the bay (stations B4-B6). In Figure 2.5, this spatio-temporal variability in the bay's surface waters can be observed as greater NO_3-N levels can be seen to occur during the rainy season and to the north of the canal's outlet. Outside the bay at Barú Point, NO_3-N was much lower with overall average concentrations of 20 ± 18 μg/l.

Concentrations of phosphate-phosphorus (PO_4-P, also known as soluble reactive phosphorus) in the study area ranged from below detection limits (< 26 μg/l) to 140 μg/l (Fig. 2.3). PO_4-P was highest in the canal with an average concentration of 78 ± 35 μg/l and peaks in October, May, and June, similar to TSS results. In and outside the bay, PO_4-P was significantly higher in bottom waters (average: 44 ± 19 μg/l) than in surface waters (average: 35 ± 16 μg/l; Table 2.3). A significant spatial effect was observed in these bottom waters with the highest concentrations found at station B7. Seasonal effects were significant both in surface and bottom waters (Table 2.3) with greater concentrations occurring during the transitional season.

No significant difference was found between total phosphorus (TP) concentrations in the surface and bottom waters of the bay which had an overall range of <32 - 140 μg/l and an average of 61 ± 32 μg/l. However, the waters off Barú Point had higher TP concentrations at bottom depths (59 ± 47 μg/l) than at the surface (39 ± 24 μg/l). TP in the canal peaked in April at 410 μg/l (Fig. 2.4) and TP in the bay was greatest at stations B4 and B5 north of the canal (Fig. 3). Though spatial effects were deemed insignificant ($p=0.112$), significant seasonal effects were found both in surface and bottom waters (Table 2.3) with greater TP levels during the rainy season.

High concentrations of chlorophyll-*a* (Chl-*a*) were found in the surface waters of Cartagena Bay throughout the year with a range of 1.14 - 9.81 μg/l and an average of 3.23 ± 1.57 μg/l. While Chl-*a* concentrations in the bay's surface waters did not exhibit seasonal or station effects, they were significantly higher than the chlorophyll-*a* in the bay's bottom waters which had a range of 0.26 - 4.18 μg/l and an average of 0.95 ± 0.69 μg/l. These bottom waters showed significant seasonal variability (Table 2.3) with Chl-*a* concentrations increasing to above 1 μg/l in the dry season (Fig. 2.6). However, temporal variability was different at stations ZP1 and ZP2 off Barú Point where Chl-*a* at the surface and bottom depths was highest in the rainy and transition seasons (Fig 2.3). Chlorophyll-*a* in the canal varied between 4.11 and 10.62 μg/l with an average of 7.39 ± 1.55 μg/l and peaks in October and April (Fig. 2.4).

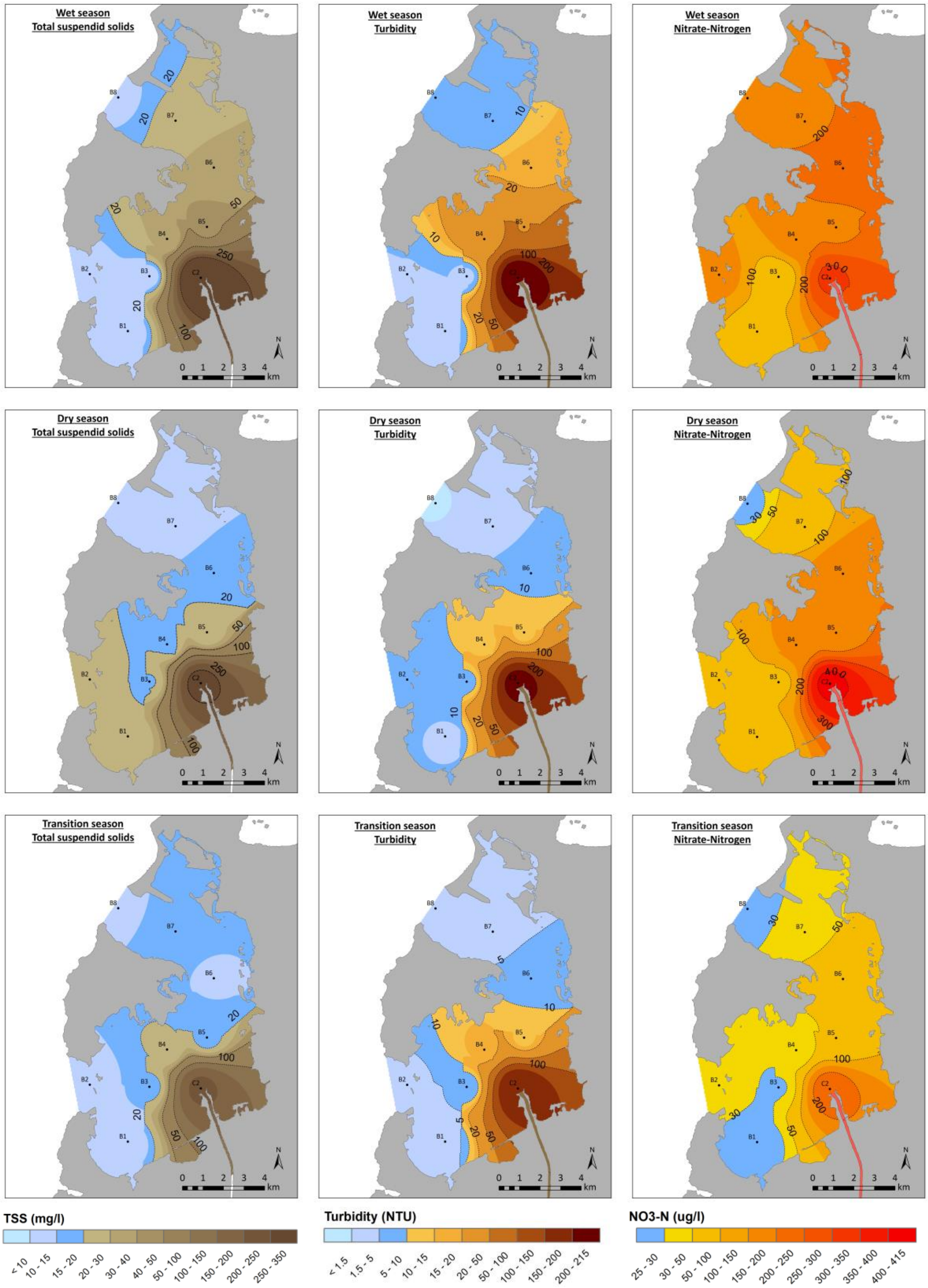


Figure 2.5: Interpolated maps of total suspended solids (mg/l; left column), turbidity (NTU: center column) and nitrate-nitrogen (µg/l; right column) measured in the surface waters of Cartagena Bay and averaged over the wet (top row), dry (middle row) and transitional season (bottom row). Blue colours represent results below the threshold values cited in text.

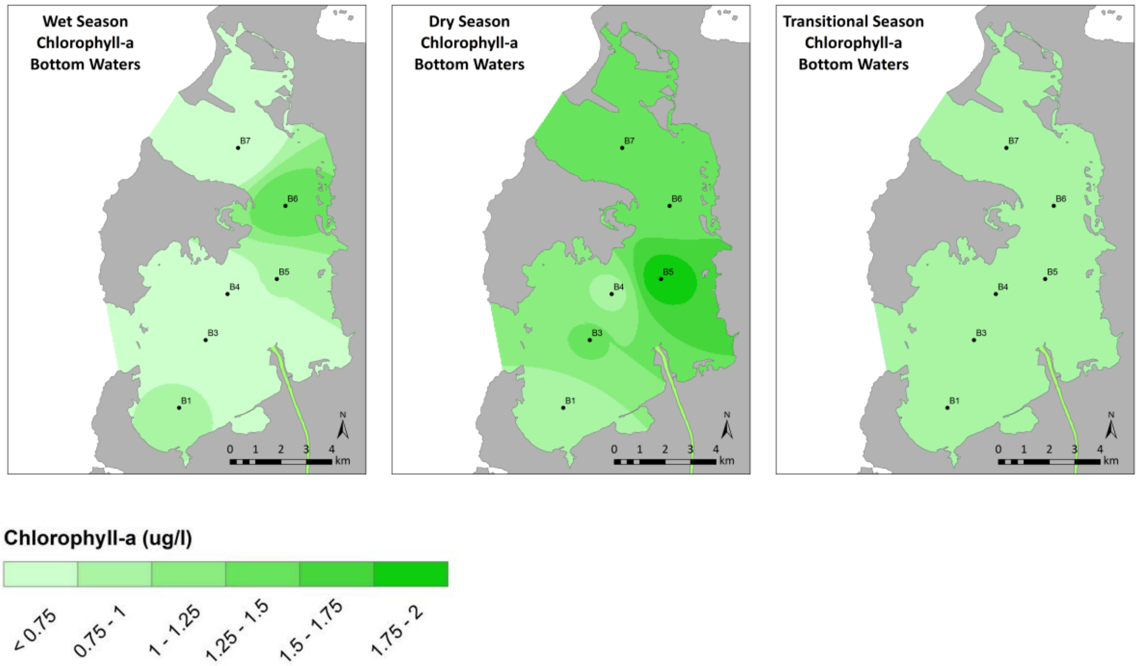


Figure 2.6: Interpolated maps of chlorophyll-*a* concentration (µg/l) measured in the bottom waters of Cartagena Bay and averaged over the wet (left), dry (center) and transitional season (right).

Concentrations of biological oxygen demand (BOD) in the bay's surface and bottom waters were not significantly different and had an overall average of 1.15 ± 0.90 mg/l. Though spatial effects were not observed in the bay, both its surface and bottom waters had significant seasonal variability with greater BOD during the dry season and lower BOD in the transition season (Fig. 2.3; Table 2.3). BOD concentrations in the canal had an average of 0.94 ± 0.69 mg/l (Fig. 2.4). Meanwhile, BOD levels were lowest in the surface waters outside the bay off Barú Point with an average of 0.55 ± 0.49 mg/l (Fig. 2.3).

The bay's hypoxic conditions are evident in the dissolved oxygen (DO) and O_2 saturation data measured along vertical profiles at each station (Fig. 2.7). The extent of the hypoxic conditions varied greatly by season, both in surface and bottom waters (Table 2.3). During the rainy season, low DO concentrations (< 4 mg/l) were found at depths greater than 10 - 20 m, while during the transitional season such concentrations could be found below depths of 5 - 10 m. During the peak of the windy season (Jan.-Feb.), DO measurements were all above 4 mg/l. Similarly, adequate O_2 saturation values (> 80%) were observed in the top 10 m of the water column during the windy season, while in other months this level of saturation was only found in the top 5 m. Bottom water DO also showed significant spatial variability (Table 2.3) with lower concentrations found at stations B1 and B7. Meanwhile, in November the lowest concentrations of DO were found in the bottom waters of stations B5 and B6 (Fig 2.5). In contrast, outside the bay at Barú Point, DO and O_2 saturation values were consistently above 4 mg/l and 80%, respectively, except in the transitional season when ranges of 3.76 - 4.69 mg/l and 68 - 92% were registered.

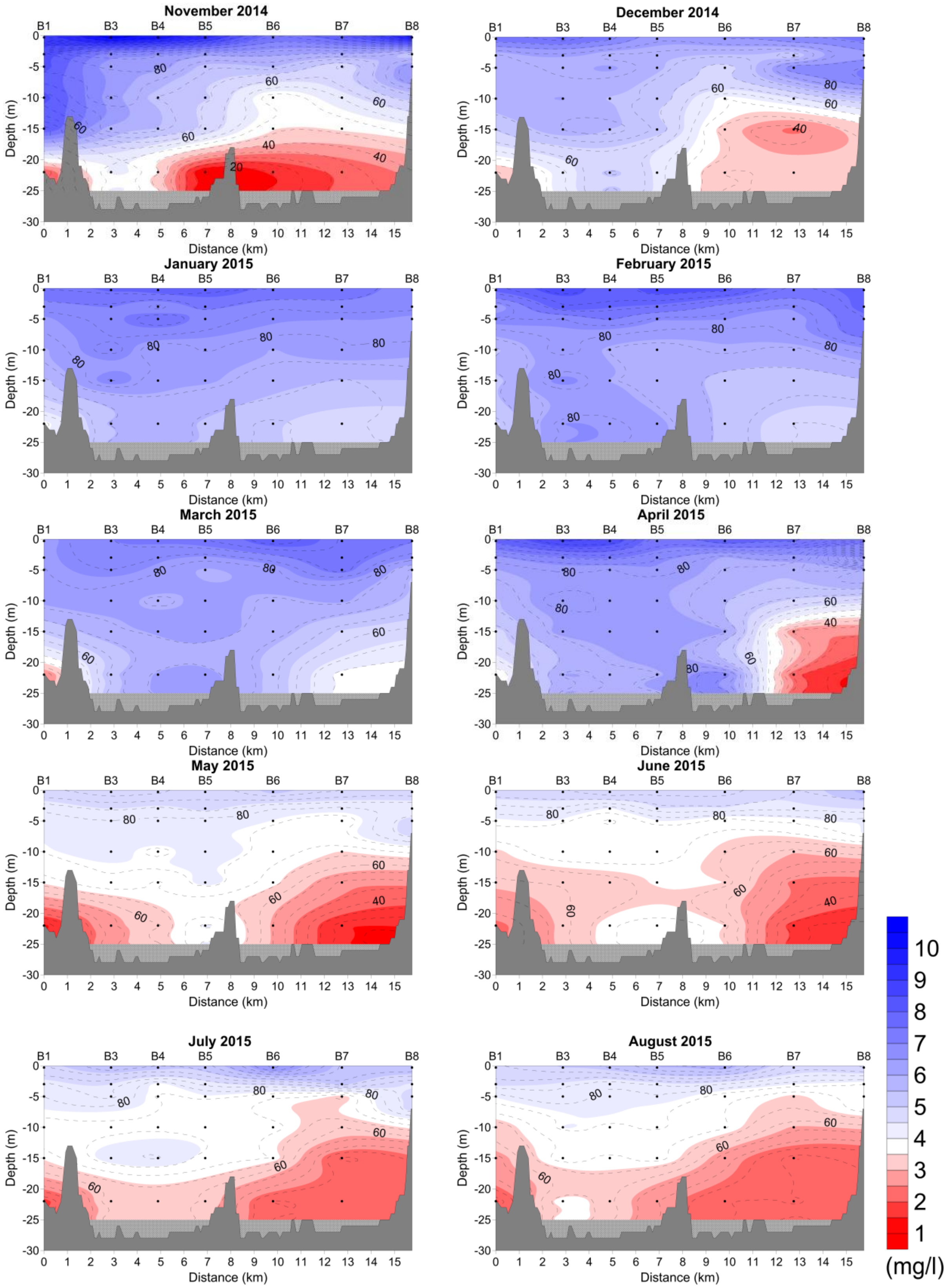


Figure 2.7: Monthly measurements of oxygen profiles in Cartagena Bay. Dissolved oxygen concentration (mg/l) is shown as a colour gradient. Oxygen saturation (%) is shown as dashed contour lines. Points (●) represent measurement locations at stations (B1, B3-B8) shown in Figure 2.1.

Results of fecal coliforms in the bay varied greatly, ranging from below detection limits (<1.8 MPN/100ml) to 2,400 MPN/100ml yielding an average value of 251.8 ± 528.4 MPN/100ml, though a median value of just 22.5 MPN/100ml (Fig. 2.3). A significant seasonal effect was observed (Table 2.3) with much higher concentrations found in the bay during the rainy season (median: 745.0 MPN/100ml) than the rest of the year (median: 9.0 MPN/100ml). While the greatest concentrations in the bay were found in the rainy season at stations B5 and B6 north of the canal, the greatest concentrations in the canal were found during the dry season (Fig. 2.4).

Enterococcus results in the bay had much less variability than those of fecal coliforms, with a range of <1-120 CFU/100ml, an average of 22 ± 27 CFU/100ml, and a median value of 14 CFU/100ml. Enterococcus concentrations in the bay were also significantly higher during the rainy season (median: 30 CFU/100ml) than the rest of the year (median: 7 CFU/100ml) and peak concentrations in the canal were also found during the dry season (Fig. 2.4). However, contrary to fecal coliform results, enterococcus concentrations in the bay did not differ greatly from those outside the bay at "Playa Blanca" beach and Barú Point which had a range of <1-98 CFU/100ml, an average of 20 ± 23 CFU/100ml, and a median value of 16 CFU/100ml.

Concentrations of phenol were found to be greatest in the canal (average: 1.4 ± 1.5 mg/l) followed by the bay (average: 0.4 ± 0.5 mg/l) and Barú Point (average: 0.2 ± 0.3 mg/l). Though spatio-temporal effects were not significant in the bay (Table 2.3), peak concentrations above 2 mg/l were observed during the rainy season at station B5 north of the canal (Fig. 2.3). Higher concentrations were found in the canal in the rainy season with a range of 1.4-6.9 mg/l (Fig. 2.4).

Concentrations of mercury (Hg) in the sediments at the bottom of Cartagena Bay had a range of 65-302 µg/kg and an average of 131 ± 55 µg/kg (Fig. 2.8). Not included in this average value is a single sample taken at station B5 in November 2014 which was analyzed in triplicate and yielded a result of 1,339 ± 113 µg/kg. Mercury concentrations were lower in the Dique Canal (average: 91 ± 56 µg/kg) and much lower in the bay's straights and Barú Point (average: 29 ± 25 µg/kg). Hg levels in the bay were greatest in November 2014 and decreased with each subsequent sampling (Fig. 2.8). Meanwhile, methyl-mercury concentrations in the bay and the canal had a range of 1.4-24.5 µg/kg and an average of 8.6 ± 6.9 µg/kg, showing that approximately 2-20% of the total mercury detected was bio-available. Methyl-mercury in the fishing zone of Barú Point was lower with an average of 3.8 ± 4.2 µg/kg.

Concentrations of cadmium (Cd) found in the Dique Canal's sediments averaged 1,267 ± 779 µg/kg. Cd levels in the bay were lower but varied greatly by season, as rainy season results had a range of 232-877 µg/kg and an average of 511 ± 208 µg/kg, while dry season results had a range of 13-244 µg/kg and an average of 60 ± 88 µg/kg (Fig. 2.8). Results found in the bay's straights and outside the bay at Barú Point were much lower throughout the year with an average concentration of 32 ± 21 µg/kg.

Results of chromium (Cr), copper (Cu), nickel (Ni) and lead (Pb) were highest in the central part of the bay (stations B4-6) and in the canal (Fig. 2.8). However, much lower concentrations were observed in the bay's straights and outside the bay. These metals did not display temporal trends, though lower concentrations were found in June 2015 for Ni and in March 2015 for Pb.

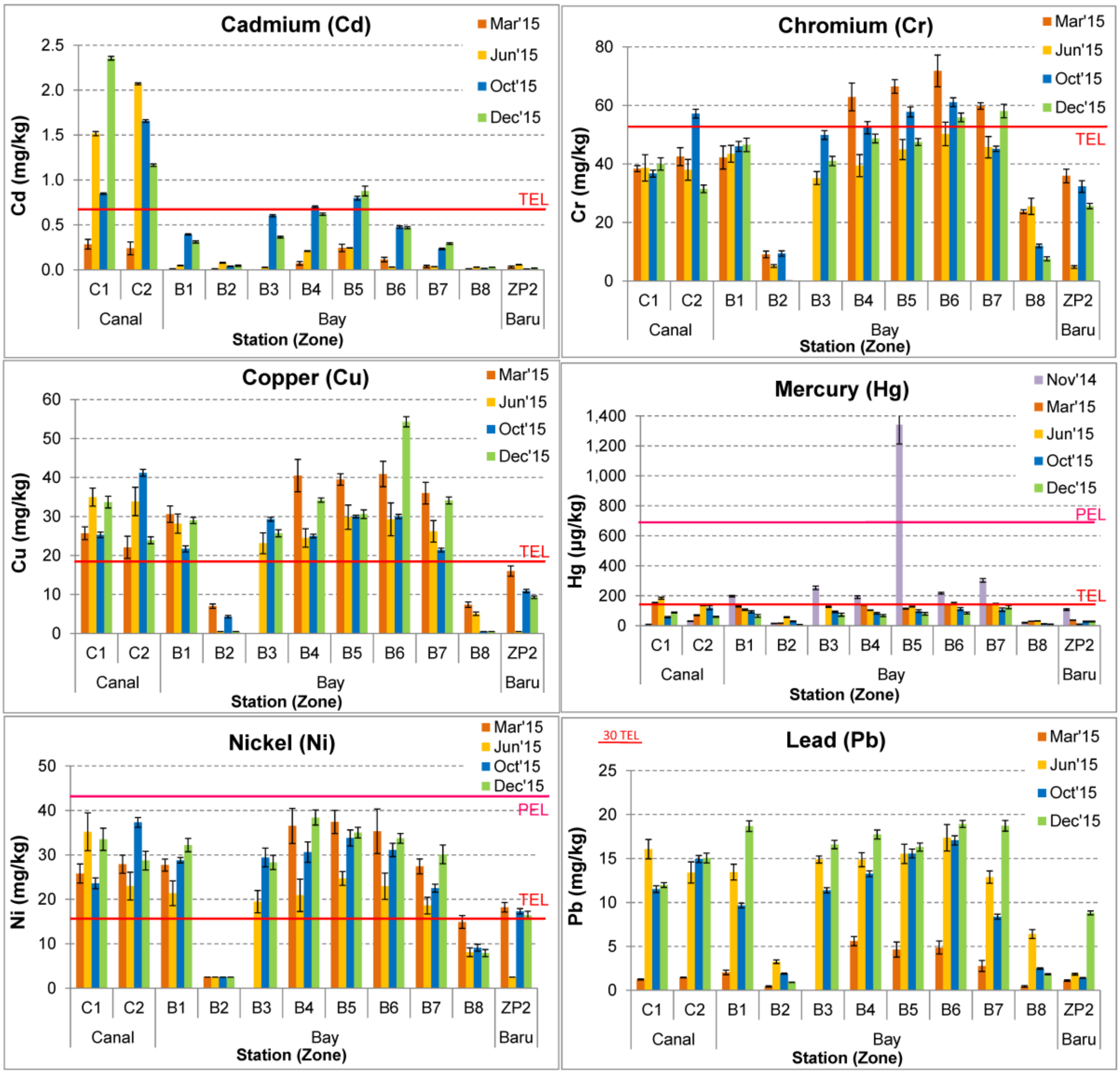


Figure 2.8: Bar graphs of metal concentrations in the sediments of the Dique Canal, Cartagena Bay and Barú Point collected in March, June, October and December 2015, as well as in November 2014 for mercury. Red lines drawn across the plots show the Threshold Effects Level (TEL) and the Probable Effects Level (PEL). Station locations are shown in Figure 2.1; Parameter abbreviations are shown in Table 2.1.

2.6 Discussion

2.6.1 Seasonal Variability

The dynamics of seasonal pollution in Cartagena Bay are apparent during the rainy season when higher concentrations of fecal coliforms, enterococcus, sediments and nutrients are found in the bay. This is likely related to upstream anthropogenic activities as peak concentrations in the Dique Canal are observed in October at the onset of high discharge season, resulting in elevated concentrations in the central part of the bay where the canal's freshwater plumes tend to disperse during this season (Lonin et al., 2004). However, given that higher concentrations of NO_3-N were consistently found at station B6 (5.4 km from the canal's outlet) than at station B5 (2.5 km from the outlet), it would suggest that the waters near station B6 are also influenced by other sources of NO_3-N in addition to the canal's plume. This is likely due to domestic wastewater coming from populations along the coast, as intense rain events can result in overflow of the city's sewage system and the latrine wells used in smaller nearby communities. Nearby sources of domestic wastewater were also apparent during the dry season when the greatest concentrations of fecal coliforms and enterococcus were observed in the canal. This may be a result of continuous wastewater discharges in the canal that receive less dilution in the dry season when the canal's discharge is lowest.

The influence of the windy season on the bay's water quality is evident in higher salinity values as the winds increase the seawater's vertical mixing. Temperature was cooler at the surface and bottom of all stations during this season due to increased circulation and the influence of the southern Caribbean upwelling system (Andrade & Barton, 2005; Rueda-Roa & Muller-Karger, 2013). Wind-driven circulation would also explain the seasonal improvement in oxygen levels, along with the associated lower water temperature, and may contribute to the resuspension of bottom sediments resulting in higher turbidity found in bottom waters during this period. The wind's influence on the dispersion patterns of the canal's plume is also observed as higher TSS and turbidity levels can be found to the west of the outlet during this season (Lonin et al., 2004).

The dry/windy season also resulted in increased chlorophyll-*a* concentrations in the bay's bottom waters, in agreement with previous findings of Cañon et al. (2007). This seasonal variability of Chl-*a* does not coincide with that of the nutrient concentrations (NO_3-N, PO_4-P, TP) suggesting that the bay's primary productivity is not limited by nutrients but rather by light (Tuchkovenko & Lonin, 2003). Water transparency in the bay is significantly reduced during the wet season by the canal's sediment load (Fig. 2.5) but once these turbid freshwater plumes decline, light penetrates deeper in the water column allowing for increased primary productivity beneath the surface. This seasonal dynamic of the bay's primary productivity may also be reflected in the higher levels of BOD and turbidity found in the bottom waters during this time.

The importance of the transitional season on coastal water quality is an aspect that has not been considered by some studies (INVEMAR, 2001-2015). Results from the transitional season in the bay showed the year's most hypoxic conditions with levels of dissolved oxygen and oxygen saturation less than 4 mg/l and 80%, respectively, reaching depths of just 5-10 m from the surface (Fig. 2.7). These hypoxic conditions were most prevalent at station B7 which may indicate sources of organic matter as this station is located near an old submarine outfall that discharged untreated sewage into the bay from 1960 until 2013.

Greater concentrations of PO_4-P were also found in the transitional season and in bottom waters. This may be related to lower TSS concentrations resulting in higher dissolved phosphorus concentrations due to phosphorus' high potential for sorption to sediment particles (Schlesinger, 1997). On the contrary, greater TP levels were observed during the rainy season, similar to TSS. This dynamic between TP, TSS and PO_4-P demonstrates how sediments can act as sources or sinks for phosphorus.

2.6.2 Threshold Value Comparison

Potential impacts of pollution were evident in Cartagena Bay as the majority of water and sediment quality parameters analyzed were found to exceed national and international threshold values. Occasional sanitary risks to recreational waters were found as the Colombian national water quality standard of 200 MPN/100ml for fecal coliforms in bathing waters (MinSalud, 1984) was exceeded in the rainy season in the bay (Fig. 2.3). Meanwhile, the World Health Organization's guideline value of 40 CFU/100ml for enterococcus in recreational waters (WHO, 2003) was occasionally exceeded in the bay and at the beach "Playa Blanca" outside the bay.

Potential impacts to the ecosystem were evident as dissolved oxygen concentrations were often found below 4 mg/l, the minimum threshold value used for preservation of marine flora and fauna in Colombia (MinSalud, 1984), and O_2 saturation values beneath the surface were persistently less than 80%, which may be considered a minimum threshold value to maintain healthy biota (Newton & Mudge, 2005). Meanwhile, average BOD concentrations in the bay were slightly above the max threshold value of 1 mg/l for fishing resources in Cuba (NC, 1999).

Threshold values for coral health cited by Fabricius (2005) were occasionally surpassed in and outside the bay in the case of TSS (50 mg/l) and PO_4-P (62 μg/l), and often exceeded in the bay in the case of dissolved inorganic nitrogen (140 μg/l). The conservative threshold values of Barbados (1998) for coral preservation were greatly exceeded in and outside the bay in the case of TSS (5 mg/l) and turbidity (1.5 NTU). Outside the bay in the bottom waters at Barú Point, salinity never dropped below 32, a minimum threshold value for corals (Hoegh-Guldberg, 1999), although temperature exceeded 30°C in Sept. 2014 presenting a risk of coral bleaching (Wilkinson & Souter, 2008; Vega et al., 2011).

The guideline values of Australia and New Zealand (ANZECC, 2000) for estuarine waters were frequently exceeded in and outside the bay in the case of nitrate-nitrite (NO_x-N: 30 μg/l), PO_4-P (5 μg/l) and TP 20 (μg/l), indicating ample nutrient enrichment. The bay's waters also surpassed ANZECC's threshold values for chlorophyll-*a* (2 μg/l) and turbidity (20 NTU), and were occasionally above phenol guideline values defined for species protection levels of 95% (0.40 mg/l) and 80% (0.72 mg/l).

Guideline values of the U.S. Environmental Protection Agency (U.S.EPA, 2015) for Caribbean waters would classify Cartagena Bay's waters as "poor" in the case of NO_3-N (> 100 μg/l), PO_4-P (> 10 μg/l) and chlorophyll-*a* (>1 μg/l), while these waters would also be classified by U.S.EPA as "poor" when compared to Gulf of Mexico waters in the case of PO_4-P (> 50 μg/l).

Many of the sediment quality results were above the Threshold Effects Level (TEL) used to indicate potential risk by the U.S. National Oceanic and Atmospheric Administration (Buchman, 2008). Such was the case of concentrations of mercury (>130 µg/kg), chromium (>52.3 mg/kg), copper (>18.7 mg/kg) and nickel (>15.9 mg/kg) in the bay and canal (Fig. 2.8). Cadmium concentrations also exceeded the TEL value (680 µg/kg) during the rainy season, clearly showing the canal to be the source. The Probable Effects Level (PEL) for mercury (700 µg/kg) was surpassed on one occasion at station B5 in Nov. 2014, while nickel concentrations were occasionally near the PEL value of 42.8 mg/kg. Concentrations of lead (Pb) were all below the TEL value of 30.2 mg/kg.

2.6.3 Comparison with Other Studies

Nearly all of the parameters analyzed in Cartagena Bay produced data ranges similar to those of previous studies (Garay & Giraldo, 1997; Lonin & Tuchkovenko, 1998; Sierra Misco, 1999; Tuchkovenko & Lonin, 2003; Lonin et al., 2004; UniNorte & Cormagdalena, 2004; Cañon et al., 2007; Lonin, 2009; Parra et al., 2011a; Cogua et al., 2012; INVEMAR, 2001-2015; Restrepo JD et al., 2016). However, there were a few exceptions such as the ranges of BOD (<0.5-4.9 mg/l), PO_4-P (<26-140 µg/l) and enterococcus (<1-120 CFU/100ml) in the present study which were similar to those reported by Tuchkovenko & Lonin (2003), Cañon et al. (2007), and INVEMAR (2001-2015) but far below the values reported by Sierra Misco (1999) of up to 100 mg/l of BOD, 820 µg/l of PO_4-P, and 2,300 CFU/100ml of enterococcus. The maximum fecal coliform concentration found in the bay in the present study (2,400 MPN/100ml) was also much lower than previously reported maximum concentrations of 1,100,000 MPN/100ml (Tuchkovenko & Rondon, 2002) and 240,000 MPN/100ml (INVEMAR, 2011). These differences could be attributed to sanitary improvements in the bay such as the closure of the bay's sewage outfall in 2013. The current results of phenol concentrations up to 2 mg/l present a new maximum value reported for the bay compared to a previous maximum value of 0.6 mg/l (Sierra Misco, 1999; Lonin, 2009).

The present study's results of concentrations of Cr (31.5-57.2 mg/kg), Cu (22.1-41.2 mg/kg), and Hg (9-183 µg/kg) in the Dique Canal's sediments are higher than those previously reported by UniNorte & Cormagdalena (2004) of Cr (<1-21.3 mg/kg), Cu (2.4-26.2 mg/kg), and Hg (<4-20 µg/kg) from six sampling sessions in the canal between 1996-2003, though attributing this difference to long- or short-term temporal variation in the canal would be difficult. Meanwhile, Tejeda-Benitez et al. (2016) have reported similar concentrations of Cd (1.44 mg/kg), Hg (50 µg/kg) and Pb (11.3 mg/kg), along with lower concentrations of Cu (16.3 mg/kg) and Ni (12.4 mg/kg) upstream in the Magdalena River at Calamar. The present study's results in the bay were similar to previous reports of Cr (Parra et al., 2011a), Cu (Parra et al., 2011b; Restrepo JD et al., 2016), Hg (Cogua et al., 2012) in Cartagena Bay. Jaramillo et al. (2016) have recently reported concentrations of Cd and Pb similar to the present study and even higher concentrations of Cr (31.2–189.2 mg/kg) and Cu (44.1–924.6 mg/kg) in the bay's sediments.

Historical levels of mercury in the bay's sediments are of particular interest due to pollution caused by a chlor-alkali plant operating from 1967 to 1978. This led to Hg concentrations between 7 mg/kg (FAO & CCO, 1978) and 33.2 mg/kg (Guerrero et al., 1995) and the plant's eventual closure in 1978. A recent study by Parra et al. (2011a, 2011b) used ^{210}Pb dating of sediment cores to reconstruct the historical levels of metals in the bay for the period 1964-2009 and confirmed that high levels of Hg (max: 18.8 mg/kg) still exist at sediment depths of 55-65 cm, corresponding to the plant's operational period. Mercury levels have since gradually declined and recent studies have found average surface concentrations of 1.88 mg/kg (Alonso et al., 2000), 0.30 mg/kg (Parra et al., 2011a, 2011b), 0.18 mg/kg (Cogua et al., 2012) and 0.13 mg/kg (this study). Though current levels are still at the Threshold Effects Level of 0.13 mg/kg (Buchman, 2008), of more concern may be the occasional finding of Hg concentrations above the Probable Effects Level of 0.70 mg/kg (Buchman, 2008) such as the maximum value of 10.3 mg/kg found by Alonso et al. (2000) and that of the present study (1.3 mg/kg; Fig. 2.8). While the latter single-sample result could be considered an outlier, it is similar to the Hg concentrations found by Parra et al. (2011b) in sediments 30-40 cm beneath the surface in the same area of the bay. Consequently, this single-sample result in the present study is deemed important as it may be indicative of a pollution event, such as the resurfacing of contaminated sediments due to dredging activities that were underway in Sept. 2014, or the inflow of another land-based pollution source. Given that Hg levels in the bay decreased with each subsequent sampling following the peak value in Nov. 2014 (Fig. 2.8), this further supports the possibility of a pollution event prior to the start of monitoring.

This study's findings of Cr, Cu and Ni show that there are other metals of concern to the ecosystem as well. Historical data of these metals (Parra et al., 2011a) show that similar concentrations date back to the period of 1965-1975, suggesting that there are continued inputs of these metals or that such concentrations are normal in this bay. These potential risks to the ecosystem, along with that due to high Cd concentrations in the canal, are supported by previous findings of high metal concentrations accumulated in the food chain of this coastal zone, including Cd in oysters (Manjarrez et al., 2008); Cd, Pb and Hg in corals (Torres & Torres, 2004); and Hg in fish, crabs, birds and humans (Alonso et al., 2000; Olivero-Verbel et al., 2008, 2009, 2013; Cogua et al., 2012).

2.6.4 Recent Trends of Fluvial Fluxes into Cartagena Bay

The downstream Magdalena and its distributary channel, the Dique Canal, show significant trends in the water discharge and sediment load records. Between 2000 and 2011, trends in fluxes were more pronounced and annual discharges increased up to 48%. For example, the Magdalena streamflow and sediment load experienced increases of 24% and 33%, respectively, with respect to the pre-2000 period. Meanwhile, fluvial fluxes from the Dique Canal were also more pronounced after 2000. A mean water discharge of 398 m^3/s before 2000 increased to about 508 m^3/s during the 2000–2010 year period, corresponding to an increase of 28%. Also, sediment load displayed an increase of 48% when comparing the mean load of 16,153 t/day during the 1984-2000 year period with the observed inter-annual mean of 23,906 t/day for the 2005-2010 year period. These results are in close agreement with the observed trends during the period between 1980 and 2010 in sediment loads of the main tributaries of the Magdalena River and also with the steep increase in deforestation during the last three decades (Restrepo et al., 2015).

The increasing behavior of sediment flux into Cartagena Bay is a major environmental concern in terms of water and sediment quality. According to a recent analysis of sedimentation and pollutant tracers (Restrepo JD et al., 2016), sedimentation rates in the outlets of the Dique Canal also show major clastic fluvial sediment inputs and a remarkable transference of sediments in the inner shelf of the bays of Cartagena and Barbacoas. While the clastic sedimentation of mud in a calcareous inner shelf under natural conditions is almost undetectable, the human-induced input of muddy sediments to these bays through the Dique Canal and secondary artificial channels is high, and average sedimentation rates are 0.7–0.8 cm/y in this coastal area. Sediment core analyses in Barbacoas Bay (Fig. 2.1), upstream of the city of Cartagena, also showed high levels of metals (As, Zn, Cu, Ni, Cr, Mn and Fe) that are sourced through the Magdalena Basin upstream of the Dique Canal at Calamar, showing that the anthropogenic activities around Cartagena Bay are not the only sources of coastal pollution.

2.7 Conclusions

This study has demonstrated various aspects of seasonal variability in the water quality of Cartagena Bay. During the rainy season (Sept. - Dec.), the bay's waters are characterized by lower salinities and greater concentrations of total suspended solids, turbidity, nitrate-nitrogen, total phosphorus, fecal coliforms and enterococcus. Increased concentrations of these parameters are particularly prevalent in the central part of the bay where freshwater plumes from the Dique Canal tend to disperse during this season. During the dry/windy season (Jan. – Apr.), lower water temperatures are found in the bay along with greater levels of biological oxygen demand and an increase in chlorophyll-*a* concentration in bottom waters. The latter result suggests that the process of primary productivity in the bay is limited by water transparency rather than nutrients which are abundant. Higher phosphate concentrations are found during the transitional season (May – Aug.), possibly due to lower concentrations of suspended solids resulting in less sorption of dissolved phosphorus. Oxygen levels were low during the rainy season, adequate during the dry/windy season, and lowest during the transitional season, evidencing the hypoxic conditions found in the bay for most of the year.

Water quality in the Dique Canal exhibited peak concentrations of total suspended solids, turbidity, chlorophyll-*a*, phosphate, total phosphorus, and phenols during the months of October and April coinciding with the onset of periods of increased runoff. On the contrary, greater concentrations of fecal coliforms and enterococcus were observed in the canal during the dry season (Jan. – Apr.) suggesting nearby sources in the canal that are otherwise diluted during high runoff conditions. Increases of cadmium concentrations both in the canal and the bay during the rainy season suggest that the canal is the principal source of cadmium to the coastal zone.

The majority of water and sediment quality parameters analyzed were found to exceed national and international threshold values. This would suggest that potential impacts of pollution to the ecosystem are likely. These water quality parameters include total suspended solids, turbidity, phosphate, total phosphorus, and chlorophyll-*a* which exceeded less-strict threshold values throughout the year and exceeded even stricter thresholds during the wet and transitional seasons. Dissolved oxygen, oxygen saturation, biological oxygen demand, and nitrate-nitrogen also exceeded threshold values during the wet and transitional seasons, demonstrating that water quality is inadequate for the ecosystem for most of the year. Increased concentrations of fecal coliforms and enterococcus during the rainy season also present a potential sanitary risk to bathers in the bay, though water quality was adequate at the beach "Playa Blanca" for the most part.

While mercury has long been studied in Cartagena, this research shows that other metals may also be of concern. Concentrations of chromium, copper, nickel, and mercury in the bay's sediments were above the Threshold Effects Level (TEL) indicating a potential risk to the ecosystem. Cadmium also exceeded the TEL during the rainy season, while nickel approached the Probable Effects Level (PEL) and lead was well-below the TEL. The finding of mercury at a concentration of nearly double the PEL in a sediment sample from November 2014 suggests a recent a pollution event, such as the resurfacing of contaminated sediments due to dredging activities or the inflow of another land-based pollution source.

2.8 Connecting Text

This chapter revealed the extent of the multi-factorial pollution issues currently found in the waters and sediments of Cartagena Bay. The general finding that pollution is in fact an issue was confirmed by the monitoring program that showed that the majority of water and sediment quality parameters exceeded national and international threshold values. This chapter also demonstrated various aspects of the bay's water and sediment quality which were previously undocumented, such as its pronounced spatial and seasonal variability, while also identifying various contaminants that have been underrepresented in previous research, such as phenols, chromium, copper and nickel. By establishing the extent of they bay's pollution issues, this chapter's findings are essential to coastal management, as they justify mitigation actions and support policy development. This knowledge will also orient the planning of future monitoring programs.

The obvious question that arises from this established knowledge is: what are the sources of pollution causing these issues? This chapter's analysis of spatio-temporal variability permitted various hypotheses and suggested some potential sources. For example, the concurrence of higher concentrations of fecal coliforms, enterococcus, sediments and nutrients both in the bay and in the Dique Canal during the rainy season would suggest that anthropogenic activities upstream of the canal are the sources of these contaminants. Yet reduced water quality in terms of NO_3-N and dissolved oxygen in the northern part of the bay suggest that other sources are also present, such as nearby sources of domestic or industrial wastewater. Impacts of local domestic wastewater were further evidenced by increased concentrations of fecal coliforms and enterococcus in the canal during the dry season, which could possibly be due to wastewater discharges from the community adjacent to the canal that receive less dilution in this season when the canal's discharge is lowest. However, in order to confidently answer the question of pollution sources, an assessment is required that compares the pollutant loads flowing from these different potential sources. Such an assessment can be complicated by limited data availability and the need to apply different methods to quantify each source's pollutant load. In the following chapter, a methodology that surmounts these limitations is presented in order to assess and compare the land-based sources of pollution discharged into the coastal zone.

CHAPTER III

An integrated approach for the assessment of land-based pollution loads in the coastal zone

"Our imagination is struck only by what is great; but the lover of natural philosophy should reflect equally on little things."

- Alexander von Humboldt, c. 1802

3.1 Abstract

The identification and prioritization of pollution sources is essential to coastal zone management. This task is complicated when a variety of pollution sources are found and by limited data availability, which can result in an inconclusive assessment and differing public perceptions, ultimately hindering the progress of management actions. This is the case in Cartagena Bay (Colombia), a Caribbean hot-spot of pollution, which receives large freshwater discharges from the Magdalena River drained via the Dique Canal along with coastal industrial effluents and untreated domestic wastewater from parts of the coastal population. This study presents a methodology for the integrated assessment of anthropogenic pollution sources discharged into the coastal zone by estimating their loads and comparing their relative contributions to receiving coastal waters. Given the lack of available data on discharges and water quality, an integrated approach is applied by combining various methods of load estimation while emphasizing the importance of calculating confidence intervals for each load value. Pollution loads from nearby sources of domestic wastewater, coastal industrial effluents and continental runoff were assessed with respect to their contributions of coliforms, total suspended solids, nitrogen, phosphorus, and biological oxygen demand (BOD). Loads from the canal's surface runoff were calculated with monthly discharge and water quality data. Domestic loads were computed using GIS analyses of population and sewerage coverage in combination with export coefficients of daily load per capita. Industrial loads were estimated based on previous studies. Results show that each type of land-based source is responsible for different pollution impacts observed in Cartagena Bay. Occasionally, inadequate recreational water quality can be attributed to nearby sources of domestic wastewater, which contribute the highest coliform load ($6.7 \pm 3.9 \times 10^{15}$ MPN/day). Continental runoff via the Dique Canal contributes the greatest sediment load ($2.5 \pm 1.9 \times 10^{3}$ t/day) causing the bay's turbid plumes and related ecosystem issues. Hypoxic conditions in the bay can be attributed to all three pollution sources which all discharge significant BOD loads (2-8 t/day), while the highest total phosphorus load comes from the Dique Canal (3.2 ± 2.4 t/day) and the highest nitrogen loads flow from the canal (3.7 ± 3.1 t·NO_3-N/day) and the industrial sector (3.1 ± 4.1 t·N/day). Given that these loads are projected to increase in future years, this study highlights the importance of prioritization and mitigation in coastal pollution management and demonstrates a method that could be applied in other places with similar problems in the Wider Caribbean Region.

3.2 Introduction

A common challenge in coastal management is to balance the priorities of environmental conservation and economic development. These priorities often create conflicts for the sustainable development of the coastal zone as the growth of human populations and industrial activities generate pollution that leads to environmental degradation. This conflict is particularly relevant in the Caribbean Region, where the economy is very dependent on tourism (15% of GDP; WTTC, 2017) drawing on the appeal of beaches and clear seawater. However, the growth of coastal populations and tourism in the region has also increased the discharge of wastewater in the Caribbean Sea, about 85% of which is untreated, posing threats to public health (UNEP/GPA, 2006). Moreover, human population growth and pollution are among the principal causes of widespread marine ecosystem change across the Caribbean, where coral reefs have declined drastically since the 1970s (Jackson et al., 2014).

A particularly challenging example of such conflicts can be found in Cartagena, Colombia. This coastal city is home to a population of 1 million people, has one of the country's largest ports and industrial zones, and represents Colombia's principal touristic destination, though various environmental issues are evident in the waters of Cartagena Bay (Tosic et al., 2017). Hypoxic conditions are likely related to the drastic reductions in artisanal fisheries observed in recent decades by the bay's rural communities (personal communication with fishermen). Turbidity has been linked with marine ecosystem degradation in the adjacent Rosario Islands Marine Protected Area (Restrepo et al., 2006, 2016a), while findings of high coliform concentrations present a potential risk to the city's beaches.

It is accepted that these environmental issues are due to land-based sources but there are many such pollution sources flowing into Cartagena Bay, including domestic and industrial wastewater along with continental runoff from the Magdalena Watershed discharged via the Dique Canal. To identify which pollution sources are primarily responsible for the bay's degraded water quality, a quantitative assessment of their pollutant loads is required. Such assessments of loads entering Cartagena Bay have been carried out previously (Garay & Giraldo, 1997; Tuchkovenko & Lonin, 2003). However, the comparison and interpretation of their results are limited as these studies did not report estimates of the uncertainty or variability involved in their load calculations, without which a decision-maker cannot confidently reach a conclusion.

In this study, we propose an integrated approach for pollution load assessment with a focus on data uncertainty and variability. The approach combines different methods of load estimation, including effluent monitoring, spatial analyses with geographical information systems (GIS), and previously published results. In order to permit a comparison of the estimates calculated by differing methods, a novel approach is proposed for the approximation of confidence intervals for each load, considering the uncertainty and variability inherent in each value used for load calculation. The application of this approach is demonstrated in Cartagena, where we aim to address the research question of "which land-based sources of pollution are responsible for the issues of hypoxia, turbidity and unsanitary conditions in the waters of Cartagena Bay?" Using this approach, ultimately the decision-makers can confidently identify the primary pollution sources and this could be applied to other coastal zones across the Wider Caribbean Region where data availability and monitoring programs are common limitations.

3.3 Study Area

Cartagena Bay is a micro-tidal estuary located on the north coast of Colombia in the Caribbean Sea (Fig. 3.1). The bay has a surface area of 84 km^2, a maximum depth of 32 m, and a strong vertical salinity stratification. It receives water from two seaward straits (Bocachica and Bocagrande) and various freshwater sources. Foremost to these freshwater sources is continental runoff from the Magdalena River via the Dique Canal which discharges into the southeastern part of the bay with strong seasonal variability. Other freshwater sources include domestic and industrial wastewater, as well as runoff from a small coastal catchment area around the bay, though this runoff contribution is considered negligible in comparison to that of the Dique Canal.

Freshwater runoff from the Dique Canal has a seasonality related to the Magdalena River, with greater discharges from October to December and lower levels from February to April (Molares & Mestres, 2012a). The Dique Canal is a man-made distributary channel that diverges from the Magdalena River at Calamar (Fig. 3.1) and flows along 114 km to Cartagena Bay, where previous studies have reported an approximate discharge of 55-250 m^3/s (Tuchkovenko & Lonin, 2003) and a total sediment flux of 1.9 Mt/y (Restrepo et al., 2018) which has increased over the past decade (Restrepo et al., 2015). Modelling predictions show these fluxes are intensifying as sediment loads are projected to increase by as much as 317% by the year 2020 (Restrepo et al., 2018).

The Magdalena watershed has an area of 260,000 km^2, covers approximately 25% of the country's land area, and includes approximately 80% of the national population. It is the main Andean catchment of Colombia as well as the main contributor of fluvial fluxes to the Caribbean Sea (Restrepo & Kjerfve, 2000; Restrepo, 2008). As such a large amount of industrial, agricultural and mining areas can be found in the Magdalena basin, the waters that flow to Cartagena Bay via the Dique Canal carry many potential pollutants along with a significant sediment load.

Domestic wastewater enters the bay from various small populations around the bay without sewerage service. These include the rural communities of Ararca, Bocachica, Caño del Oro, Punta Arena, and Tierra Bomba (Fig. 3.1) as well as some neighborhoods to the south of the city whose sewage flows to subterranean wells that can be susceptible to seepage or overflow during storm events. The community of Pasacaballos does have a sewage system but it discharges without treatment into the Dique Canal about 2.5 km upstream of its outlet. In previous years, about 40% of the city's sewage system itself (~48,000 m^3/day) was also discharged directly into Cartagena Bay without treatment via an 800 m submarine outfall (UNDP-UNOPS, 1999; Tuchkovenko & Lonin, 2003), though since 2013 Cartagena's sewage system has been routed to a new outfall sufficiently far north of the city to not affect the bay. However, on occasion the city's sewerage system overflows, particularly during intense rain storms, resulting in direct discharge into the bay through the old submarine outfall and backup outlets along the coast (personal communication with the city water authority, AcuaCar).

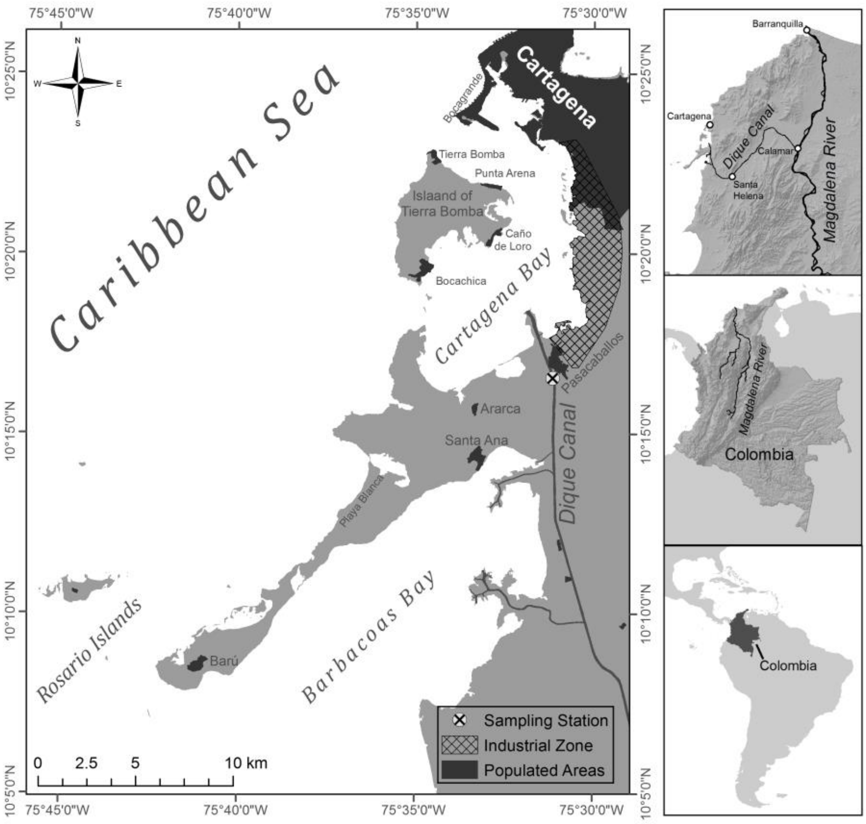


Figure 3.1: Study area showing the populated areas and industrial zone around Cartagena Bay along with the Dique Canal, Magdalena River and location in South America and the Wider Caribbean Region.

Cartagena's industrial zone runs along the east coast of the bay (Fig. 3.1). While some of these industries discharge to the city's sewage system, the majority of them have their own treatment and discharge directly to the bay or indirectly via small canals (UNDP-UNOPS, 1999). Activities in this zone include a petroleum refinery with multiple distribution terminals, chemical plants, cement factories, aquaculture, electric plants, food processing industries (carbonated beverages, dairy, poultry, fish), production of plastics, leather, and other manufacturers (INVEMAR-MADS, 2011). This industrial zone represents 50% of Cartagena's GDP, of which at least 70% can be attributed to the petro-chemical sector (Cardique & AGD, 2006).

3.4 Materials and Methods

3.4.1 Load Assessments

The land-based sources of pollution leading to the water quality issues of turbidity, hypoxia, and inadequate recreational waters were analyzed by assessing the loads of five water quality parameters: total suspended solids (TSS), nitrogen (various forms), total phosphorus (TP), biological oxygen demand (BOD), and coliforms. TSS loads were analyzed as an indicator of the land-based sources responsible for high turbidity levels in Cartagena Bay, while coliform loads were analyzed to identify sources affecting the bay's adequacy for recreational purposes. To assess the pollution sources responsible for the bay's hypoxia, the loads of BOD, nitrogen and phosphorus were analyzed. BOD is an indicator of hypoxia issues as it measures the amount of dissolved oxygen consumed by the biological decomposition of organic matter. Nitrogen and phosphorus are also indirect indicators of hypoxia issues as the excess supply of these nutrients leads to the primary production of organic matter in the water column that eventually decomposes and consumes dissolved oxygen as well (Newton & Mudge, 2005).

The loads of the five water quality parameters were assessed for the three most likely sources: coastal domestic wastewater, coastal industrial wastewater and continental runoff. The sources of domestic and industrial wastewater considered here are those flowing from nearby sources in the coastal zone itself. Continental runoff is considered as the water flowing from the Dique Canal, which also contains other upstream sources of domestic and industrial wastewater, along with agricultural and mining areas. But for the purposes of this study, the three groups of land-based sources will be termed: domestic wastewater (from nearby coastal sources), industrial wastewater (from nearby coastal sources), and continental runoff (from the Dique Canal, which includes a great variety of upstream discharges of wastewater and non-point sources).

These three most likely sources have very different characteristics, requiring different methods of load assessment. To overcome this challenge, the approach presented here integrates different methods for the assessment of each pollutant source (Fig. 3.2). Methods of effluent monitoring were carried out to characterize continental runoff, GIS analyses of population were employed to assess coastal domestic wastewater, and previously published results were used to estimate coastal industrial wastewater loads. While each of these methods has been previously applied (e.g. McPherson et al., 2002; Tsuzuki, 2006), the novelty of the present approach is in the integration of these different methods and the focus on incorporating confidence intervals into the analysis.

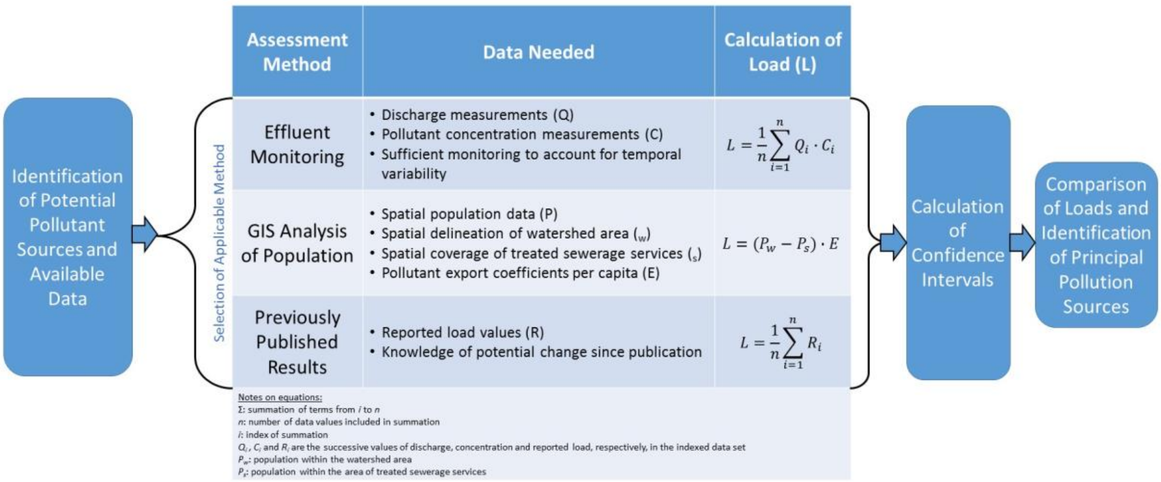


Figure 3.2: Framework of the integrated approach for pollutant load assessment.

The approximation of confidence intervals is the key to comparing results from different methods. This was done by quantifying the uncertainty and variability inherent in each value used for load calculation, and by carrying these quantities through the load calculation process in accordance with the rules for error propagation (Fig. 3.3). Qualitative estimates of uncertainty have previously been used to better inform environmental management on target setting of pollutant loads in the rivers of the Great Barrier Reef catchment area (Brodie et al., 2009). By quantitatively approximating confidence intervals, the present study goes a step further to provide environmental management with an idea of the potential range of values above and below each calculated load, allowing for a comparison of results with greater confidence.

Assessment Method	Estimates of Uncertainty or Variability	Calculation of Confidence in Values	Calculation of Load Confidence Interval (δL)
Effluent Monitoring	• Triplicate discharge measurements (δQ) • Triplicate pollutant concentration measurements (δC) • Temporal variability among 12 monthly loads (δL_T)	$\delta Q = S.D.(Q_{1-3})$ $\delta C = S.D.(C_{1-3})$ $\delta L_T = S.D.(L_{1-12})$	$\delta L_i = L_i\sqrt{\left(\frac{\delta Q_i}{Q_i}\right)^2 + \left(\frac{\delta C_i}{C_i}\right)^2}$ $\delta\bar{L} = \frac{1}{12}\sqrt{\sum_{i=1}^{12}(\delta L_i)^2}$ $\delta L = \delta L_T + \delta\bar{L}$
GIS Analysis of Population	• Fraction of uncertainty in reported population value (α) • Fraction of change in population over time since report (β) • Variability of available pollutant export coefficients (δE)	$\delta P = P \cdot (\alpha + \beta + \alpha \cdot \beta)$ $\delta E = S.D.(E_{1-n})$	$\delta L = L\sqrt{\left(\frac{\delta P}{P}\right)^2 + \left(\frac{\delta E}{E}\right)^2}$
Previously Published Results	• Fraction of uncertainty in each reported load value (γ) • Variability of loads reported (δR_V) • Fraction of potential change in loading over time since report (θ)	$\delta R_i = \gamma \cdot R_i$ $\delta R_V = S.D.(R_{1-n})$ $\delta R_t = \theta \cdot L$	$\delta\bar{R} = \frac{1}{n}\sqrt{\sum_{i=1}^{n}(\delta R_i)^2}$ $\delta L = \delta R_t + (\delta R_V + \delta\bar{R}) \cdot (\theta + 1)$

Notes on equations:
δX: estimate of uncertainty or variability for variable X
n: number of data values included in series
$S.D.$: sample standard deviation of data values 1 to n (for example, $Q_{1\text{-}3}$, $C_{1\text{-}3}$, $L_{1\text{-}12}$, $E_{1\text{-}n}$, $R_{1\text{-}n}$)
L_i, Q_i, C_i and R_i are the successive values of load, discharge, concentration and reported load, respectively, in the indexed data set
$\delta\bar{L}$ and $\delta\bar{R}$ are the uncertainties calculated for the average pollutant load ($\bar{L}$) and average reported load ($\bar{R}$), respectively
Σ: summation of terms from i to n
i: index of summation

Figure 3.3: Integrated approach for the approximation of confidence intervals for pollutant loads (L).

3.4.2 Effluent Monitoring

Given the complexity and immense variety of pollution sources found in the Magdalena River watershed, the most practical method deemed for the assessment of its pollutant loads was to measure them at the catchment's coastal outlet. This was made possible by an extensive 2-year monthly water quality monitoring program. Water quality and discharge were measured monthly from September 2015 through August 2016 in the Dique Canal at a location upstream of the Pasacaballos community (Fig. 3.1). The quantification of runoff loads using measured discharge and pollutant concentration data has been well-established in many previous studies (e.g. McPherson et al., 2002; Restrepo et al., 2006; Tosic et al., 2009; Joo et al., 2012).

Water velocity was measured with a Sontek mini-ADP (1.5 MHz) along a cross-stream transect three times on each monthly sampling date and discharge values were subsequently calculated with the Sontek River Surveyor software. The average discharge value of the three repeated transect measurements was used for the monthly calculation of pollutant loads. The sample standard deviation of the three discharge measurements was used in the computation of the confidence interval (Fig. 3.3). Henceforth, the term "standard deviation" will be understood as the sample standard deviation due to the data's small sample size. Triplicate surface water samples were collected between the hours of 9:00-12:00 (Karydis & Kitsiou, 2013). Samples were stored, cooled, and transported to the Cardique Laboratory for analysis of TSS, BOD, TP, and nitrate-nitrogen (NO_3-N) and as well as to the AcuaCar Laboratory for analysis of fecal coliforms, all by standard methods (APHA, 1985). The average concentration and standard deviation of the three samples were used to calculate the monthly pollutant loads and confidence intervals, respectively.

Monthly loads were calculated as a product of the discharge and pollutant concentration, and converted into tonnes per day (t/d). As the results showed great temporal variability, an annual average was calculated from the 12 monthly measurements in order to compare with the loads of the nearby sources of domestic and industrial wastewater. The confidence interval for the annual average of daily runoff loads was computed by factoring in the uncertainty due to measurement (standard deviations of discharge and pollutant concentration) as well as the temporal variability of loads, which was calculated as the standard deviation of the 12 monthly measurements (Fig. 3.3).

3.4.3 GIS Analysis of Population

Spatial analyses were applied using GIS to assess the population of inhabitants without a connection to the city's sewage system. Spatial coverages of the inhabited areas within the bay's coastal watershed were obtained from the online geoportal of Colombia's National Administrative Department of Statistics (DANE, 2017b). Population values for each inhabited sector were also obtained online from the same department (DANE, 2017a) and associated with corresponding sectors of the spatial coverage of inhabited areas in order to create a spatial coverage population. A map of the city's sewage network was obtained from Cartagena's water authority (AcuaCar, 2016) and digitized into a single polygon of the sewage system's coverage area. This area was then overlaid with the population coverage in order to identify the populations without connection to the sewerage service. The populations of these areas without sewage services were then summed and termed the "non-serviced population."

The loads of these nearby sources of domestic wastewater were estimated for the non-serviced population using export coefficients, which approximate the load of a given pollutant generated by each non-serviced habitant per day. This approximation method has been used in previous studies (Tsuzuki, 2006), including research at the national level in Colombia (INVEMAR, 2009) and at the regional level in the Wider Caribbean Region (UNEP-UCR/CEP, 2010). Among the available export coefficients, the values proposed by INVEMAR (2009) were ultimately selected for this study based on the criteria that they have been previously applied in Colombia and were near the average values of all of the reviewed references.

To approximate confidence intervals for the domestic wastewater loads, the sources of uncertainty and variability were considered for the values of population and the export coefficients (Fig. 3.3). For the total value of the non-serviced population, two sources of uncertainty were considered: the potential error of measurement in the original census reported in the year 2005, and the change in population between 2005 and 2016. The calculation of a confidence interval for the potential error of measurement is reported by the census itself as ±13.7% (DANE, 2008). A confidence interval for the change of population between 2005 and 2016 was approximated as ±15%, based on the projections of the census that predict a population increase of 13% in the urban parts of Cartagena and a decrease of 17% in the rural parts of Cartagena over this time period. To approximate the confidence interval of the export coefficient values, the standard deviation of the coefficient values reported in the reviewed references was used (Tchobanoglous et al., 2003; Tsuzuki, 2006; INVEMAR, 2009; UNEP-UCR/CEP, 2010). However, in the case of the total coliforms parameter, only one coefficient value was reported and so half of this value was used as an approximation of its confidence interval.

3.4.4 Previously Published Results

The industrial sector located along the eastern coast of Cartagena Bay was characterized through the identification and classification of the different industries currently operating. As data about the pollutant loads discharged from Cartagena's industrial sector was either not available or incomplete, for the present study we used the average value of the total loads reported by previous works of Tuchkovenko & Lonin (2003), Ramírez et al. (2006), and INVEMAR (2009). For the approximation of confidence intervals, the uncertainty in the individual loads reported as well as the variability among reported loads were considered (Fig. 3.3). As the previous publications did not report confidence intervals with the reported loads, a robust approximation of 50% uncertainty was assumed for each reported load value. The standard deviation of the reported loads was then calculated to account for the variability among reported values. However, in the case of the total coliforms parameter, only one total load was reported (Ramírez et al., 2006) and so half of this value was used as an approximation of its confidence interval. In consideration of the changes that may have occurred in this industrial sector since the time of these previous reports, an additional ±50% was aggregated to the confidence interval. This ±50% is used as a rough compensation of the increased industrial activity that has been developed in this growing industrial sector, as well as possible reductions in pollution loads due to improved wastewater treatment that may have been introduced to some industries since the previous reports.

3.5 Results

3.5.1 Continental Runoff via the Dique Canal

Discharge measured near the coastal outlet of the Dique Canal is shown in Figure 3.4. Monthly discharge ranges from 22 to 177 m^3/s, which is lower than previous measurements of 55-250 m^3/s reported for the canal's outlet (Tuchkovenko & Lonin, 2003). These lower values are likely because the period from 2015 to early-2016 was characteristic of a low rainfall year resulting from an El Niño event. Thus the discharge values measured in the present study may be considered as an under-estimate of average flow conditions, particularly during the rainy season of Sept.-Dec 2015 and the dry season of Jan.-Feb. 2016.

Monthly measurements of water quality concentrations show a great seasonal variability (Fig. 3.5) with some parameters increasing sharply during the dry months of January and February (BOD, NO_3-N), while others are higher in October and April at the onset of the rainy season (TSS, TPP, FC). Daily loads calculated for each month also show a high amount of temporal variability with loads being 4-8 times higher in some months than others, and up to even 2 orders of magnitude higher in the cases of FC and TSS. In general, temporal variability shows a much greater influence on the confidence intervals of average annual loads than that due to the uncertainty of measurements.

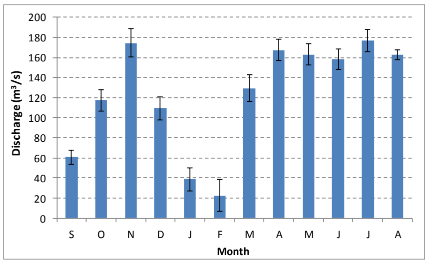


Figure 3.4. Surface water discharge (m^3/s) measured monthly in the Dique Canal, 3 km upstream of its coastal outlet, between Sept. 2015 and Aug. 2016. Sample standard deviations are shown as error bars.

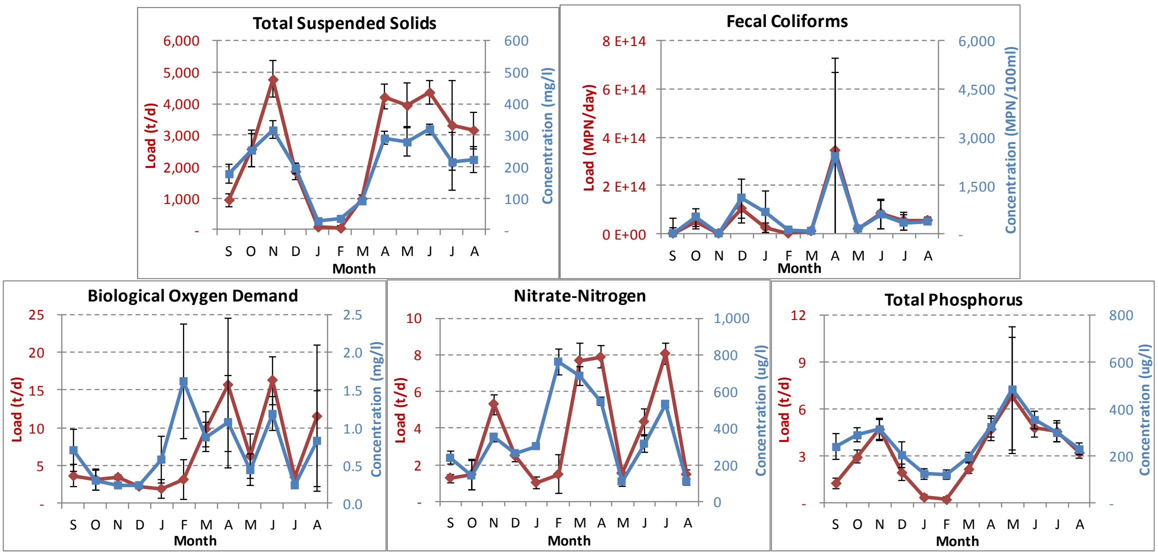


Figure 3.5. Monthly concentrations of water quality parameters (blue, right axes) and average daily loads for each month (red, left axes) near the coastal outlet of the Dique Canal, between Sept. 2015 and Aug. 2016. Error bars represent the confidence intervals approximated for each value.

3.5.2 Coastal Sources of Domestic Wastewater

The total population of inhabitants with no connection to the city's sewage system was calculated as 33,381 people, with a confidence interval of ±10,274 people. The non-serviced populations were distributed in various parts around the bay (Fig. 3.6a). These populations include various rural communities (Ararca, Bocachica, Caño del Oro, Punta Arena, Tierra Bomba, and Pasacaballos) as well as some urban neighborhoods to the south of the city.

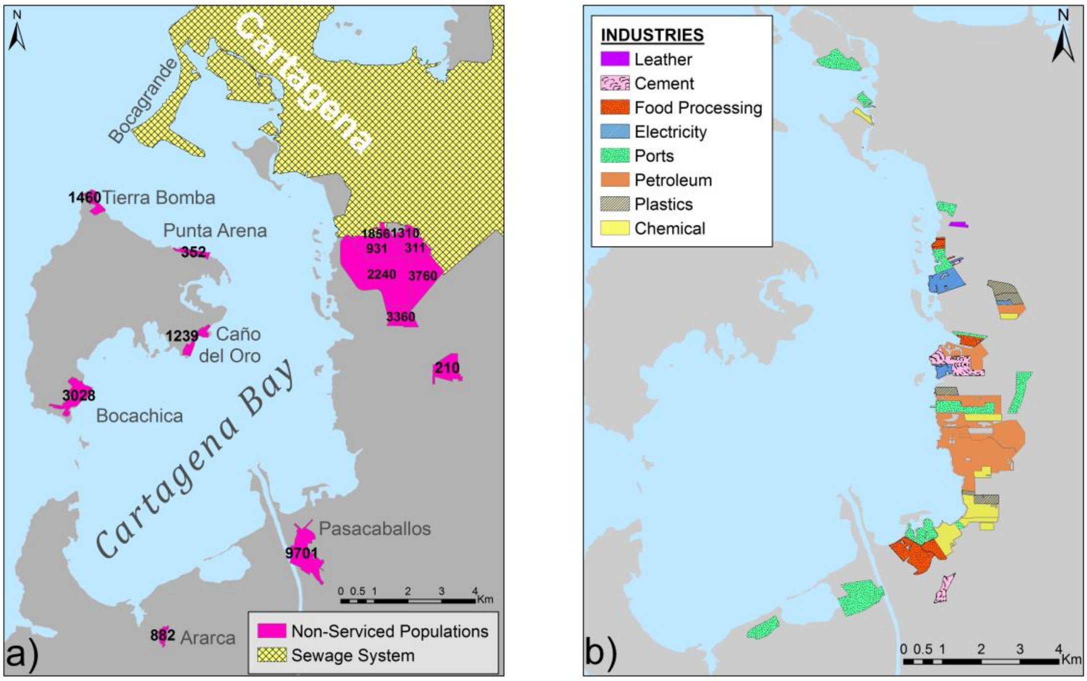


Figure 3.6. a) Populations in the coastal zone of Cartagena Bay with no connection to the city's sewage system (non-serviced populations) and spatial coverage of the sewage system. Population values projected for 2016 based on the most recent census of 2005 (DANE, 2017a) are shown for each area. b) Industries in the coastal zone of Cartagena Bay, grouped by general classifications of economic activities. Spatial data was obtained from Cardique & AGD (2006).

Daily loads of nearby sources of domestic wastewater were calculated to be 1.7 ± 0.6 t/d for BOD and 1.7 ± 0.9 t/d for TSS (Table 3.1). Smaller daily loads were calculated for inorganic nitrogen (0.4 ± 0.1 t/d) and total phosphorus (0.03 ± 0.01 t/d), while a very large load of 6.7 ± 3.9 x10^{15} of total coliforms was estimated. The large magnitude of confidence intervals attributed to these approximations is the result of the high levels of data uncertainty in both the export coefficients and population values.

Table 3.1. Export coefficients and daily loads of nearby sources of domestic wastewater.

Parameter	BOD	Inorganic Nitrogen	TP	TSS	Total Coliforms
Unit	g/hab/d	g/hab/d	g/hab/d	g/hab/d	MPN/hab/d
Export Coefficient	50	12	0.8	50	2 $x10^{11}$
Confidence Interval (±)	8	2	0.1	21	1 $x10^{11}$
Unit	t/d	t/d	t/d	t/d	MPN/d
Daily load	1.7	0.4	0.03	1.7	7 $x10^{15}$
Confidence Interval (±)	0.6	0.1	0.01	0.9	4 $x10^{15}$

3.5.3 Coastal Sources of Industrial Wastewater

The spatial distribution of the industries identified in the coastal area of Cartagena Bay is shown in Figure 3.6b. Table 3.2 shows the daily loads of nearby sources of industrial wastewater, calculated as the average value of the three previous publications to report this information. The loads reported by Ramírez et al. (2006) appear to be most similar to the average values calculated as well as the only reference that reported total coliform loads. The loads reported for BOD, TN and TP are relatively consistent with one another, though the large disparity between the TSS load values reported does highlight the uncertainty inherent in such data. In general, the confidence intervals for the average load values are quite high, with magnitudes that are greater than the average values themselves. This is due to the large amount of accumulated uncertainty in the average values, as large estimates had to be made to conservatively account for the uncertainty in the individual reported loads along with the potential change over time since these previous reports.

Table 3.2. Previously published results of daily loads from nearby sources of industrial wastewater. Dashes (-) indicate that data were not reported.

Previous Studies	BOD (t/d)	TSS (t/d)	TN (t/d)	TP (t/d)	Total Coliforms (MPN/d)
Ramírez et al., 2006	7.09	23.04	3.29	0.19	2.7 $x10^{7}$
Tuchkovenko & Lonin, 2003	6.04	39.30	3.75	0.14	-
INVEMAR, 2009	9.47	7.96	2.28	0.13	-
Average daily load	7.54	23.43	3.11	0.15	2.7 $x10^{7}$
Confidence Interval (±)	9.72	46.79	4.06	0.19	5.5 $x10^{7}$

3.5.4 Load Comparison

When comparing the loads of the three groups of land-based sources (coastal domestic wastewater, coastal industrial wastewater, and upstream continent runoff), it was found that in the case of coliforms, the principal source of pollution is clearly the nearby coastal domestic wastewater (Fig. 3.7). The high loads of total coliforms approximated in this study (6.7 ± 3.9 $x10^{15}$ MPN/d) due to non-serviced coastal populations is several orders of magnitude higher than that of the industrial sector (2.7 ± 5.5 $x10^{7}$ MPN/d) with a mean difference of 6.7 ± 3.8 $x10^{15}$ MPN/d between the two loads. The domestic load is also two orders of magnitude higher than the fecal coliform load from the Dique Canal (6.2 ± 13.1 $x10^{13}$ MPN/d). As the range covered by the confidence interval of these mean differences does not include zero, it may be said with certainty that the domestic load is significantly greater than the industrial and runoff loads with a statistical significance of 1.7 standard deviations in both cases. While it is not normally recommended to compare results of different parameters, as is the case of analyzing total coliforms due to domestic wastewater and fecal coliforms discharged by the Dique Canal, the lack of data availability make this a necessity justified by the importance of proceeding with management actions. Furthermore, fecal coliforms are included within total coliform measurements, though the difference between the two parameters typically would not differ by more than one order of magnitude. For this reason, it is considered reasonable to accept that the principal source of coliforms is domestic wastewater.

In the case of TSS and TP (Fig. 3.7), the source is very clearly the Dique Canal. The approximate TSS and TP loads of the canal are 2.5 ± 1.9 $x10^3$ t/d and 3.2 ± 2.4 t/d, respectively, which outweigh the domestic (TSS: 1.7 ± 0.9 t/d; TP: 0.03 ± 0.01 t/d) and industrial (TSS: 23.4 ± 46.8 t/d; TP: 0.15 ± 0.19 t/d) wastewater loads by 2-3 (TSS) and 1-2 (TP) orders of magnitude. When comparing the canal to industry and to domestic sources, the mean differences are 2.5 ± 1.9 $x10^3$ t/d for TSS and 3.1 ± 2.4 t/d for TP. The ranges of confidence on these mean differences do not include zero, and so it may be said with certainty that the TSS and TP loads from the canal are significantly greater than the respective industrial and domestic loads with a statistical significance of 1.3 standard deviations in all cases. In these cases of TSS, TP and coliforms, the upper bounds of the confidence intervals of the secondary sources are well-below the lower bounds of the principal source. The association between TSS and TP may be expected due to phosphorus' high potential for sorption to sediment particles (Schlesinger, 1997).

In the case of BOD (Fig. 3.7), the principal sources appear to be the Dique Canal (6.7 ± 6.5 t/d) and the industrial sector (7.5 ± 9.7 t/d), though domestic wastewater also makes a substantial contribution (1.7 ± 0.6 t/d), which lies within the lower bound of the canal's and industry's confidence intervals. The mean differences between the BOD of these sources (canal-domestic: 5.0 ± 6.6 t/d; industry-domestic: 5.9 ± 9.7 t/d) include zero within the range of confidence and so it cannot be said with certainty that $L_{canal}>L_{domestic}$ or $L_{industry}>L_{domestic}$. Furthermore, considering that the domestic wastewater load is likely an underestimate since the overflow from the city's sewage system was not included in the load calculations, the importance of the domestic BOD load should not be ignored.

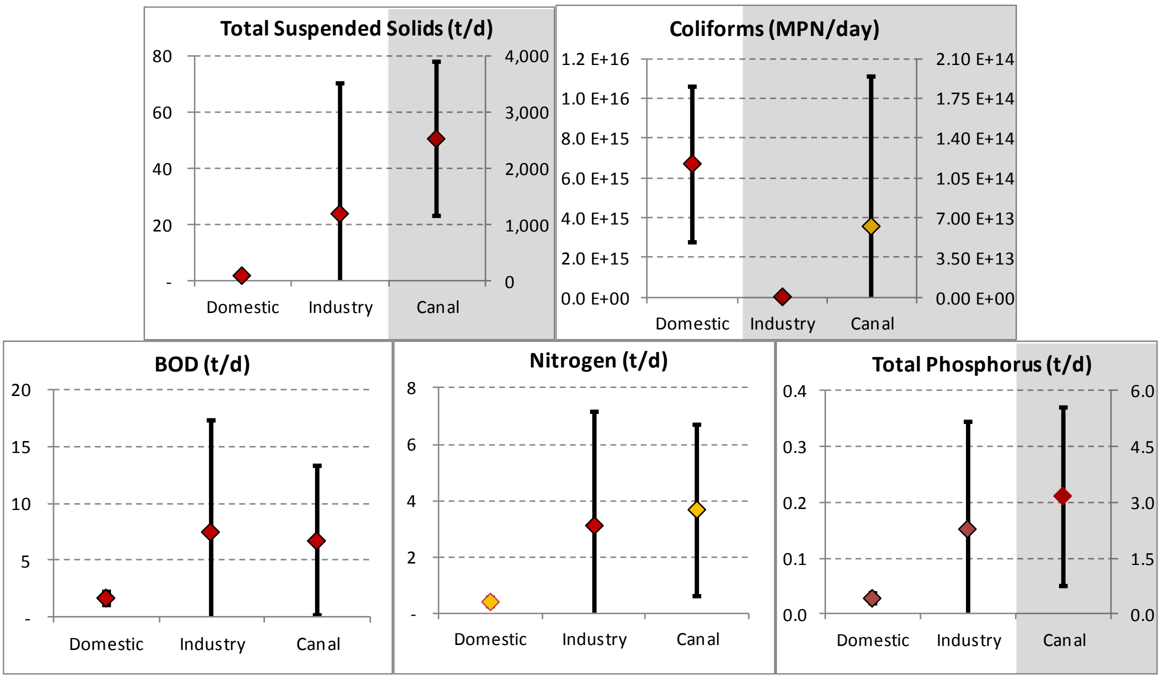


Figure 3.7. Comparison of daily loads from three groups of land-based sources of pollution: coastal domestic wastewater, coastal industrial wastewater, and upstream continent runoff. Error bars represent the confidence intervals approximated for each value. Loads in grey boxes correspond to another scale on right vertical axes. Yellow symbols represent a different but inclusive form of the water quality parameter: fecal coliforms, inorganic N (domestic) and NO_3-N (canal).

In the case of nitrogen, the principal sources also appear to be the Dique Canal and the industrial sector (Fig. 3.7). Though this analysis is complicated by the fact that each group of sources reported a different type of nitrogen (domestic: inorganic N; industrial: total N; canal: NO_3-N), a reasonable comparison can still be made considering that NO_3-N is a component of inorganic N, and inorganic N is a component of total N (i.e. NO_3-N < inorganic N < TN). It is therefore clear that the Dique Canal is a principal source of nitrogen as its TN concentration would be even higher than the NO_3-N load calculated in this study (3.7 ± 3.1 t/d), along with the industrial load reported as TN (3.1 ± 4.1 t/d). What is uncertain is how high the total nitrogen load of domestic wastewater sources would be, as they are reported here as inorganic N (0.4 ± 0.1 t/d) which represents just a portion of total N. Therefore, while we cannot be certain if domestic wastewater is a principal source of N, it can be stated that domestic wastewater is a significant source of N to Cartagena Bay, in consideration as well that the domestic load is an underestimate.

3.6 Discussion

3.6.1 Benefit of the Approach

While the assessment of coastal pollution loads has been applied in the past (Garay & Giraldo, 1997; Tuchkovenko & Lonin, 2003), the approach proposed in this study allows for the comparison of loads through the quantification of confidence intervals, thus permitting a confident identification of primary pollution sources. The persistence of pollution issues in Cartagena Bay may be due in part to how difficult it has been to identify individual sources. With the pollution source identification achieved by our approach, public perception may be easier to align with reality while financially-motivated attempts to influence those perceptions can be better constrained by data. By compiling the available data and assessing these pollutant loads, the resulting load comparison permits the coastal zone manager to better inform stakeholders about which pollution sources are the dominant problem and manage them more effectively.

The inclusion of confidence intervals in the analysis provides further clarity to decision makers in that they can consider the relative uncertainty and variability of each load estimate as they compare across sources. Data availability is a common limitation to making such assessments and finding clear differences between sources. By calculating confidence intervals, even with estimates derived using different methods, it allows the analyst to better portray differences in results and identify where additional data gathering is needed. The available data will naturally be different when applying this approach in other study areas, though the methods of confidence interval approximation for the purposes of load comparison could be applied anywhere. Ultimately, the benefit of plotting these load estimates with confidence intervals allows decision-makers to base their actions on numerically supported conclusions, which may or may not support previous assumptions.

3.6.2 Further Considerations to Nearby Sources of Domestic Wastewater

In reality, the domestic wastewater load received by the bay is even higher than the load approximated in this study because the load due to overflow of the city's sewage system was not included. The flow through this system occasionally exceeds capacity, particularly during rainy conditions as it also receives urban stormwater runoff, resulting in direct discharge into the bay through the bay's old submarine outfall and backup outlets along the coast (personal communication with the city water authority, AcuaCar). This likely explains the bay's seasonal variation of coliform concentrations which are highest during the rainy season, particularly to the north near the city, and remain high during the dry season to the south near a large non-serviced population (Tosic et al., 2017). Therefore, the solution to the issue of inadequate recreational waters would require both the mitigation of non-serviced populations as well as improved capacity to the city's sewage system.

Another potential source of pollution related to domestic wastewater could be organic matter that has accumulated over time at the bottom of the bay. The old submarine outfall in Cartagena Bay operated between the years 1960 and 2013, during which it received about 40% of the city's sewage (~48,000 m^3/day) and discharged 800 m from the coast without treatment (UNDP-UNOPS, 1999; Tuchkovenko & Lonin, 2003). Some of the lowest dissolved oxygen concentrations in the bay can be found in the area of this old outfall (Fig. 3.1) and so perhaps one of the causes of the bay's hypoxic conditions is the legacy organic matter accumulated in this area after so many years of untreated discharge.

3.6.3 Further Considerations to Nearby Sources of Industrial Wastewater

Further research is needed on the discharges of the specific industries found around Cartagena Bay. While such data could not be obtained for this study, the loads reported in previous publications along with the inclusion of a rough yet conservative approximation of changes over time (±50%) into the confidence interval permitted the finding that the industrial sector contributes significant BOD and nitrogen loads to Cartagena Bay. Though the loads of specific industries are unknown, knowledge of the general characteristics of industrial discharges can be used to narrow down the potential sources. A previous study by INVEMAR-MADS (2011) showed that among the industrial activities found in Cartagena, food processing and chemical plants are the most likely to contain high levels of organic matter and nutrients in their discharge. Therefore, while further research is needed to improve the approximation of industrial loads, it may also be recommendable to focus pollution control efforts on the food processing and chemical industries towards the mitigation of the bay's hypoxia.

3.6.4 Further Considerations to Runoff

An important question is how the flow of freshwater and sediments will change in years to come. On one hand, modelling predictions based on past trends of these fluxes show that they are intensifying and sediment loads are projected to increase by as much as 317% by the year 2020 (Restrepo et al., 2018). On the other hand, an ongoing hydraulic intervention in the Dique Canal is being implemented by the National Adaptation Fund (http://sitio.fondoadaptacion.gov.co/) which plans to construct hydraulic doors along the canal and reduce flow into the bay by ~50%. However, this plan's estimate of reduced flow does not take into account potential increases in water and sediment fluxes from the Magdalena watershed. Consequently, the resulting balance between this hydraulic intervention and the Magdalena's increasing trends remains to be seen. It should also be noted that the annual sediment load calculated by the present study (0.9 ± 0.7 Mt/year) is less than previous estimates of 1.9 Mt/year (Restrepo et al., 2018) which may be attributed to the El Niño event that occurred during the present study.

3.6.5 Community Perception

To have an idea of what the public perception is of pollution issues in Cartagena, we asked 110 local artisanal fishermen the simple open-ended question: "what do you think is the source of pollution in Cartagena Bay?" Responses were grouped in general categories, including the Dique Canal and industries which accounted for 39% and 45% of the responses, respectively. However, the fishermen did not identify domestic wastewater as a pollution sources, but rather tourism (16%). These fishermen are the people that are most affected by the bay's pollution issues, as the resulting impacts on fish populations have a direct effect on their livelihood. Their perception of the canal's impact may be expected, as it is evident from the year-round prevalence of sediment plumes flowing from the canal into the bay. While industry might have a less visible effect on water quality, perhaps their perception of industrial impacts reflects an association between the bay's increased industrial activities and the simultaneous degradation of their fish stock. The depletion of fish populations and the growth of tourism have resulted in an economic shift towards tourism for these people (Garzón, 2016; Castillo, 2016). However, the fish stock's depletion is complex as there are many likely causes in addition to hypoxia, such as past chemical and oil spills, over-fishing and changes in the bay's sewage outfall.

3.7 Conclusions

The benefits of applying an integrated approach for the assessment of land-based pollutant loads have been shown by this study for the case of the coastal zone of Cartagena Bay, Colombia. Despite the lack of available data, the approach permitted an approximation of pollutant loads by combining multiple computation methods with a focus on estimating confidence intervals for the data and calculations utilized. By implementing a monthly monitoring program at the outlet of a large coastal watershed, the consideration of seasonal variability in the assessment was made possible. While the available data will naturally be different for other study areas, the methods for estimating loads and confidence intervals shown here could easily be adapted to make this approach applicable to other coastal areas with similar problems commonly found across the Wider Caribbean Region.

When comparing the loads of three groups of land-based sources (coastal domestic wastewater, coastal industrial wastewater, and upstream continent runoff) in Cartagena Bay, it was found that the principal source of coliforms is nearby coastal domestic wastewater. The high loads of total coliforms approximated in this study ($6.7 \pm 3.9 \times 10^{15}$ MPN/d) due to non-serviced coastal populations is several orders of magnitude higher than that of the industrial sector ($2.7 \pm 5.5 \times 10^{7}$ MPN/d) and two orders of magnitude higher then the fecal coliform load from the Dique Canal ($6.2 \pm 13.1 \times 10^{13}$ MPN/d). This problem is further compounded by domestic wastewater that occasionally overflows from the city's sewage system, which is the likely explanation for the increase in coliform concentration in the bay's surface waters during the rainy season.

The principal source of TSS and TP in the bay is clearly the Dique Canal which discharges approximate loads of $2.5 \pm 1.9 \times 10^{3}$ t/d and 3.2 ± 2.4 t/d, respectively, which outweigh the domestic and industrial wastewater loads by 2-3 (TSS) and 1-2 (TP) orders of magnitude. While turbidity issues in Cartagena Bay can clearly be related to the canal, the issues of hypoxia may be associated with all three of the sources evaluated. The greatest BOD and nitrogen loads appear to come from the canal and the industrial sector, though a significant contribution of these parameters was also found for domestic wastewater sources as well.

3.8 Connecting Text

By developing a methodology that permits a confident comparison of pollution loads quantified by different methods, this chapter determined the principal land-based sources generating the pollution issues established in the previous chapter. In addition to the Dique Canal (the principal source of sediments and phosphorus), other primary pollution sources were also identified including local domestic wastewater (the principal source of coliforms) and local industrial wastewater (a primary source of nitrogen and organic matter), which confirms some of the hypotheses postulated in chapter II. These conclusions show that sole mitigation of the Dique Canal would not effectively solve all of the pollution issues in Cartagena Bay, as nearby sources of domestic and industrial wastewater would also require mitigation.

Nevertheless, mitigation strategies in the coastal zone of Cartagena to date have not been balanced among these different sources. While the city's sewage system has been improved by replacing the bay's submarine outfall with a new one discharging far north of the city, the current system still has issues of overflow into the bay and there remain various populations along the coast without sewage service. Industrial pollution control in this coastal zone remains largely a mystery as policy until now has only required industries to report TSS and BOD concentrations in their discharges twice a year, while the industries themselves are responsible for sampling and the data is not publicly available. Mitigation of the Dique Canal has received much more attention; an ongoing intervention plans to construct hydraulic doors along the Dique Canal and thus reduce flows of freshwater and pollution into Cartagena Bay (Fondo Adaptación, 2018). However, although reducing flows from a principal pollution source, like the Dique Canal, would seem logical for pollution mitigation, it should also be considered that the capacity of a coastal semi-enclosed water body, like Cartagena Bay, to buffer fluvial fluxes between the land and sea is sensitive to hydrological changes that can affect its water renewal process.

In the following chapter, a hydrodynamic model (MOHID) is calibrated for Cartagena Bay in order to assess how upstream hydrological changes will affect the bay's hydrodynamic processes and water renewal rates. In addition to plans to reduce flows in the Dique Canal, recent tendencies in the watershed also project increased flows, and so the model is applied to simulate both scenarios of increased and decreased freshwater discharge in the bay. This information can thus be used to evaluate the effect that upstream freshwater changes could have on the bay's hydrodynamics, which could be expected to play an important role in the bay's capacity to receive pollution.

CHAPTER IV

Hydrodynamic modelling of a polluted tropical bay: Assessment of anthropogenic impacts on freshwater runoff and estuarine water renewal

"To explain all Nature is too difficult a task for any one man or even for any one age. 'Tis much better to do a little with certainty, and leave the rest for others that come after you, than to explain all things by conjecture without making sure of anything."

- Sir Isaac Newton, 1704

4.1 Abstract

Coastal semi-enclosed water bodies possess valuable ecosystem services that provide for human livelihoods. However, the capacity of these water bodies to buffer fluvial fluxes between the land and sea are sensitive to hydrological changes that can affect their water renewal rates. In Colombia, Cartagena Bay is of great socioeconomic relevance as it is home to one million people, some of the country's largest ports and coastal industrial zones, and Colombia's principal touristic destination. Multifactorial pollution issues causing marine ecosystem degradation have been associated with freshwater fluxes from the Dique Canal, which are projected to increase in future years due to anthropogenic activities in the watershed. This has led to plans to reduce freshwater flows by constructing upstream hydraulic doors. Given the influence of freshwater discharge on coastal water renewal, it is important to assess how these upstream changes will affect the bay's hydrodynamic processes. To evaluate this question, this study calibrated the MOHID Water model with monthly field measurements collected over a 27-month monitoring program. The 3D model was configured with a high-resolution mixed vertical discretization to capture the bay's characteristic processes of vertical stratification and mixing. A Lagrangian transport model was implemented to analyze the flow of passive particle tracers and calculate water renewal time scales. The bay's hydrodynamics demonstrated high seasonal variability as rainy and transitional seasons exhibited strong thermohaline stratification, predominantly horizontal water velocities, increased water column stability and limited vertical mixing. Conversely, the dry/windy season was characterized by less thermohaline stratification, a shallowing of the pycnocline, lesser water column stability and intensified vertical mixing. Mean residence times of 3-6 days and flushing times of 10-20 days for canal water were found, while mean residence times of 23-33 days and flushing times of 70-99 days were calculated for the bay's complete water volume. An assessment of future scenarios showed that increases in freshwater runoff caused by human development in the upstream watershed would result in faster water renewal in the bay, while plans to decrease freshwater discharge by constructing hydraulic doors would result in slower water renewal in the bay. It is therefore imperative that any plans for reducing fluvial fluxes into the bay be accompanied by the control of local pollution sources, which are abundant and could worsen the bay's water quality issues should water renewal times become longer. This modelling study makes a valuable contribution to the development of coastal management tools for Cartagena Bay and could be applied in similar polluted tropical bays in the Caribbean Region.

4.2 Introduction

Balance is a concept on which various scientific and philosophic theories are based. Physicists rely on the principle of mass conservation to balance the fluxes of pollution models, economic theories are contingent on the balance between supply and demand, while Aztec philosophers postulated the importance of living a life in balance with an ephemeral world. Unfortunately for the natural world, the concept of balance is seldom prominent in the reality of human resource use, as currently there is no country that meets the basic needs of its citizens within globally sustainable levels (O'Neill et al., 2018). Among the natural resources that are particularly vulnerable are coastal semi-enclosed water bodies, which possess valuable ecosystem services that provide for human livelihoods and wellbeing (Newton et al., 2014, 2018). One such water body is Cartagena Bay (Fig. 4.1), Colombia, where human development has led to the eradication of coral reefs and seagrass communities (Díaz & Gómez, 2003; Restrepo et al., 2006; Restrepo, 2008) and continues to cause multifactorial pollution impacts, including hypoxic conditions and drastic reductions in artisanal fisheries (Tosic et al., 2017, 2018a).

Cartagena is indeed a "hot-spot" in terms of pollution, tourism and human development. This coastal city is home to a population of one million people, has one of the country's largest ports and industrial zones, and represents Colombia's principal touristic destination. Various problems related to the waters, sediments and biota of Cartagena Bay have been observed for decades, including excessive turbidity, eutrophication, hypoxia, fecal contamination, presence of heavy metals, hydrocarbons and pesticides (FAO & CCO, 1978; Garay, 1983: Guerrero et al., 1995; Castro, 1997; Garay & Giraldo, 1997; Alonso et al., 2000; Parga-Lozano et al., 2002; Tuchkovenko & Lonin, 2003; Restrepo et al., 2006, 2016; Cañon et al., 2007; Johnson-Restrepo et al., 2008; Olivero-Verbel et al., 2008, 2009, 2013; Cogua et al., 2012; Mogollón, 2013; Jaramillo-Colorado et al., 2015, 2016; Tosic et al., 2017). While the bay's water quality issues have been linked to local domestic and industrial wastewater, as well as continental sources of pollution (Tosic et al., 2018a), foremost to these sources is runoff from the Dique Canal flowing from the Magdalena River, which is the principal source of fluvial fluxes discharging in the Caribbean Sea (Restrepo and Kjerfve, 2000; Restrepo, 2008).

Pollution issues have motivated numerous studies of the bay's water circulation and contaminant transport (Pagliardini et al., 1982; Andrade et al., 1988; Urbano et al., 1992). Such observational-based studies when combined synergistically with hydrodynamic and transport models allow a more comprehensive understanding of these phenomena (James, 2002). In Cartagena Bay, the 3-dimensional hydrodynamic model CODEGO was developed and used to study various processes: hydrodynamics and sediment transport (Lonin, 1997a; Lonin & Tuchkovenko, 1998; Lonin et al., 2004); eutrophication and oxygen regimes (Lonin & Tuchkovenko, 1998; Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003); oil spills (Lonin, 1997b, 1999; Lonin & Parra, 2005); and pollution due to phenols, fecal coliforms, grease and fats (Lonin, 2009). Other modelling studies in Cartagena Bay have also focused on its water exchange mechanisms (Molares & Mestres, 2012a; Grisales et al., 2014), water levels (Molares & Mestres, 2012b; Andrade et al., 2013, 2017), tidal dynamics (Palacio et al. 2010; Rueda et al., 2013) and sediment distribution (Restrepo JC et al., 2016).

These modelling studies succeeded in characterizing many of the bay's hydrodynamic and ecological processes. However, no previous study has quantified time scales of water renewal in Cartagena Bay, although some authors have endorsed this need (Gomez et al., 2009; Grisales et al., 2014). Water renewal time scales can effectively reflect a water body's capacity to receive pollution and provide useful information for coastal monitoring and management (Karydis & Kitsiou, 2013). Semi-enclosed coastal water bodies are also particularly vulnerable to changes in water renewal, because these water bodies play the important role of buffering fluxes of water, sediments, nutrients and organisms between the land and sea, and so changes to water renewal rates can alter the ecosystem's composition, sensitivity to eutrophication and ultimately its dissolved oxygen concentrations (Dettmann, 2001; Anthony et al., 2009; Newton et al., 2014).

There are several metrics and terms used for water renewal time scales. Two of the more practical time scales are "residence time" and "flushing time", however, as differing terminology is occasionally confusing (Jouon et al., 2006), it is important that each study states its chosen definitions. Residence time may be defined as an estimate of the time required for a single water particle to flow out of a water body (Zimmerman, 1976; Takeoka, 1984). Flushing time may be defined as an estimate of the time required to renew a water body's entire volume of water (Steen et al., 2002; Delhez et al., 2004), which has also been termed "turn over time" (Takeoka, 1984). Hydrodynamic models are commonly used to calculate water renewal time scales in semi-enclosed water bodies (Braunschweig et al., 2003; Delhez et al., 2004; Cucco & Umgiesser, 2006; Ferrain et al., 2013).

Water renewal times are important when considering mitigation strategies that could alter a water body's hydrodynamics (Lee & Park, 2003). In Colombia, there is an ongoing hydraulic intervention that plans to construct hydraulic doors along the Dique Canal and thus reduce flows of freshwater and pollution into Cartagena Bay (Fondo Adaptación, 2018). This intervention has been in discussion for at least 20 years, motivating numerous modelling studies assess the effect that reduced discharge could potentially have on the bay. Analyzing different scenarios of reduced discharge and local pollution control, it was found that reducing the canal's flow into Cartagena Bay would only improve the bay's hypoxic conditions if these actions were also combined with the elimination of local sources of industrial and domestic wastewater (Lonin & Tuchkovenko, 1998; Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003). If these local sources of pollution were to persist, reductions in freshwater discharge would simply result in greater algal blooms and further oxygen depletion, as the lack of turbid freshwater plumes would increase residence time and improve transparency, allowing primary productivity to flourish with the continued contribution of nutrients from local wastewater (Lonin, 1997a; Tuchkovenko et al., 2000). Yet unfortunately, the city's plan to redirect local wastewater sources out of the bay has fallen behind schedule and significant domestic and industrial pollution loads continue to discharge into the bay (Tosic et al., 2018a).

On the other hand, no previous study has modelled scenarios in the bay of increased freshwater discharge from the Dique Canal. Studies of the Magdalena watershed show significant increases in streamflow and sediment load which experienced increases of 24% and 33%, respectively, during the 2000-2010 period compared to the pre-2000 period (Restrepo et al., 2018). Meanwhile, the Canal del Dique witnessed increases in water discharge and sediment load of 28% and 48%, respectively. Based on these trends, hydrological modelling predictions show that water discharge and sediment flux from the Canal del Dique will increase by ~164% and ~260%, respectively, by the year 2020 when compared with the average discharge during the 2000-2010 period. These trends are in close agreement with watershed changes in land-cover and deforestation over the last three decades (Restrepo et al., 2015) and further increases in freshwater discharge could also be expected if future precipitation increases due to climate change and the intensification of the El Niño-Southern Oscillation (IPCC, 2014; Paeth et al., 2008; Restrepo et al., 2018).

In this study, we aim to demonstrate the effectiveness of applying a hydrodynamic modelling approach as a tool for the integrated management of coastal waters in a tropical bay. The objective of this approach is to use field monitored data and a calibrated hydrodynamic model to characterize the hydrodynamic processes of water renewal and vertical exchange in Cartagena Bay. The model is then applied to evaluate possible future scenarios in order to answer the following research question: How can upstream anthropogenic impacts on freshwater runoff affect the bay's hydrodynamic processes? By basing its modelling on an extensive monitoring dataset, this study provides local environmental authorities with reliable knowledge on the bay's hydrodynamics and water renewal processes along with a novel tool used to manage these coastal water resources. Further to responding to the management necessities of Cartagena Bay, which holds both socioeconomic and ecological relevance to Colombia, the demonstrated approach would be appropriate to apply in similar tropical bays throughout the Caribbean Region where pollution impacts pose an extensive threat to coastal waters, ecosystems, tourism and human health (UNEP/GPA, 2006; UNEP-UCR/CEP, 2010; Jackson et al., 2014).

4.3 Materials and Methods

4.3.1 Study Area

Cartagena Bay is a tropical semi-closed estuary located on the north coast of Colombia, adjacent to the Caribbean Sea (10°20' N, 75°32' W, Fig. 4.1). The bay has a surface area of 84 km^2, including a small internal embayment situated to the north, an average depth of 16 m, a maximum depth of 32 m, a maximum meridian length of 16 km (N-S), and a latitudinal length of 9 km (E-W). The bay is connected to the Caribbean Sea by two straits: "Bocachica" to the south and "Bocagrande" to the north. Movement through the Bocagrande strait is limited by a defensive colonial seawall 2 m below the surface. Bocachica strait consists of a shallow section with depths of 1-3 m as well as a navigation channel which is 100 m wide and 24 m deep (Tuchkovenko & Lonin, 2003; Lonin et al., 2004; Grisales et al., 2014).

Water exchange in the bay is governed by wind-driven circulation and tidal movement through its two seaward straits and the influent discharge of freshwater from the Dique Canal in the south (Molares & Mestres, 2012a). The tides in the bay have a mixed, mainly diurnal signal with a micro-tidal range of 20-50 cm and principal tidal constituents K1 and M2 with amplitudes of 9 and 7 cm, respectively (Molares, 2004). The Dique Canal discharges approximately 55-250 m^3/s of freshwater into the bay (Tuchkovenko & Lonin, 2003), the variability of which is strongly related to the seasonality of runoff from the Magdalena River, from which the Dique Canal diverges at Calamar 114 km upstream of Cartagena Bay. This freshwater discharge produces estuarine conditions in the bay characterized by a highly stratified upper water column with a pronounced pycnocline in the upper 4 m of depth, above which turbid freshwater is restricted from vertical mixing (Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003). Additional sources of freshwater discharging into the bay include wastewater from small populations around the bay and an industrial sector along the bay's east coast (Cardique & AGD, 2006). During local rainy conditions, additional freshwater also enters the bay due to runoff from a small coastal catchment area around the bay, as well as discharge from an outdated submarine outfall (near station B7 in Fig. 4.1) and backup outlets along the coast when the city's sewerage system overflows (personal communication with the city water authority, AcuaCar).

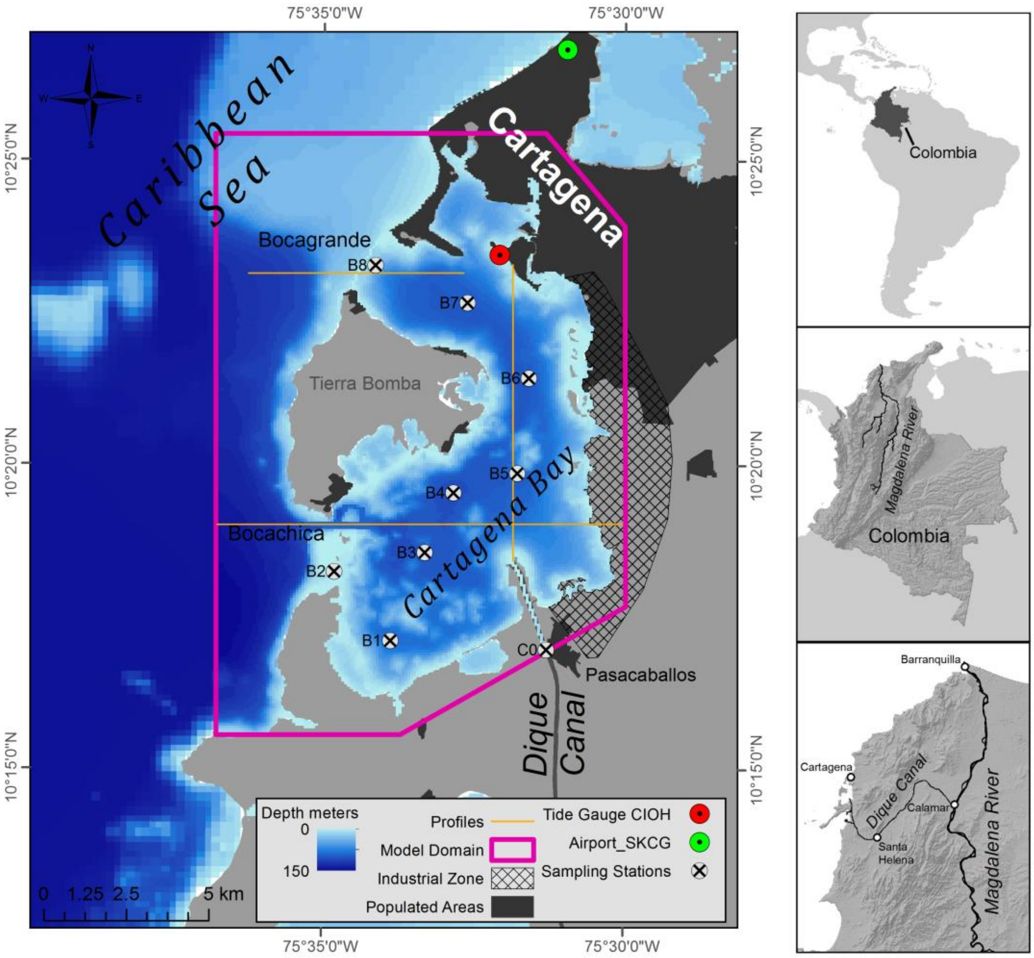


Figure 4.1. Principal panel: study area showing sampling stations (C0, B1-8), weather station (SKCG), tide station, bathymetry, model domain and profiles extracted from model results (Figs. 4.8-9). Secondary panels: location of Colombia (upper panel); location of the Magdalena River (middle panel); flow of the Magdalena into the Caribbean Sea and along the Dique Canal into Cartagena Bay (lower panel).

The bay's seasonal conditions may be categorized by the variability of winds, freshwater discharge and water quality. The Dique Canal's highest discharges typically occur from October to December, while its lowest levels occur from February to April. Winds are strongest and predominantly northerly from January to April due to the trade winds which coincide with the strengthening of the southern Caribbean upwelling system and the cooling of water temperatures (Andrade & Barton, 2005; Lonin et al., 2010; Rueda-Roa & Muller-Karger, 2013). Breezy conditions are observed from August to November, when weaker winds come from highly variable directions. Tosic et al. (2017) demonstrated the seasonal variability of the bay's water quality between the rainy season (Sept.-Dec.) when the waters are characterized by lower salinities and greater concentrations of suspended sediments, nutrients, and pathogenic bacteria; the windy season (Jan.-Apr.) which produces lower water temperatures and greater levels of organic matter; and the transitional season (May-Aug.) when the highest temperatures and lowest levels of dissolved oxygen are found.

4.3.2 Data Collection

Bathymetric data with 0.1 m vertical resolution were digitized from georeferenced nautical maps (#261, 263, 264) published by the Colombian Navy's Centre for Oceanographic and Hydrographic Research (CIOH-DIMAR). In the 3x2 km area of Bocachica strait, the digitized bathymetry was updated with high-precision (1 cm) bathymetric data collected in the field on 17 Nov. 2016. Depth was measured along 30 transects perpendicular to the navigation canal using a GPS-linked Knudsen 200 kHz mono-beam echo-sounder mounted on the side of a boat.

Water quality was monitored monthly in the field from Sept. 2014 to Nov. 2016 between the hours of 9:00-12:00 (Karydis & Kitsiou, 2013). Measurements were taken from 9 stations (Fig. 4.1), including one station in the Dique Canal (C0) and eight stations in Cartagena Bay (B1-8). At all stations, CTD casts were deployed using a YSI Castaway measuring salinity and temperature every 30 cm of depth. Grab samples were taken from surface waters and analyzed at the nearby Cardique Laboratory for total suspended solids and chlorophyll-*a* by standard methods (APHA, 1985). A Secchi disc was also deployed to measure water transparency while a Kestrel 4500 Pocket Weather Tracker was used for *in situ* measurements of winds, air temperature, and relative humidity. Crude observations of cloud cover were also recorded. At station C0 in the Dique Canal, water velocity was measured with a Sontek mini-ADP (1.5 MHz) along a cross-stream transect three times per sampling date and discharge values were subsequently calculated with the Sontek River Surveyor software.

CTD data were used to calculated the Brunt–Väisälä frequency, or buoyancy frequency, along the water column. The Brunt–Väisälä frequency (N) computes the natural frequency of oscillation of a water parcel given a small vertical displacement from its equilibrium position using the following equation: $N = \sqrt{-(\frac{\partial \rho}{\partial z} \cdot \frac{g}{\rho_0})}$, where $\frac{\partial \rho}{\partial z}$ represents the vertical density gradient, g is the acceleration due to gravity, and ρ_0 is the *in situ* density (Pingree & Morrison, 1973).

Hourly METAR data of wind speed, wind direction, air temperature and relative humidity were obtained from station SKCG at Rafael Núñez International Airport (approximately 10 km north of the bay; Fig. 4.1). Albedo, cloud cover, and precipitation data were obtained at a location 3 km offshore of Cartagena Bay from datasets available from NOAA's Global Forecast System (GFS) with 3-hour frequency. Daily profiles of temperature and salinity were obtained from the European Union's Mercator Ocean Model at a location 10 km offshore of the bay. Tidal components were obtained for numerous locations offshore of the bay from the finite element solution tide model FES2004 (Lyard et al., 2006) using the MOHID Studio software. Hourly measurements of water levels at a location within the bay (Fig. 4.1) were also obtained from the Colombian Navy's Centre for Oceanographic and Hydrographic Research (CIOH-DIMAR).

4.3.3 Model Application

The flow regime of Cartagena Bay was simulated using the MOHID Water Modelling System (Leitão et al., 2008; Mateus & Neves, 2013). The MOHID Water model is a 3D free surface model with complete thermodynamics along with Eulerian and Lagrangian transport models. MOHID is based on the finite volume approach and calculates hydrodynamic fields based on the solution of the Reynolds equations of motion, assuming hydrostatic balance and the Boussinesq approximation. The model utilizes an Arakawa C grid for its horizontal discretization which computes scalars at the center of each cell and vectors at the face of each cell. It also implements a semi-implicit time-step integration scheme and permits combinations of Cartesian and sigma coordinates for its vertical discretization (Martins et al., 2001). Vertical turbulence is computed by coupling MOHID with the General Ocean Turbulence Model (GOTM), which consists of a generic library with several different turbulence closure schemes for the parameterization of vertical turbulent fluxes in marine waters (Burchard, 2002). In this work, a scheme similar to a two equation K-epsilon solution is used, corresponding to level 2.5 of the Mellor & Yamada classification.

Model configuration for Cartagena Bay was based on an equally-spaced Cartesian horizontal grid with a resolution of 75 m and a domain area of 196 km^2 (Fig. 4.1). This included an offshore area 2.3 km outside the bay, though only the results inside the bay are considered within the limits of the monitoring stations used for calibration. A mixed vertical discretization of 22 layers was used by incorporating a 7-layer sigma domain for the top five meters of depth and a variably-spaced (depth-incrementing) Cartesian domain below that depth. Depths within the bay were covered by 18 of the 22 vertical layers, while the bottom four layers covered the greater depths outside the bay. This was possible due to the generic vertical geometry of MOHID and was chosen to reproduce the mixing processes of the highly stratified bay. Given these defined dimensions, the MOHID Studio software was used to generate a bathymetric grid based on the digitized nautical maps that were updated with field-measurements in the area of Bocachica strait.

The model includes the main factors determining the circulation within the bay. The horizontal density gradients occurring within the bay are influenced by the freshwater input of the Dique Canal, the tide-driven input of seawater at the model domain's open boundary, and atmospheric fluxes of heat (solar radiation, longwave radiation, latent heat and sensible heat) and water (precipitation, evaporation) at the bay's surface. The gradient of water level is influenced by lateral tidal forcing and elevated water levels created by freshwater accumulation and wind-driven "pile-up". The horizontal gradient of atmospheric pressure is ignored. The Coriolis force is considered, though its effect is minimal in this case given the low latitude of 10°20' N. Wind stress is applied to the surface layer as a boundary condition.

Open boundary conditions were imposed by prescribing values based on reference solutions (Dirichlet boundary conditions). The lateral forcing of the tides was determined by the harmonic components extracted from the FES2004 model at numerous locations along the boundary. Daily profiles of seawater temperature and salinity extracted from the Mercator model were imposed at the open sea boundary. Measured monthly values of discharge and velocity for the inflowing Dique Canal were imposed as a time series from which the model interpolates values of water flow and momentum for each time step during the simulation period. Atmospheric variables were all prescribed as spatially-constant fields with frequencies of 1-hour (METAR data: wind velocity, air temperature, relative humidity) or 3-hours (GFS data: albedo, cloud cover, precipitation data). These meteorological datasets were compared with field observations for verification. The evaporation rate and heat fluxes (solar and non-solar radiation) at the bay's surface were calculated by the model.

Given the importance of surface water transparency on short-wave radiation penetration and the resulting heat flux, multiple calculation methods of the short-wave light extinction coefficient (K_d) were compared utilizing the field measurements of total suspended solids (TSS), chlorophyll-*a* and Secchi depth. These included previously established relationships between K_d-TSS and K_d-Secchi in Cartagena Bay (Lonin, 1997a), K_d-TSS and K_d-Secchi relationships in coastal and offshore waters around the United Kingdom (Devlin et al., 2008), a relationship for K_d-chlorophyll-*a* in oceanic waters (Parsons et al, 1984), a K_d-TSS relationship developed for the Tagus estuary, Portugal (Portela, 1996), and a combined K_d-TSS-chlorophyll-*a* relationship based on Portela (1996) and Parsons et al. (1984). The K_d-TSS relationship of Portela (1996) was finally selected for use in this study as model simulations using these K_d values produced thermoclines most similar to field observations.

Three simulation periods of a lunar-month duration in the year 2016 were focused on for the present modelling study, each considered representative of distinct seasonal conditions in the bay (Tosic et al., 2017): the windy season (27 Jan. – 24 Feb.), the transitional season (28 June – 26 July), and the rainy season (19 Oct. – 17 Nov.). Start and end times for each simulation coincided with the dates of monthly field sessions. After testing various time steps, an optimal Δt of 20 seconds was chosen. Initial conditions of water temperature and salinity were defined as vertical profiles based on CTD measurements made on the corresponding start date. To avert numerical instabilities, a spin-up period of one day was applied to gradually impose wind stress and open boundary forcings (Franz et al., 2016).

Outputs from the simulations were then processed, analyzed and compared with field measurements. Each simulation's final water temperature and salinity were extracted as profiles at each sampling station location (B1-B8; Fig. 4.1) and compared to the CTD measurements of the respective end date. Measurements from the CIOH tidal gauge were compared to hourly time series of water height extracted from each simulation's output at the gauge's location (Fig. 4.1). Instantaneous and mean water velocity fields and water heights were also analyzed from each simulation's output.

To evaluate model performance, various descriptive statistics were calculated to compare the simulated (predicted: *P*) and measured (observed: *O*) values of water temperature, salinity and water height. Being N_s the sample size, these included the mean of observed ($\bar{O}$) and predicted ($\bar{P}$)

values, and the sample standard deviation of observed (s_O) and predicted (s_P) values (Willmott, 1982). The average error (AE = $\Sigma(P_i - O_i)/N_s$) was calculated as a measure of aggregate model bias in order to identify the model's tendency to over- or under-estimate observed values, while keeping in mind that positive and negative discrepancies can cancel one another (Zhang & Arhonditsis, 2008). The magnitude of the model's prediction accuracy is shown by the mean absolute error (MAE = $\Sigma|P_i - O_i|/N_s$)) and the root mean squared error (RMSE = $(\Sigma(P_i - O_i)^2/N_s)^{0.5}$), the latter of which is sensitive to the inaccuracies of outliers (Willmott & Matsuura, 2005). The relative error (RE = $\Sigma|P_i - O_i|/ \Sigma O_i$) was also calculated in order to express error as a percentage of the observed values (Arhonditsis & Brett, 2004), noting that this measure is dependent on the magnitude of the variable itself.

Model calibration and validation were based on the analysis of the aforementioned statistics for each of the three simulations. The rainy season simulation was utilized for calibration, while the windy and transitional season simulations were used for validation. In order to select calibration parameters, a sensitivity analysis was carried out to assess the influence of parameters with uncertain values on model results. Simulations were run in which each of the following parameters were reduced by one half and increased two-fold (Arhonditsis et al., 2000): horizontal viscosity, surface water roughness and bottom roughness. Additionally, two other horizontal advection methods were tested: 2nd order upwind and Total Variation Diminishing. The effect that each parameter modification had on the salinity, temperature and water level (quantified by the aforementioned statistics) was then compiled and compared to determine the most important parameters for model calibration. Horizontal viscosity and bottom roughness (used to compute the drag coefficient) were ultimately selected as calibration parameters, yielding final calibrated values of 12.0 m^2/s and 0.006, respectively.

A Lagrangian approach was used to compute residence times and flushing times of the water in Cartagena Bay during the different seasonal simulations (Braunschweig et al., 2003). MOHID's Lagrangian transport model was used to trace the evolution of passive particles emitted at slack water tide from two origins: 1) the Dique Canal and 2) throughout the bay itself. To assess the residence times of canal water in the bay, a total of 1,080 particles were emitted from the canal over a period of six hours. To assess the residence time and flushing time of the full volume of water in the bay, another 320,657 particles were instantaneously emitted throughout all areas and depths of the bay (1 particle per cell). Both particle emissions were set to occur four days following the start of the simulation in order to postpone particle evolution until after the initial phase of hydrodynamic stabilization. Monitoring boxes set up in the model tracked the total volume of particles during the simulation in three layers: the surface (top 5 m of depth), mid-layer (5-14 m depth) and bottom layer (14-32 m depth). Given the probability that a few particles may become trapped in small inlets within the bay, for practical purposes flushing time was defined as the time necessary for 95% of the particles to flush out of the bay (Jouon et al., 2006). As the particles emitted throughout the bay (origin 2) did not meet this 95% criteria within the time period of the simulations, exponential curves were fitted to the particle evolution trend in order to calculate the bay's flushing time (Monsen et al., 2003). These curves were also used to estimate a mean residence time for the bay's water given as τ in the exponential equation: $m(t)=m(0)exp(-t/\tau)$, which is the time at which the volume of tracers is reduced to approximately 36.8% ($1/e$) of its initial volume (Braunschweig et al., 2003). This methodology was chosen to evaluate the magnitude and importance of both seawater renewal and vertical exchanges in the bay.

Additionally, numerical experiments were carried out with the model to evaluate two hypothetical scenarios. The first scenario consisted of doubling the freshwater discharge from the Dique Canal in consideration of hydrological trends projecting significant future increases in runoff associated with watershed deforestation and climate change (Restrepo et al., 2018). The other scenario entailed halving the canal's discharge in order to reflect the reduced flow planned by the ongoing hydraulic intervention, which proposes to construct hydraulic doors along the canal (Fondo Adaptación, 2018). For each of the six numerical experiments (two scenarios, three simulated seasons), residence and flushing times were calculated using the Lagrangian approach and water velocity fields were analyzed in order to evaluate the simulated impact on the bay's hydrodynamics.

4.4 Results

4.4.1 Monitoring Results

Over the 27-monthly monitoring sessions, salinity, temperature and density in the bay ranged from 0-36.5, 26.7-33.2°C and 995.9-1023.6 kg/m^3, respectively. The bay's minimum values of salinity and density were found at the surface of the Dique Canal outlet from Oct.-Dec. 2014. Maximum temperatures were found at the surface of station B5 in Oct. 2015. Among surface waters, the maximum salinity (35.8), minimum temperature (27.6°C) and maximum density (1022.8 kg/m^3) values were found at station B8 (Bocagrande) in Jan. 2015. Bottom waters were much more homogenous horizontally, with multiple stations presenting the maximum density and salinity values in April and May, respectively, of 2016. Measurements in bottom waters yielded minimum values of salinity (35.3) and density (1022.1 kg/m3), along with maximum temperature (29.2°C) at multiple stations in Nov. 2016. Overall minimum temperatures were found in the bay's bottom waters in March 2015.

Excluding the Dique Canal outlet itself, only minor horizontal spatial variation is observed among the different locations (B1-8) of Cartagena Bay (Fig. 4.2). A greater influence of freshwater, in terms of lower salinity, density and higher temperature, was found at stations B4 and B5 in front of the Dique Canal. Conversely, a stronger influence of marine water was seen at the furthest station (B8) from the canal.

Temporal and vertical spatial variations were remarkably pronounced at all stations. Graphical visualizations are focused on the results of station B5 (Fig. 4.3) which exhibited slightly more variability than the other stations due to the canal's proximal freshwater influence. Well-defined seasonal conditions were observed with vertically mixed marine conditions during the windy season (Jan.-April) and highly stratified gradients of salinity and temperature in the rainy (Sept.-Nov.) and transitional (May-Aug.) seasons, respectively.

Inter-annual variation was also apparent, reflecting the influence of the El Niño phenomena of 2015, when reduced precipitation and runoff resulted in higher temperatures and salinities in the bay. Of particular note are the surface temperatures occurring between Aug.-Oct. 2015 (31.5-33.2°C) which were greater than the corresponding months in 2014 and 2016 (avg. 30.8 ± 0.2°C). The effect of less runoff during 2015 is also visible in the vertical salinity gradients, as the rainy season influence of freshwater during Sept.-Oct. 2014 and Oct.-Nov.2016 reaches greater depths than in the 2015 rainy season.

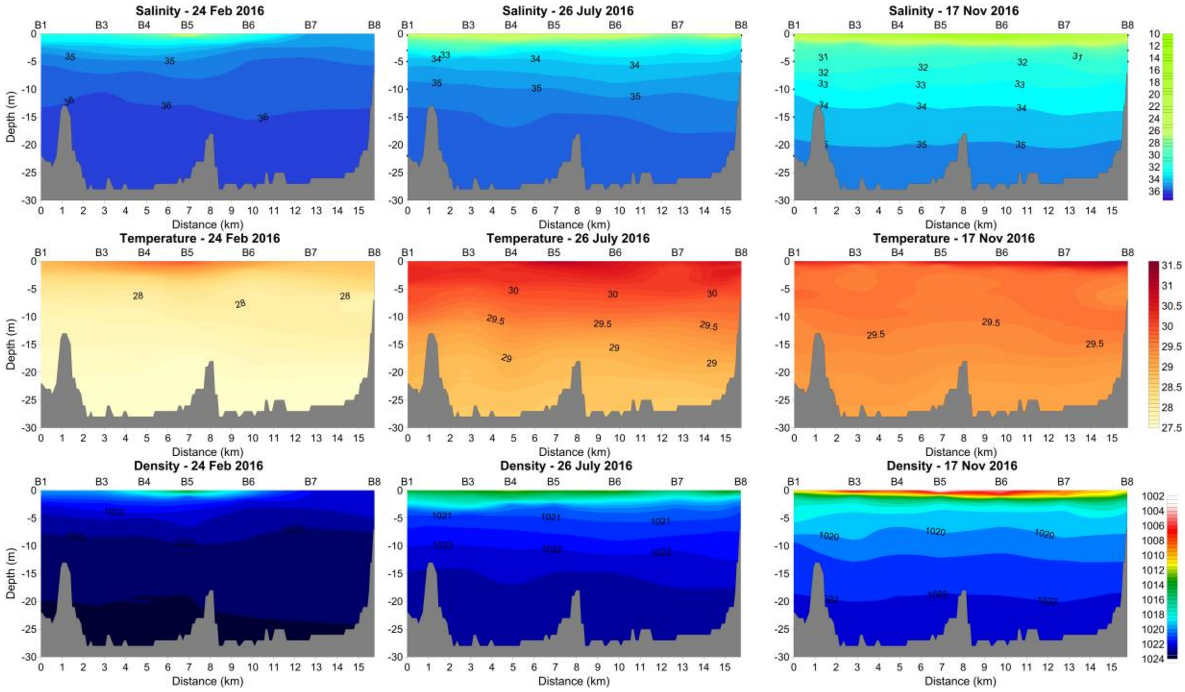


Figure 4.2. Vertical profiles of salinity (top), temperature (middle) and density (bottom) measured during the windy (24-Feb; left), transition (26-July; center) and rainy (17-Nov; right) seasons of 2016 and horizontally interpolated between sampling stations (B1-8).

The effects of surface heating and water mass fluxes can be distinctly observed in the vertical density profiles. Temporal variation of surface heating, particularly between April and October 2015 but also from May to July 2016, results in greater vertical density gradients and lower bottom densities. Abundant freshwater surface fluxes between September and November have a similar effect along the vertical density gradient, though the influence of freshwater can also be seen in the salinity and density of surface waters in April and May.

The seasonal variation of vertical mixing is well-pronounced as less thermohaline stratification is observed between January and March. This is due to increased wind speeds during this season as well as to the effect that decreased freshwater input has on reducing the pycnocline, which in turn allows wind-generated turbulence at the surface to penetrate lower depths. Cooler bottom waters are also likely related to the strengthening of the southern Caribbean upwelling system (Andrade & Barton, 2005; Rueda-Roa & Muller-Karger, 2013). The overall effect of this thermohaline variation results in the bay's water density gradually decreasing from April to November and then in December sharply reverting to conditions more similar to seawater.

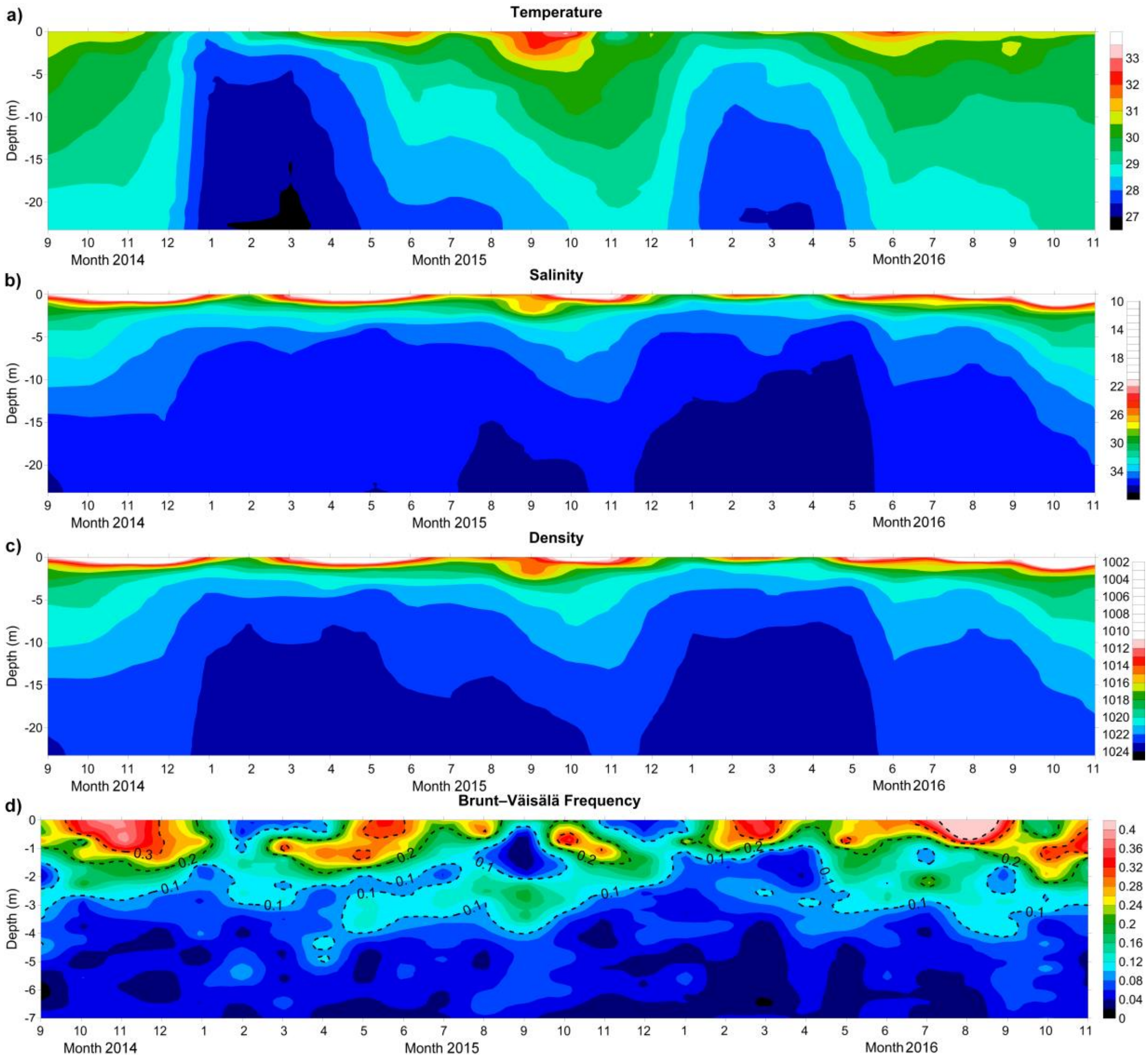


Figure 4.3. Hovmöller diagrams of vertical profiles of temperature (a), salinity (b) and density (c) measured at station B5 during 27 monthly sampling sessions (Sept. 2014 – Nov. 2016) and interpolated over time. Calculations of the Brunt-Väisälä frequency (d) are shown for the top 7 m of surface water.

There also appears to be a time-lag between surface and bottom effects. For example, decreased surface densities in May and October, caused by freshwater runoff, are not transferred to the bottom waters until June and November, respectively. This is reflective of the time required for vertical mixing and the impeding effect of strong vertical stratification.

Brunt–Väisälä frequencies further illustrate the processes of vertical mixing (Fig. 4.3d). Higher frequencies are indicative of steeper density gradients where there is resistance to vertical mixing, while lower frequencies are characteristic of a stratification closer to neutral, facilitating the vertical displacement of turbulent eddies and increasing the efficiency of vertical mixing.

Measurements between Sept. 2014 and Nov. 2016 revealed greater Brunt–Väisälä frequencies in surface layers (Fig. 4.3d). Lower frequencies below 1 m depth from Jan.-Apr. 2016 demonstrate the greater potential for vertical mixing during the windy season. Maximum frequencies were typically found within the top 1 m of surface water with the exception of Feb. and Sept. 2015 when maxima were found 1.5 and 2.5 m depths, respectively. In Feb. 2015, the depth of this maximum value may be a reflection of reduced freshwater input and greater vertical mixing during this month which led to higher salinities being found at the surface. In contrast, the deeper maximum value found in Sept. 2015 could be a result of fluxes of heat and freshwater which penetrated deeper below the surface than usual (Fig. 4.3a, b), producing a homogeneously mixed surface layer, a corresponding frequency minimum near the surface, and the vertical displacement of the pycnocline down to 2.5 m depth.

4.4.2 Model Performance

The calibrated model adequately reproduced field observations of temperature, salinity and water level, as shown by the different performance statistics analyzed (Table 4.1), indicating that the model effectively simulates the hydrodynamics of the system. Mean values of temperature, salinity and water level observed during the three simulation months were in close agreement with mean values produced by the model, yielding differences of 0.05-0.26°C and 0.15-0.53 in temperature and salinity, respectively, and just a 1 mm difference in mean water levels. Similar standard deviations between observed and predicted values show that the model captured the observed spatio-temporal variation. The standard deviations of observed data were consistently slightly greater than the standard deviations of modelled values. This may be expected due to the variety of factors in the natural environment which can generate additional variability (e.g. waves, storms, urban wastewater) that are not considered in the model.

Table 4.1. Statistics of model performance: sample size (N_s), mean and standard deviation of observed ($\bar{O}$, s_O) and predicted ($\bar{P}$, s_p) values, average error (AE), mean absolute error (MAE), root mean squared error (RMSE), and relative error (RE).

Parameter	Season	N_s	$\bar{O}$	$\bar{P}$	s_O	s_P	AE	MAE	RMSE	RE
Temperature (°C)	**Rainy**	108	29.62	29.57	0.26	0.18	-0.05	0.21	0.28	0.7%
	Tran	108	29.85	29.59	0.52	0.30	-0.26	0.36	0.39	1.2%
	Dry	108	28.23	28.05	0.49	0.32	-0.17	0.20	0.29	0.7%
Salinity	**Rainy**	108	30.54	31.03	4.49	3.71	0.49	1.36	2.13	4.5%
	Tran	108	33.04	32.89	2.83	2.82	-0.15	0.80	1.09	2.4%
	Dry	108	34.99	35.52	1.40	0.62	0.53	0.60	1.03	1.7%
Water Level (m)	**Rainy**	496	0.390	0.386	0.112	0.092	-0.001	0.034	0.044	8.8%
	Tran	625	0.284	0.283	0.110	0.094	-0.001	0.031	0.039	10.9%
	Dry	277	0.338	0.342	0.097	0.092	0.001	0.039	0.048	11.4%

Hourly observations of water level were in very close agreement with those generated by the model (Figure 4.4). Despite the occurrence of periods with missing observational data, calibration of the mean water level allowed for closely corresponding tidal oscillation in the model results. Small differences between observed and modelled water levels were prevalent at peaks of high- and low-tides. Nevertheless, values of MAE (0.031-0.039 m) and RMSE (0.039 – 0.048 m) were quite low, suggesting that these differences were of minimal consequence. While values of RE may appear high (8.8-11.4%), this is relative to the small magnitude of water levels in general.

Temperature profile observations were particularly well reproduced by the model in the rainy and windy season simulations (Figure 4.5), which yielded MAE and RMSE values of 0.20-0.21°C and 0.28-0.29°C, respectively. The transitional season simulation produced slightly higher values of 0.36°C and 0.39°C, respectively, which may be a result of this season having a more dynamic thermal structure as it also yielded the highest mean (29.85°C) and standard deviation (0.52°C) of observed temperatures. In general, the model showed an overall bias of underestimating temperatures as AE values were all negative (Table 4.1). However, the plots of observed and predicted temperature profiles (Figure 4.6) show that this underestimation commonly occurred at shallower depths, while temperature at deeper depths was usually overestimated. Therefore, the model produced temperature profiles that were less vertically stratified than in the observed profiles. Regardless, the low values of MAE, RMSE, and RE (0.7-1.2%) showed that temperature was adequately reproduced by the model, especially considering that the model's vertical resolution (71-78 cm at the surface, depending on the tide) cannot completely reproduce the strong vertical gradients at the surface.

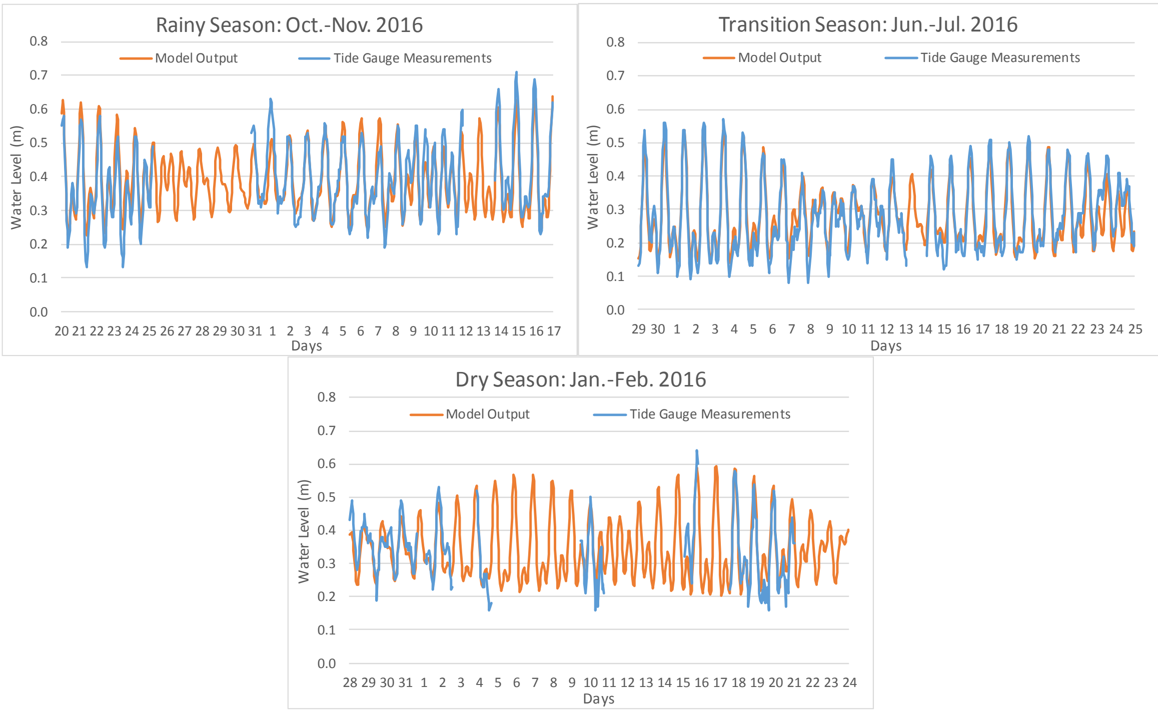
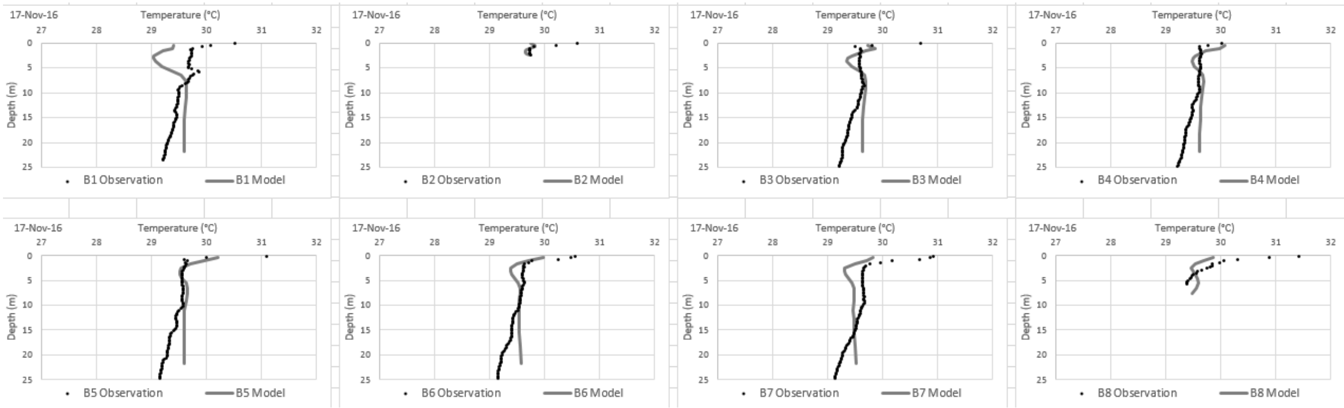


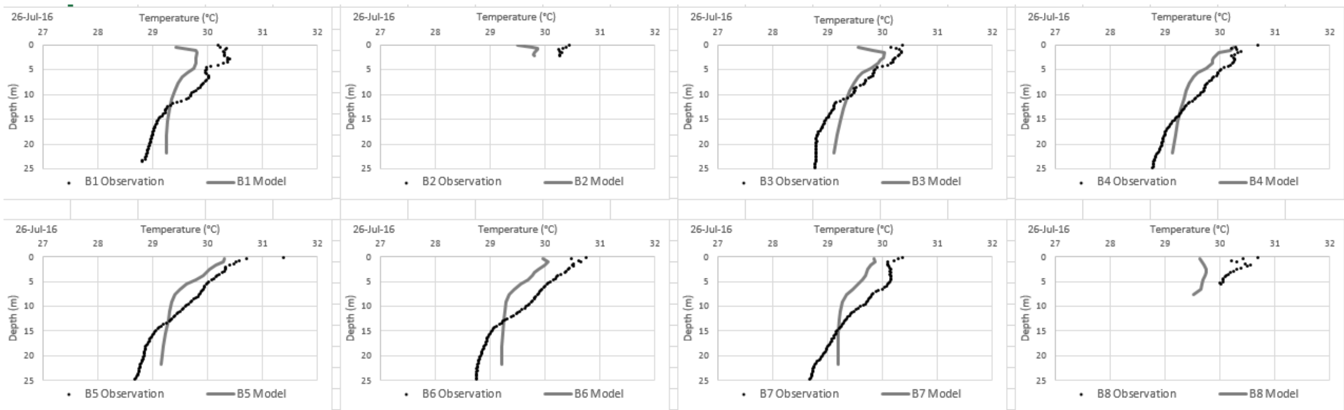
Figure 4.4: Model results compared to measurements of water height during three simulations (rainy, transition, windy seasons) at the tide gauge in Cartagena Bay (Fig. 4.1).

Predicted salinity profiles produced higher RE values (1.7-4.5%) than those of temperature profiles. However, visual comparison of the predicted and observed salinity profiles (Figure 4.6) suggest that the model was quite effective in simulating the bay's haline stratification, as modelled salinity profiles were very similar to observations. The principal source of error can be seen to occur at the surface where observed salinities are occasionally much lower than predicted values. This overestimation in surface water salinity is to be expected given that various additional sources of freshwater input are not included in the model, such as urban runoff and local sources of domestic and industrial wastewater. This discrepancy is reflected in values of RMSE, a statistic which tends to amplify the effect of outliers, yielding greater error in the rainy season (2.13) than in the other two season (1.03-1.09).

Rainy Season: Oct.-Nov. 2016



Transition Season: Jun.-Jul. 2016



Windy Season: Jan.-Feb. 2016

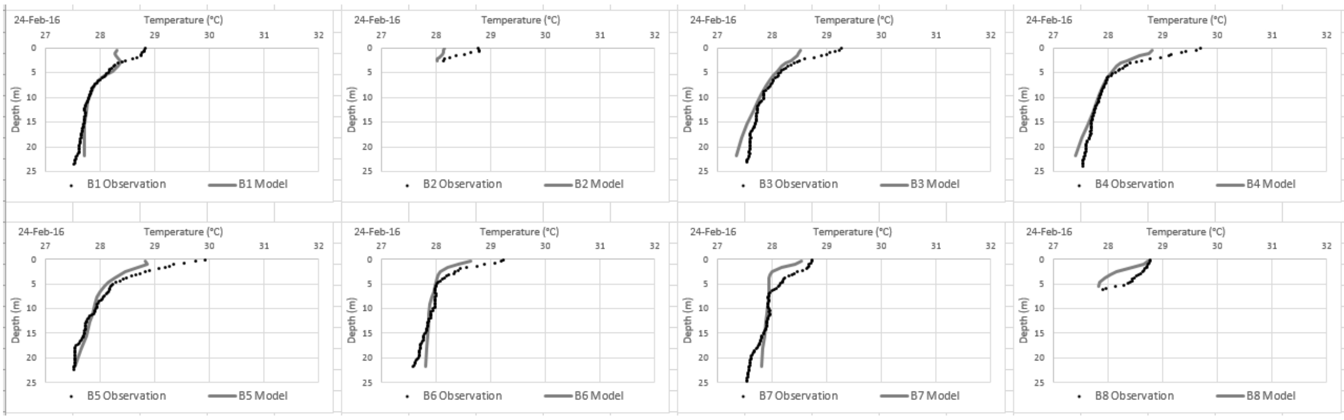
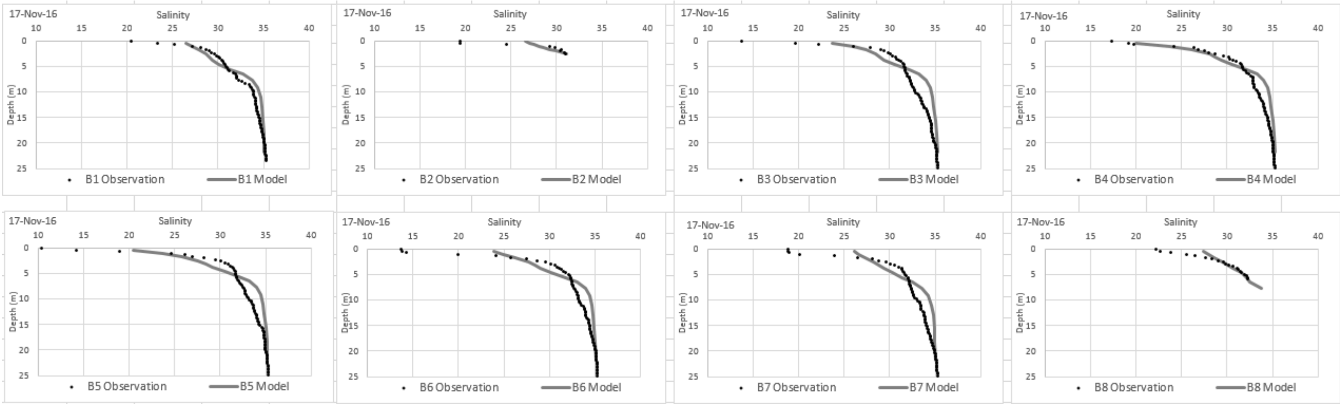
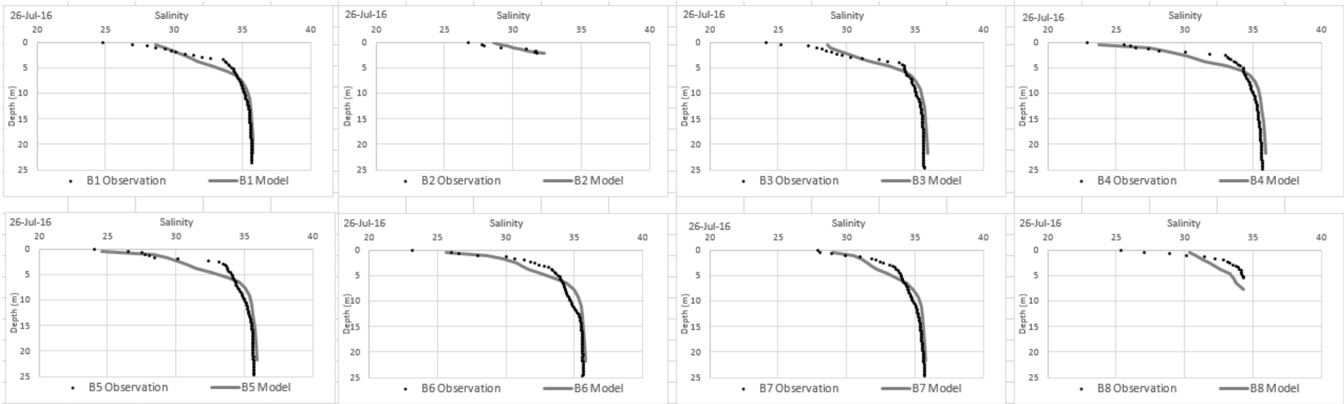


Figure 4.5: Model results compared to measurements of temperature profiles at the end of three simulations (rainy, transition, windy seasons) at eight stations in Cartagena Bay (B1-B8)

Rainy Season: Oct.-Nov. 2016



Transition Season: Jun.-Jul. 2016



Windy Season: Jan.-Feb. 2016

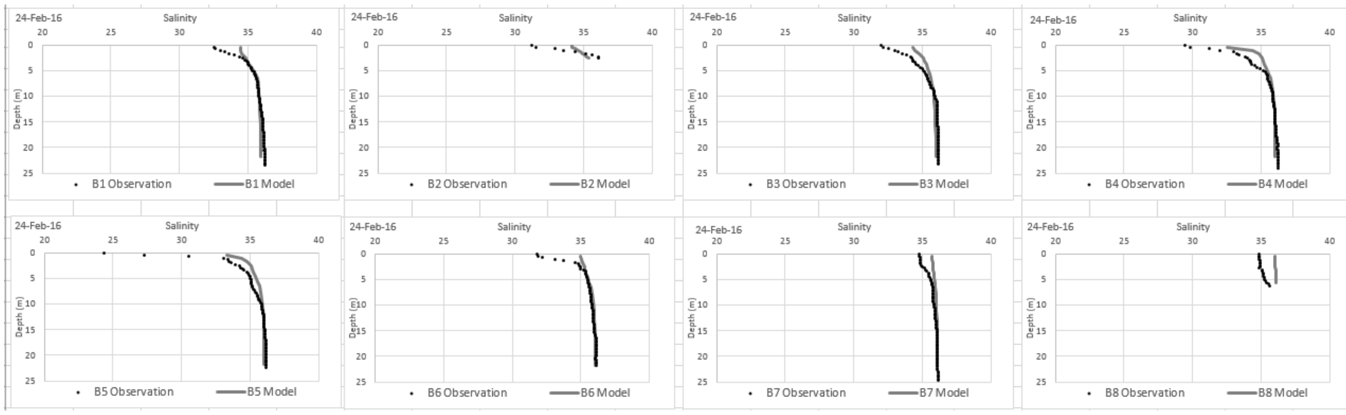


Figure 4.6: Model results compared to measurements of salinity profiles at the end of three simulations (rainy, transition, windy seasons) at eight stations in Cartagena Bay (B1-B8)

4.4.3 Hydrodynamic Modelling Results

The mean water levels during each simulation showed slight spatial variation across the bay with differences of approximately 1.2 cm (Fig. 4.7). The most pronounced variation of mean water level was observed during the windy season simulation when strong northerly winds generated a north-south gradient with higher mean water levels at the southern end of the bay than at the northern end. The rainy season simulation resulted in higher mean water levels in the central part of the bay due to the accumulation of freshwater discharge from the Dique Canal. Meanwhile, during the transitional season, a combined effect is observed with elevated water levels found both around the Dique Canal's outlet area and in the bay's southern end. Seasonal variability was also observed as mean water levels were greatest in the rainy season (38.0-39.2 cm), followed by the windy season (34.1-35.3 cm) and lowest during the transitional season (27.9-29.1 cm) with respect to the mean lower low water (MLLW).

Mean surface water velocities during the three seasonal simulations (Fig. 4.7) exhibit the dominant influence of discharge from the Dique Canal on the bay's surface currents. At high (rainy season) and medium (transitional season) discharge levels, the Dique Canal forces surface currents to run north and west to the two seaward straits. Mean surface currents during the transitional season are stronger in the westward direction and weaker in the northward direction as a result of winds coming predominantly from the north. During the windy season, strong northerly winds overpower the low discharge levels and force mean surface currents throughout the bay in a southwest direction. The effect of windy season conditions on the bay's seaward outflow can be seen as mean surface currents flow out of the southern strait, Bocachica, but appear to flow parallel to the seawall in the northern strait, Bocagrande, resulting in reduced seaward exchange and recirculation of bay water.

Vertical profiles of mean water velocities during the three simulations illustrate some of the bay's processes of circulation and water exchange with the sea (Fig. 4.8). In all cases, the pronounced vertical stratification generates distinct surface and sub-surface currents that form the estuary's gravitational circulation. At Bocachica strait (Fig. 4.8, top), freshwater flows out from the bay at the surface, while seawater flows in via the navigation canal below. While these currents in the rainy and transitional season simulations are quite similar, the stronger westward surface velocities during the transitional season result in higher velocities flowing into the bay in the bottom depths of the navigation canal. During the windy season simulation, outflowing currents run deeper into the navigation canal (0-6 m depths) than in the other two seasons (0-3 m depths), resulting in a deepening of the inflowing seawater current as well. This deepening of outflowing water may be expected during the windy season given that thermohaline stratification is reduced and that mean surface outflow was concentrated through Bocachica (Fig. 4.7).

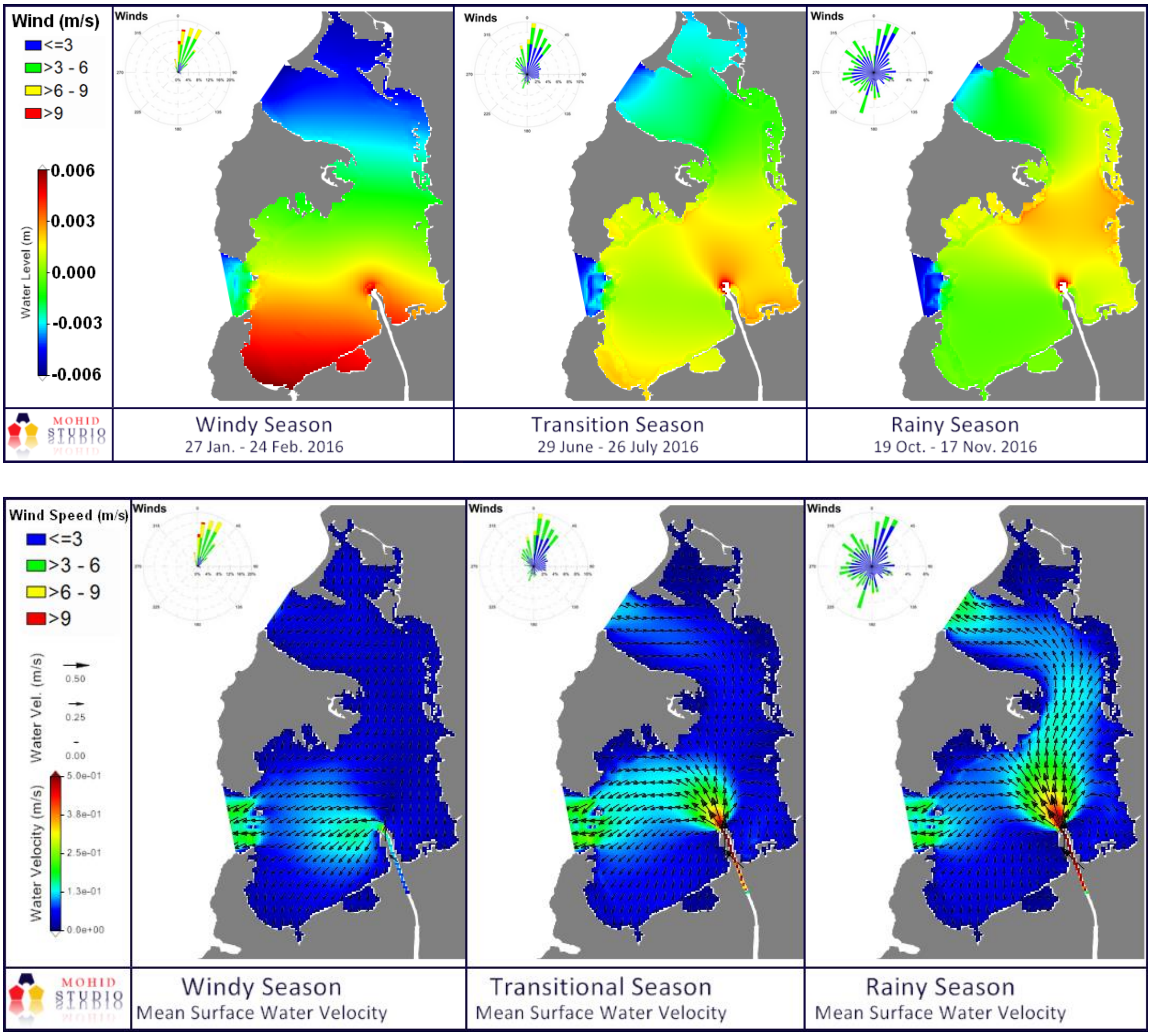


Figure 4.7. Mean water level (above), mean surface water velocities (below) and wind roses during simulations of the windy (left), transition (center) and rainy (right) seasons of 2016.

At Bocagrande strait (Fig. 4.8, middle), outflowing surface water and inflowing sub-surface water were also observed, though the latter was inhibited by the sub-surface seawall. Again the currents in the rainy and transitional season simulations were quite similar, though in this case the stronger northward surface velocities during the rainy season (Fig. 4.7) were balanced by faster sub-surface inflow atop the seawall. Velocity fields during the windy season are quite different as little inflow or outflow is observed over the seawall (Fig. 4.8); but rather, surface currents plunge downwards on the inner side of the seawall and create a vertical circulation cell.

Distinct windy season currents were also evident in vertical velocity profiles along the north-south central axis of the bay (Fig. 4.8, bottom). During the windy season, surface currents ran southwards and two sub-surface vertical circulation patterns are observed in opposite directions: one in the north part of the bay at a depth of approximately 5 m and another in the south part of the bay at approximately 8 m depth. During the rainy and transitional seasons, northward surface currents were balanced by southward sub-surface currents, though in the case of the transitional season this sub-surface current was slightly deeper than during the rainy season. Mean bottom currents ran northwards during the rainy and transitional seasons, and southwards during the windy season. A distinct column of upward and downward vertical exchange in the center of the bay was detected in the rainy and transitional seasons (Fig. 4.8).

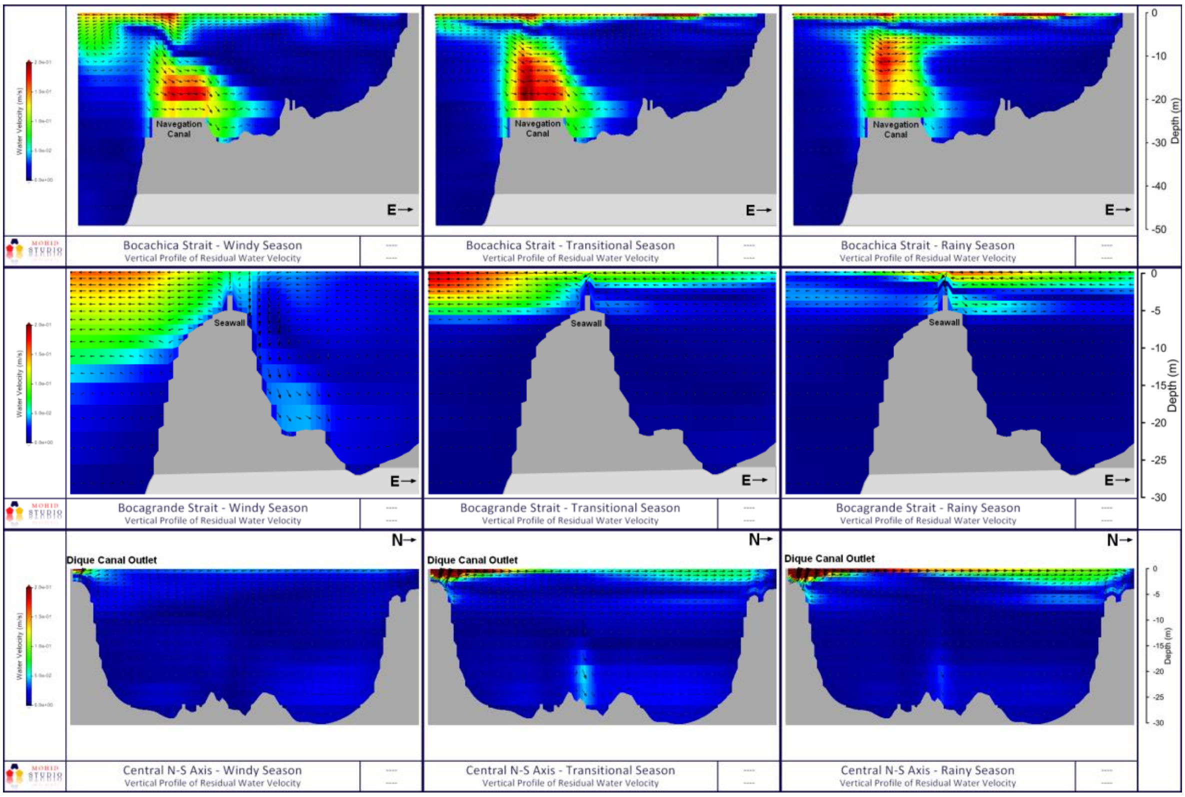


Figure 4.8. Vertical profiles of mean water velocities along the latitudes of the Bocachica navigation canal (above), Bocagrande strait (middle) and along the longitude of the Dique Canal outlet and central part of the bay (below) for simulations of the windy (left), transition (center) and rainy (right) seasons. See Fig. 4.1 for locations of profiles.

Mean water velocities revealed a sub-surface horizontal eddy in the southwest part of the bay. At 2.5 m depth, this eddy flowed counter-clockwise during all three seasons as the northern part of the eddy contributed to the surface outflow at Bocachica. At 5.5 m depth, the eddy was reversed to a clockwise circulation pattern during the rainy and transitional season, as the northern part of the eddy corresponded to seawater inflow through Bocachica at this depth. However, during the windy season, the deepening of the surface outflow at Bocachica also resulted in a deepening of the counter-clockwise eddy which was maintained at 5.5 m depth, while the clockwise eddy was found at deeper depths of approximately 10 m.

Deviations from mean water velocities were exhibited by instantaneous water velocity fields. The direction of instantaneous surface velocities during the rainy and transitional seasons occasionally varied when moderate winds came from the north or west in opposition to the northward and westward flowing surface waters, though velocities around the Dique Canal outlet were consistently in a north-northwest direction. Instantaneous surface velocities in the two straits during the rainy and transitional seasons were consistently in a seaward direction and only rarely changed direction when flood tides coincided with moderate westerly winds. In the windy season, instantaneous surface water velocities typically followed the wind direction throughout the bay, including the Dique Canal outlet area, but with the exception of the Bocachica strait where a seaward surface outflow was maintained.

The effect of tidal currents was clearly observed in vertical profiles of instantaneous velocities in Bocachica strait where the speeds at all depths and direction at sub-surface depths (3-5 m) oscillated with the tides in all three seasonal simulations. During flood tides, bottom waters flowing into the bay increased in speed and occupied a greater proportion of the navigation canal's water column. Ebb tides resulted in increased speeds of outflowing surface waters that extended to deeper depths of the water column. Occasionally, a layer of outflow was also observed at bottom depths of the navigation canal during the rainy and transitional seasons. In the windy season, a vertical eddy was sometimes observed in the navigation canal between the layers of surface outflow and sub-surface inflow.

A similar tidal oscillation was observed in the instantaneous velocities in Bocagrande strait. In the rainy and transitional seasons, surface water outflow intensified during ebb tides. Occasionally, the intensified surface outflow during ebb tides in the rainy season dominated water exchange in the strait such that outflowing currents were observed at all depths. During flood tides, sub-surface inflow intensified in the rainy season, though in the transitional season, sub-surface currents atop the seawall were characterized by the downwelling and recirculation of surface waters. During the windy season, the vertical circulation cell found in mean water velocities on the inner side of Bocagrande (Fig. 4.8) was maintained throughout nearly the entire simulation, while the waters atop the seawall oscillated between inflow and outflow at all depths uniformly.

Along the north-south central axis of the bay, vertical profiles of instantaneous water velocities (Fig. 4.9) were very different from mean water velocities during the windy season. While mean water velocities portrayed a velocity field characterized by low speeds (Fig. 4.8, bottom), instantaneous water velocity fields revealed fluctuating hydrodynamics with high speeds. Numerous vertical circulation cells were frequently observed at varying locations while current directions changed often. A column of upward and downward vertical exchange in bay's center was also detected, similar to that of mean water velocities during the rainy and transitional seasons.

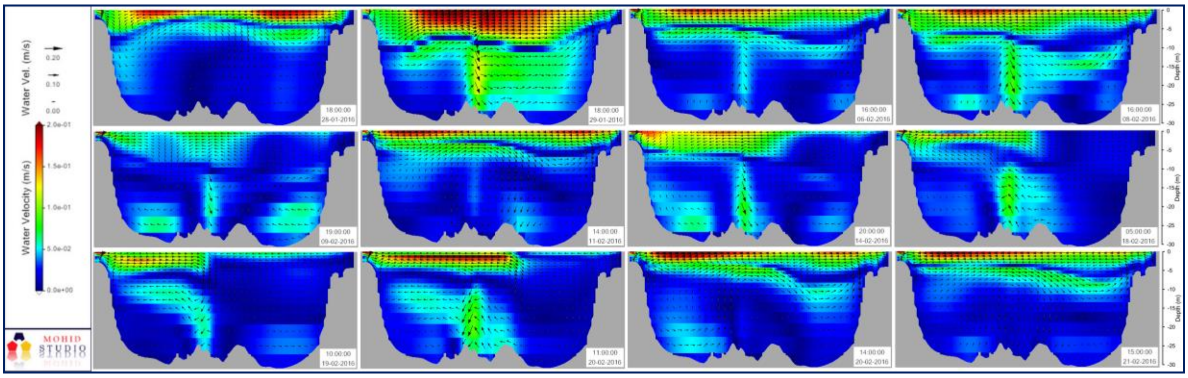


Figure 4.9. Snapshots of vertical profiles of instantaneous water velocities along the longitude of the Dique Canal outlet and central part of the bay (see Fig. 4.1 for location) during the windy season simulation. The Dique Canal outlet is to the left and north is to the right.

4.4.4 Water Renewal Time Scales

Under the present conditions, the mean residence time of particle tracers emitted from the Dique Canal varied between 3-6 days, while their flushing time ranged from 10 to 20 days (Table 4.2). During the rainy and transition seasons, particles were transported north due to the canal's influence, though their extension was slightly reduced during the transition season due to lower discharge and stronger wind (Fig. 4.10). In the windy season, intense northerly winds confined particles in the southern lobe of the bay. The shorter residence time of canal water during the rainy season (2.7 ± 4.2 days) is reflective of the greater surface water velocities resulting from the high discharge and weak variable winds of the rainy season (Fig. 4.7). The constraining effect of the wind was particularly apparent in the flushing time of the windy season (19.9 days) when compared to the shorter flushing times of transitional (13.2) and rainy (10.2) seasons.

For particle tracers emitted throughout the bay's complete volume of water, the mean residence time varied between 23-33 days while the overall flushing time ranged from 70 to 99 days. In this case, the residence (32 days) and flushing times (97-99 days) of the rainy and transitional seasons were quite similar (Table 4.2). However, the shorter time scales during the windy season are indicative of the increased vertical mixing during this season bringing particles from lower depths up to the surface where they can flow out of the seaward straits. Processes of vertical exchange and the shorter flushing time during the windy season are evident in Figure 4.11, illustrating the oscillation of particles between surface and sub-surface depths in unison with the tides.

Table 4.2. Mean residence times (± standard deviation) and flushing times (in days) of particles emitted from the Dique Canal and the entire bay during three seasons under three scenarios: the present, double-discharge (Qx2) and half-discharge (Qx0.5) of water from the Dique Canal. Environmental parameters of canal mean discharge values (m^3/s) along with mean wind speed (m/s) and direction (°) are also presented.

	Rainy Season			Transition Season			Windy Season		
	Present	Qx2	Qx0.5	Present	Qx2	Qx0.5	Present	Qx2	Qx0.5
Environmental Parameters									
Mean Canal Discharge (m^3/s)	225.0	450.0	112.5	167.3	334.7	83.7	33.9	67.7	16.9
Mean Wind (m/s) & Dir. (°)	3.0 (309°)			3.3 (10.3°)			5.0 (16.6°)		
Canal Emission									
Mean (± St.Dev.) Residence Time (d)	2.7 ± 4.2	1.8 ± 2.5	5.6 ± 4.6	5.0 ± 4.2	2.8 ± 2.2	7.6 ± 4.9	5.7 ± 4.6	5.6 ± 4.4	3.8 ± 5.1
Flushing Time (d)	10.2	5.0	18.8	13.2	7.5	21.7	19.9	16.9	18.2
Bay Emission									
Mean Residence Time (d)	32.5	11.7	56.3	32.2	22.6	36.5	23.1	12.2	23.7
Flushing Time (d)	98.5	34.8	171.8	97.3	68.8	113.4	70.2	35.3	69.9

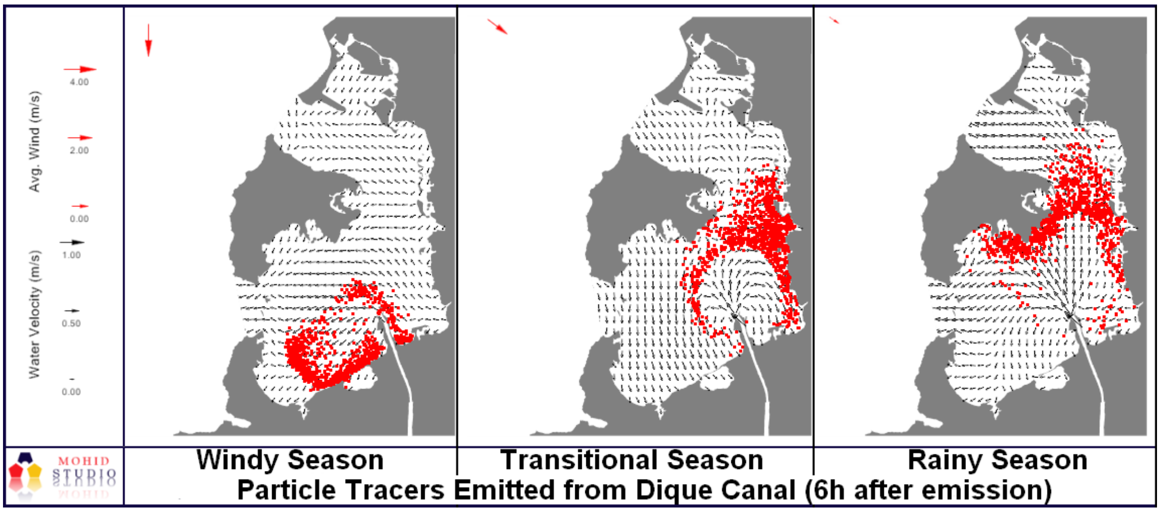


Figure 4.10. Evolution of particle tracers six hours after emission from the Dique Canal in the windy (left), transition (center) and rainy (right) season simulations. Wind conditions (mean speed and direction) during the six-hour period and instantaneous surface water velocities are also presented.

Under scenarios of increased or decreased discharge conditions, water renewal time scales were greatly affected in most cases. Doubling the Dique Canal's discharge substantially reduced the residence and flushing times of particles emitted throughout the bay in all three seasons. A similar effect was observed on the water renewal time scales of canal particles during the rainy and transitional season. While a reduction in water renewal time scales under double-discharge conditions also occurred during the windy season, the effect was less significant.

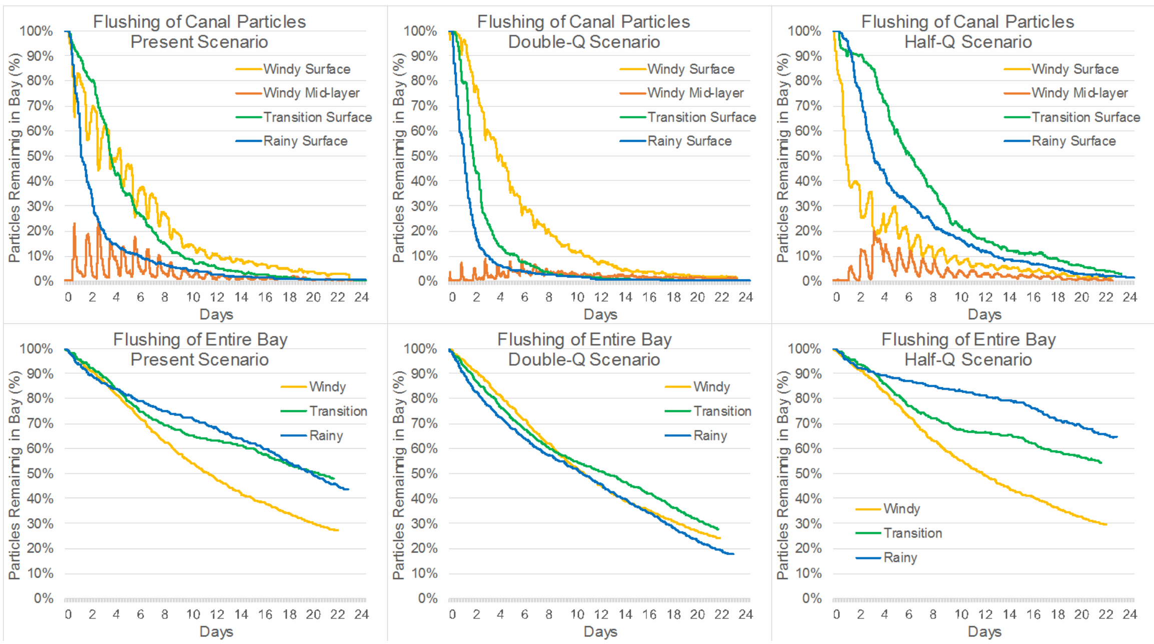


Figure 4.11. Evolution of particle tracers emitted from the Dique Canal (above) and throughout the entire bay (below) in the present (left), double-discharge (center) and half-discharge (right) scenarios for three seasons. Windy season particles emitted from the Dique Canal (above: yellow and orange lines) are divided between surface and sub-surface (<5 m) depths to illustrate vertical exchange through tidal oscillation.

By halving the canal discharge, water renewal time scales nearly doubled under rainy season conditions. This scenario also increased time scales during the transitional season, though not as much as during the rainy season. However, halving the canal discharge did not increase time scales during the windy season, but rather, had the effect of slightly decreasing the time scales of canal particles and had relatively no effect on the particles emitted throughout the bay.

Overall, by doubling the freshwater discharge, the bay's flushing times were reduced and there was less variation between the seasons (Fig. 4.11). Conversely, halving the freshwater discharge resulted in increased flushing times, with exception to the windy season, and greater variation between the seasons. A clear relationship was observed between discharge and flushing time of canal water yielding an exponential trend line with an R^2 value of 0.939 (Fig. 4.12). Discharge-dependent relationships were less pronounced for canal water residence time as well as bay water residence and flushing time due to the exceptions occurring under windy, low-discharge conditions. However, by grouping results of only the rainy and transitional seasons, clear exponential relationships can be observed for residence and flushing times of canal and bay water with R^2 values of at least 0.859 (Fig. 4.12). These clear relationships, and the lack of discernable relationships when grouping all three seasons together, reflect the different mechanisms of seaward exchange that occur during the windy season.

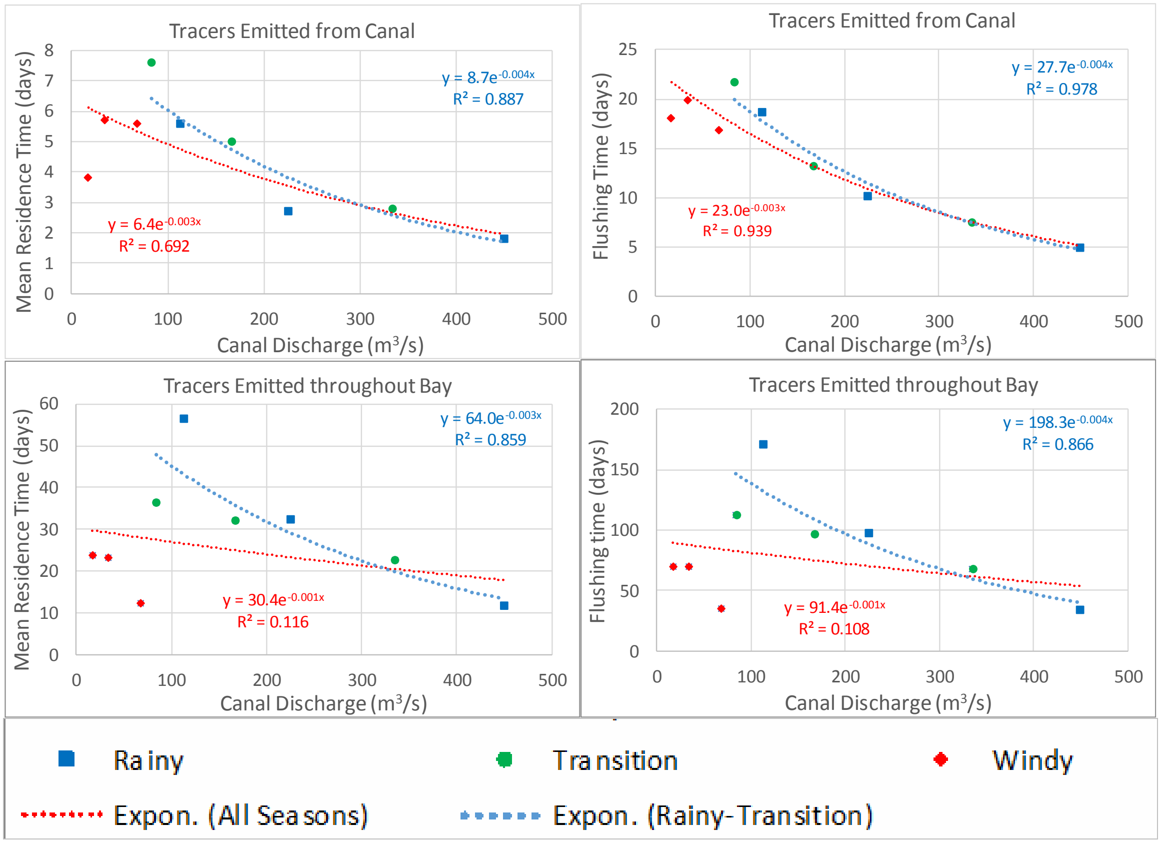


Figure 4.12. Relationships between canal discharge and the mean residence times (left) and flushing times (right) of particle tracers emitted from the canal (above) and throughout the entire bay (below). Blue trend lines correspond to a grouping of the Rainy and Transitional season, while red trend lines correspond to a grouping of all three seasons.

4.5 Discussion

4.5.1 Model Improvement

In consideration of the highly dynamic and stratified thermohaline structure of Cartagena Bay, it may be stated that the model performed quite well. The model's performance could be further improved though, specifically in its prediction of surface salinity and vertical temperature gradients. Inconsistencies in the model's reproduction of surface salinities were expected as there were additional sources of freshwater not considered in the model, including a series of smaller canals flowing from the industrial sector along the bay's east coast, surface runoff from the small catchment area of the bay itself, and occasional discharges from the city's sewerage system through an outdated submarine outfall and backup outlets along the coast when the system overflows. While the influence of the submarine outfall was not detected in the sub-surface salinity values of this study's monitoring results, the model's bias to overestimate surface salinity demonstrates the influence of additional freshwater sources which should be incorporated.

The model's ability to reproduce vertical temperature profiles would benefit from further investigation, particularly in consideration of the transitional season simulation. While monitoring results showed that this season had the strongest vertical temperature gradient (Figs. 4.2, 4.3a), modelling results produced a less pronounced thermal stratification (Fig. 4.6). Given the importance of water transparency on the penetration of solar radiation through surface waters, the model could be improved by incorporating the transport of suspended sediments which play a prominent role in the bay's water transparency. The effect of these sediments on water transparency was considered in the present study by calculation of the short-wave light extinction coefficient (K_d) with field measurements of total suspended solids (TSS), however, K_d values were computed using monthly mean TSS surface concentrations and held spatio-temporally constant during the simulation period. By incorporating TSS as a water property transported in the model, K_d values could be calculated independently for each cell and at each time-step, thus incorporating the spatio-temporal variability of water transparency into the model's computations of heat fluxes.

Model calibration could also benefit from additional investigation. While further exploration of parameters found to be less sensitive (surface water roughness, horizontal advection methods) may be inconsequential, continued evaluation of the more sensitive parameters, horizontal viscosity and bottom roughness, may be considered a relevant topic for future research. Coefficients for particle turbulence in the model's Lagrangian transport could also be optimized through calibration with field measurements, as the current study utilized default values known to have worked well in similar coastal systems. Lastly, model calibration based on measured velocity fields in the bay could also be attempted and compared to this study's results.

4.5.2 Hydrodynamic Phenomena

The small differences observed between modelled and measured water levels were prevalent at peaks of high and low water, while the frequency of the tidal signal was quite well matched. These differences may be associated with local and offshore meteorological anomalies, such as storms, waves and atmospheric pressure systems, as well as additional sources of freshwater discharge.

Nevertheless, values of MAE (0.031-0.039 m) and RMSE (0.039 – 0.048 m) were quite low, suggesting that these differences were of minimal consequence. The result showing higher mean water level in the rainy season is in agreement with the previous work of Torres & Tsimplis (2012) which showed a seasonal cycle of water level along the north coast of South America with a peak in October.

The importance of utilizing a 3D model with a high vertical resolution in order to understand the processes of a sharply stratified system such as Cartagena Bay was demonstrated in this study. Some previous studies which utilized 2D models (Palacio et al., 2010) or vertically integrated their results (Molares & Mestres, 2012a) presented a simplified view of the bay's hydrodynamics based on net fluxes, suggesting that water simply flows into the bay through Bocachica, runs south to north, and flows out through Bocagrande. However, results of the present study show more clearly how dual currents are found at both straits (surface outflow, sub-surface inflow) and that the direction of currents along the bay's north-south axis varies both by depth and by seasonal conditions.

The process of vertical exchange is one of the defining characteristics of the bay's seasonal variability. The rainy and transitional seasons exhibited strong thermohaline stratification, predominantly horizontal water velocities and increased stability in density fields near the surface. These conditions restrict vertical advection and vertical turbulent mixing in the bay and result in shorter time scales for surface water renewal. Conversely, the strong winds and lower freshwater discharge of the windy season result in less thermohaline stratification, a shallowing of the pycnocline and greater instability in the water column, which in turn permit the penetration of turbulent diffusion below the surface. This process likely occurs as a positive feedback loop, as the reduction in stratification permits more vertical mixing and increased vertical mixing further reduces stratification. The wind-driven accumulation of water in the southwest part of the bay (Fig. 4.7) probably also contributes to this process as it causes a gradient in water level which would generate downwelling that transports surface waters to depth. Under these conditions, velocity fields throughout the bay experience increased vertical velocities and highly fluctuating circulation patterns, which result in shorter time scales for water renewal of the bay's overall volume.

The exchange between surface and sub-surface waters was clearly illustrated by the evolution of particle tracers emitted from the canal, which showed substantial vertical exchange during the windy season scenarios though almost none during the other seasons (Fig. 4.11). This vertical transport coincided with tidal oscillation and was most prevalent under the present and half-discharge scenarios of the windy season. While tidal forces were also present during the rainy and transitional seasons, it appears that the strong vertical stratification during these seasons, along with the increased stratification of the windy double-discharge scenario, caused resistance to vertical mixing. As vertical exchange during the present and half-discharge scenarios of the windy season were quite similar, this suggests that there is a threshold value of freshwater discharge between the present (33.9 m^3/s) and double-discharge scenario (67.7 m^3/s) at which point vertical stratification becomes strong enough to result in resistance to vertical advection.

4.5.3 Factors Controlling Water Renewal

Modelling results show that freshwater discharge is a principal factor controlling water renewal in Cartagena Bay. This effect is particularly well-defined in the water renewal of surface waters, as tracer particles emitted from the canal were flushed out of the bay faster with increased discharge levels (Fig. 4.12). However, as observed during the windy season scenarios, under low-discharge conditions the freshwater discharge level is no longer the controlling factor, as has been found in other coastal systems as well (Janeiro et al., 2008). A potential explanation for the reduced effect of discharge on surface water renewal times during windy season scenarios could be that because the strong northerly winds confine the tracer particles within the southern lobe of the bay (Fig. 4.10), their transport towards the seaward straits becomes more random and less dependent on discharge-driven surface velocities.

Water renewal of the bay's complete volume differs from that of surface waters due to the effect of vertical mixing (Fig. 4.10). Under the medium- and high-discharge levels of the transitional and rainy season scenarios, respectively, increased discharge resulted in reduced time scales. However, some of the shortest time scales for water renewal of the complete bay were found with the windy season scenarios, despite their low-discharge levels. Therefore, while surface water flushing is almost completely dependent on discharge levels, the flushing of the bay's total volume is dependent on both discharge levels and wind-driven vertical circulation, the latter of which becomes more important at low-discharge levels.

The effect of the winds as a factor controlling water renewal was also evident in modelling results. For instance, the water renewal times for the windy season double-discharge scenario were shorter than those of the transitional season half-discharge scenario, despite the fact that the former had a lower mean discharge of 67.7 m^3/s than the latter's 83.7 m^3/s (Table 4.2). This difference, contrary to the general discharge-renewal time relationship (Fig. 4.12), may be attributed to the influence of the winds on increasing surface water velocities and vertical exchange. Similarly, under the transitional season half-discharge scenario, the mean residence time (36.6 days) and flushing time (113.4 days) of the overall bay are shorter than those of the rainy season half-discharge scenario (56.3 and 171.8 days, respectively), again despite the fact that the former has a lower mean discharge (83.7 m^3/s) than the latter (112.5 m^3/s). While the greater discharge of the rainy season half-discharge scenario did result in shorter water renewal times for the canal water, the transitional season half-discharge scenario generated shorter water renewal times for the overall bay, presumably due to the effect of stronger winds on vertical exchange. Therefore, while canal discharge appears to be the dominant factor controlling water renewal in the bay, under similar discharge conditions the factor of winds becomes prevalent.

Tidal exchange is also known to be a common factor controlling residence times (Rynne et al., 2016). Although the tides are small in Cartagena Bay, instantaneous velocity fields showed increased outflow during ebb tides and the tidal effect on vertical mixing was prominent under low-discharge conditions (Fig. 4.11). However, as the model simulations were run for complete lunar cycles, tidal effects on residence times would only result in temporal variability within the simulation period, which should balance out in the final result.

4.5.4 Implications of Development Scenarios

4.5.4.1 Increased Freshwater Discharge

Modelling results show that an increase in upstream watershed runoff, which has been projected for future years (Restrepo et al., 2018), would reduce water renewal time scales in the bay, for both the case of canal waters and the bay's overall volume. This result may be expected in the case of canal water as increased discharge would result in faster surface water velocities that flush the freshwater out of the bay at a higher rate. Reduced time scales for the renewal of the bay's entire water volume, however, may not be as intuitively foreseen because increased freshwater discharge would also cause stronger thermohaline stratification which restricts vertical mixing. Yet model results show that freshwater discharge is a primary factor controlling water renewal in the bay under scenarios of increased runoff. Given that the bay's seaward outflow occurs almost exclusively at surface depths, a potential explanation of this result could be that despite decreases in vertical advection and turbulent mixing due to stronger stratification, when particles finally do rise to the surface layers their seaward flushing by faster surface velocities is achieved in a time scale that outweighs the time lost due to slower vertical circulation.

Though the double-discharge scenario results show that the bay's water renewal would improve as runoff continues its increasing tendency over time, it is essential to recall that the Dique Canal is also one of the bay's principal sources of pollution. Therefore, the implication of this scenario on the bay's water quality is more complex as increased discharge is also likely to bring greater amounts of pollution to the bay. Furthermore, the behavior of pollutants in the bay would differ from the behavior of passive particles due to additional processes such as gravitational percolation, chemical reactions and biological cycling. The question then becomes in regard to the balance that would eventually be reached between increases in both pollution and water renewal rates. Further research using water quality models is recommended to evaluate this question.

4.5.4.2 Reduced Freshwater Discharge

Presented with the scenario of the upstream hydraulic intervention planned to reduce freshwater flows into the bay by constructing hydraulic doors along the Dique Canal (Fondo Adaptación, 2018), modelling results show that this would affect the bay by increasing water renewal time scales. This result of longer water renewal time scales due to decreased freshwater discharge may be explained by discharge level being a principal controlling factor on the bay's hydrodynamic processes. This relationship was clear for rainy and transitional season conditions, though discharge level becomes less of a controlling factor under low-discharge conditions thus having minimal effect on the windy season's water renewal times.

These results suggest that the plan to improve Cartagena Bay's pollution issues by constructing hydraulic doors upstream in the canal may be flawed. Such a plan would surely reduce the sediment and pollution loads flowing into the bay from the watershed. However, the increased water renewal time scales that would result from decreased discharge-driven water velocities in the bay could in fact worsen the bay's water quality. This risk has been suggested by other authors (Gomez et al., 2009; Grisales et al., 2014) and previously demonstrated using water quality models (Lonin & Tuchkovenko, 1998; Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003). These modelling studies evaluated various mitigation options and found that the best solution would be to reduce upstream discharge but only in combination with the elimination of local sources of industrial and domestic wastewater, without which the reduction of upstream discharge would result in increased eutrophication and a worsening of oxygen conditions in the bay. The present study corroborates these findings as increased water renewal time scales under reduced discharge scenarios would inhibit the bay's mixing and oxygenation, while the continuous input of local sources of domestic and industrial wastewater (Tosic et al., 2018a) could maintain or worsen the bay's multifactorial pollution issues.

4.5.5 Management Recommendations

This study provides a valuable step in the development of coastal management tools for Cartagena Bay. The application of the hydrodynamic model permitted the identification of risks associated with the upstream anthropogenic impacts on freshwater runoff and how these changes can affect the bay's hydrodynamic processes. The assessment of such modifications to coastal circulation is particularly relevant as semi-enclosed coastal water bodies are quite vulnerable to changes in water renewal rates (Dettmann, 2001; Anthony et al., 2009; Newton et al., 2014). For the continued development of modelling tools for the management of Cartagena Bay, it would be recommended to carry out further studies in water quality modelling and operational modelling (e.g. Kenov et al., 2014) which could provide environmental authorities and stakeholders with a real-time early warning system.

The most urgent solution needed for the mitigation of upstream increases in runoff, sediment and pollution loads is improved management of the Magdalena watershed. This is both a daunting and challenging task due to the basin's large scale, widespread population and diverse anthropogenic activities, and so integrated management of this watershed would require collaboration between a great number of municipal and regional governments and environmental authorities. Currently, there is no such initiative for the joint management of this watershed that is being effectively implemented. Nevertheless, the increasing trends in fluvial fluxes from the watershed (Restrepo et al., 2018) associated with land-cover changes and deforestation (Restrepo et al., 2015) along with the potential for further runoff increases due to climate change and intensification of the El Niño-Southern Oscillation (IPCC, 2014; Paeth et al., 2008; Restrepo et al., 2018) make the need for integrated watershed management urgent.

Considering that the plan to reduce freshwater discharges by constructing upstream hydraulic doors could in fact worsen water quality issues in Cartagena Bay, it is crucial to focus efforts on the reduction of local pollution sources. While many of these local wastewater sources have begun to discharge their wastewater to the city's sewage system which flows out to sea north of the city, a large many sources continue to discharge into the bay, as does the city sewage system itself when it overflows. The number of wastewater sources may in fact be increasing as residential populations and the industrial sector are likely growing faster than the city's sewage system. Meanwhile, the wastewater sources that continue to discharge into the bay do so with little to no control on their effluents. As such, there is an urgent and long-standing need for the development of effluent control policies in Cartagena Bay (Tosic et al., 2017, 2018a).

Unfortunately, there appears to be a lack of balance in the progress of economic activities and management plans in Cartagena. On the one hand, new industries and ports have rapidly been developed in Cartagena Bay in recent years. On the other hand, the city has yet to successfully complete a development plan over the past seven years, in which time the city has had 10 different mayors, many of whom have faced scandals of alleged corruption. In this light, perhaps the unbalanced development of Cartagena Bay should not be surprising, though by no means should it be condoned.

4.6 Conclusions

The approach presented in this study was successfully applied to the assessment of water renewal times and future mitigation scenarios, which represent important knowledge needed by environmental managers and decision makers. The pollution issues studied in Cartagena Bay are commonly found in other highly populated coastal areas of the Wider Caribbean Region. This approach could also be applied to other tropical bays in the region facing similar environmental challenges and thus contribute towards resolving common issues in the Caribbean.

Cartagena bay is characterized by strong thermohaline stratification created by fluxes of freshwater runoff and surface heating which inhibit vertical circulation for most of the year. Variability in these fluxes and winds create distinct seasonal conditions. Rainy season conditions consist of high levels of freshwater runoff from the Dique Canal which generate strong vertical density gradients, a deepening of the pycnocline, increased stability in the water column's density structure and heightened water levels in the canal outlet area. These conditions result in increased horizontal velocities which predominantly flow northwards and westwards to the bay's seaward straits where the outflow of less saline surface water is balanced by the inflow of sub-surface seawater. Most of this seawater flows into the bay occur through the southern strait, Bocachica, due to the depth of its navigation canal while the inflow through the northern strait, Bocagrande, is limited by its sub-surface seawall. These conditions are mostly maintained in the transitional season though an increase in northerly winds and decreased freshwater discharge result in an increased westward component in surface flow towards the bay's southern strait.

Further strengthening of northerly winds and reductions in freshwater discharge during the windy season have a much greater effect on the bay's conditions. This season is characterized by much weaker vertical density gradients, a shallowing of the pycnocline and an unstable density structure in the water column. These conditions permit greater vertical exchange generated by wind-driven turbulence and tidal oscillation. Circulation within the bay changes drastically in this season with increased vertical velocities and highly fluctuating circulation patterns. On the surface, the intensification of winds and reduced discharge result in predominantly southwestward currents and heightened water levels in the bay's southwest lobe. This concentrates seaward flow in the southern strait which causes various features (surface outflow, sub-surface inflow and an associated sub-surface anticyclonic eddy) to all occur at lower depths.

Presently, water flowing from the Dique Canal has a mean residence time of 3-6 days and a flushing time of 10-20 days. The variability of these time scales is strongly dependent on canal discharge levels. Meanwhile, the bay's entire water volume has a mean residence time of 23-33 days and a flushing time of 70-99 days. These time scales for the water renewal of the bay's entire volume are principally dependent on discharge level, while wind speed becomes a more prevalent factor at low-discharge levels.

The assessment of future scenarios showed that increases in freshwater runoff caused by human development in the upstream watershed would result in faster water renewal in the bay, while plans to decrease freshwater discharge by constructing hydraulic doors would result in slower water renewal in the bay. Therefore, ongoing plans to mitigate pollution issues in Cartagena Bay by reducing freshwater discharges could in fact worsen the bay's water quality issues as local wastewater discharged into the bay would remain in the system for longer time periods. In this consideration, there is an urgent need for the reduction of local pollution sources and effluent control policies in Cartagena Bay.

While current trends of increasing fluvial fluxes from the upstream watershed would result in faster water renewal times, these fluxes would also be accompanied by greater pollution loads flowing from the canal into the bay. But would these increased water renewal rates be capable of assimilating the increased pollution? One could postulate with regards to the balance that would eventually be reached between these fluvial alterations, and such hypotheses could be evaluated with studies of water quality modelling. In the meantime, there is a long-standing need for improved management of the Magdalena watershed, which is essential to the water resources of Colombia regardless of future scenarios in Cartagena Bay.

4.7 Connecting Text

This chapter showed that the Dique Canal is a principal factor controlling Cartagena Bay's water renewal process, and so current mitigation plans to reduce the canal's flows could in fact worsen the bay's water quality issues because under such conditions the local sources of wastewater would remain in the system for longer time periods. Therefore, as previously suggested in chapter III, the sole mitigation of the canal would not effectively solve the bay's pollution issues, and the currently proposed plan could even worsen them if not combined with local pollution control. The question then remains: how do we resolve the pollution issues in Cartagena?

Clearly there is an urgent need to control land-based discharges both in the upstream watershed and in nearby sources of wastewater. To control any of these sources, we need management objectives and policy that regulates discharges in accordance with these objectives. Such objectives could be in the form of effluent limitations or end-of-river targets. However, for these objectives to be effective, they need to consider the coastal zone's nearshore hydrodynamics in order to verify that the given discharge objective will result in adequate water quality in the receiving waters. As described in chapters I and II, for water quality to be considered adequate it must comply with criteria (e.g. international standards) established for the water's use at a given location (e.gs. a marine protected area, a touristic beach). Fortunately, the hydrodynamic model calibrated in this chapter presents management with a valuable tool that can be coupled with a water quality model in order to resolve the question of pollution dispersion processes in the coastal zone. In the following chapter, a novel method is presented that applies a coupled hydrodynamic-water quality model in order to set targets for coastal water quality objectives.

CHAPTER V

A practical method for setting coastal water quality targets: Harmonization of land-based discharge limits with marine ecosystem thresholds

"Nature may reach the same result in many ways. Like a wave in the physical world, in the infinite ocean of the medium which pervades all, so in the world of organisms, in life, an impulse started proceeds onward, at times, may be, with the speed of light, at times, again, so slowly that for ages and ages it seems to stay, passing through processes of a complexity inconceivable to men, but in all its forms, in all its stages, its energy ever and ever integrally present. A single ray of light from a distant star falling upon the eye of a tyrant in bygone times may have altered the course of his life, may have changed the destiny of nations, may have transformed the surface of the globe, so intricate, so inconceivably complex are the processes in Nature. In no way can we get such an overwhelming idea of the grandeur of Nature than when we consider, that in accordance with the law of the conservation of energy, throughout the Infinite, the forces are in a perfect balance, and hence the energy of a single thought may determine the motion of a universe."

- Никола Тесла, 1893

5.1 Abstract

The Caribbean Sea provides significant ecosystem services to the livelihood and well-being of countries in the region. Protection of the marine ecosystem requires policy on coastal water quality that considers ecologically-relevant thresholds and has a scientific foundation linking land-based discharges with marine water quality. This study demonstrates a practical method for setting local-scale coastal water quality targets by applying this approach to the example of Cartagena Bay, Colombia, and setting targets for end-of-river suspended sediment loads in order to mitigate offshore coral reef turbidity impacts. The presented approach considers reef thresholds for suspended sediment concentration and uses monitoring data to calibrate and apply a coupled 3D hydrodynamic-water quality model (MOHID) to link the marine thresholds to fluvial loads. Monitoring data showed that suspended sediment concentrations in the study area were consistently above the coral reef ecosystem threshold of 10 mg/l, and the calibrated model adequately reproduced field observations. It was shown that ecosystem total suspended solids (TSS) thresholds could be maintained within the extent of the bay by reducing suspended sediment loads in the Dique Canal from current load estimates of 6.4 $x10^3$ t/d (rainy season) and 4.3 $x10^3$ t/d (transitional season) to target loads of 500-700 t/d, representing load reductions of approximately 80-90%. These substantial reductions reflect ongoing issues in the Magdalena watershed which has experienced severe erosional conditions and intense deforestation over the past four decades. The presented method is practical for countries developing water quality policy without access to long-term datasets, and could also be applied to other parameters or discharge types. The method is particularly beneficial for the development of site-specific targets, which are needed in consideration of the natural and anthropogenic variability between different coastal zones and water bodies.

5.2 Introduction

The Convention for the Protection and Development of the Marine Environment in the Wider Caribbean Region (WCR), also known as the "*Cartagena Convention*", is a regional legal agreement for the protection of the Caribbean Sea, adopted in 1983. Its implementation is of great importance to the region, as the ecosystem services provided by the Caribbean Sea are quite relevant to the economy and public health of the WCR's member states, many of which are small island developing states that are dependent on fishing and tourism (WTTC, 2017). Marine ecosystems in the region have suffered impacts such as the widespread degradation of coral reefs, which have declined from an average of 34.8% in live coral cover to just 14.3% between 1970 and 2012 (Jackson et al., 2014). This decline can be partly attributed to water quality impacts from land-based pollution, as approximately 85% of wastewater is discharged to the Caribbean coast without treatment (UNEP, 2004). In recognition of this issue, the Cartagena Convention includes a protocol concerning pollution from land-based sources and activities (LBS Protocol) which obliges countries to take measures to prevent, reduce and control coastal pollution, including the establishment of legally binding standards for sewage effluent and discharges (UNEP, 1999). While the country of Colombia is both a member state and depository of the Cartagena Convention, it has yet to ratify the LBS Protocol. Ironically, 35 years after the convention's adoption in the city of

Cartagena, Colombia, the coastal zone of Cartagena has become one of the region's hot-spots of pollution (Tosic et al., 2017, 2018a).

To effectively manage marine pollution issues at the local, national or regional level, there is a need for water quality standards that both: i) have a scientific foundation that links land-based discharge limits with marine water quality standards; and ii) are relevant to the ecological thresholds of the receiving marine ecosystems (Brodie et al., 2009, 2017; Kroon, 2012; Wooldridge et al. 2006, 2015). In terms of land-based pollution sources, standards are needed to control point-source effluents of domestic and industrial wastewater, which are contemplated in the LBS Protocol, and should consider pollutant loads rather than just pollutant concentration. There is also a need to develop targets for river outlets, such as those outlined by the European Union's Water Framework Directive (WFD, 2000/60/EC) which requires river basin management plans to link coastal and river objectives, or Australia's Reef Plan (Queensland Government, 2018) which includes targets for end-of-river reductions in nutrients and sediments.

The method by which land-based discharge targets are established will determine whether or not they are relevant to the receiving ecosystem. For example, the USA's Clean Water Act entails the establishment of Total Maximum Daily Loads (TMDL) which should be designed to meet a water quality standard in an impaired receiving water body (Karr & Yoder, 2004). Yet effluent limits in the USA are determined by methods, such as the Best Available Technology Economically Achievable (EPA, 2018), which are actually focused on the economic feasibility of implementing wastewater treatment rather than on the assimilative capacity of the receiving environment. Other methods of target setting for land-based discharges have been based on pre-industrial loads (Brodie et al., 2001) or the feasibility of improved agricultural practices (Brodie et al., 2012). However, if land-based discharge targets do not consider the coastal zone's hydrodynamic characteristics and marine ecological thresholds, they may not ensure the compliance of marine water quality standards (Schernewski et al., 2008) and thus cannot be effective in protecting marine ecosystems (Kroon, 2012).

The coral reefs offshore of Cartagena, Colombia, adequately demonstrate the ecosystem impacts resulting from inadequate policy on water quality standards. These reefs are in the Marine Protected Area of the Rosario Islands (Fig. 5.1) and have an average coral cover of approximately 23% (Restrepo & Alvarado, 2011). But they incur a chronic stress caused by river sediment plumes as data show that over the 2000-2013-period, the Rosario Islands were exposed to turbid waters between 19.6 and 47.8% of the time (Restrepo JD et al., 2016). Excessive suspended sediments concentrations in reef waters can impair coral health due to the restriction of light, the smothering of coral polyps and the associated nutrients transported with the sediments (Fabricius, 2005). However, current policy in Colombia does not include targets for river outlets nor does it include marine ambient water quality standards for suspended sediments or nutrients (MinSalud, 1984). Policy has recently been developed for point-source discharges in Colombia's coastal zone (MADS, 2018) though without established marine water quality thresholds or consideration of nearshore dispersion processes, the policy lacks the science-based foundation needed to ensure marine ecosystem relevance. Given the socioeconomic importance of Cartagena, as the country's #1 touristic destination, and the ecological relevance of the Rosario Islands, Colombia has a need for water quality standards that adequately protect its natural marine resources.

Various science-based methods have been developed to link water quality objectives for land-based discharges and marine waters. At the large scale of the Baltic Sea, Schernewski et al. (2015) coupled a river basin flux model with a marine ecosystem model to set target concentrations of nitrogen, phosphorus and chlorophyll-*a* in river outlets and marine waters. This approach established a marine water reference as 150% of modeled pre-industrial conditions and then targeted the reductions in river nitrogen load needed to comply with this marine reference. Research in the Great Barrier Reef has linked end-of-river and marine water targets based on ecosystem thresholds determined by long-term ecological assessments (Brodie et al., 2017). This approach established marine thresholds of photic depth for seagrass and chlorophyll-a for coral reefs, and then used linear relationships to determine river load targets for sediments and nitrogen. While these approaches may be appropriate for large areas like the Baltic Sea or Great Barrier Reef, they may not be applicable at smaller scales where local hydrodynamics and dispersion processes could deviate from the generalized relationships of these seasonally averaged, spatially integrated approaches. Furthermore, the aforementioned studies utilized decades-worth of monthly monitoring data, which a developing country such as those of the WCR is unlikely to have.

Some local-scale studies have quantified the link between land-based loads and receiving waters by applying hydrodynamic and water quality models. Deng et al. (2010) and Han et al. (2011) used similar methods of modelling and linear programming in Jiazhou Bay and the Yangtze River Estuary, China, respectively, to calculate the maximum river nutrient load that would maintain marine chlorophyll-*a* below the given criteria. However, this approach did not consider ecosystem-relevant seawater quality targets. Ramin et al., (2011) successfully calibrated a water quality model of Hamilton Harbour in Lake Ontario, Canada, to relate land-based phosphorus loads to the harbour's water quality criteria for phosphorus and chlorophyll-*a*. However, this method was also based on seasonal averages which disregard the temporal variability that in reality could result in the frequent exceedance of the water quality criteria.

In this study, we aim to demonstrate a practical method for setting local-scale coastal water quality targets. Such methods are needed for local-scale environmental management in countries like those of the WCR that are still developing their water quality policies. We apply this method to the example of Cartagena Bay, Colombia, by setting targets for end-of-river suspended sediment loads in order to mitigate offshore reef turbidity impacts. The study's objectives are to present a method that is: i) appropriate at the local-scale, ii) science-based, iii) ecosystem-relevant, and iv) applicable to the coastal zones of developing countries where historical datasets are unlikely to be available. By considering marine ecological thresholds and applying a coupled 3D hydrodynamic-water quality model (MOHID) to link the marine thresholds to fluvial loads, the presented approach is both science-based and ecosystem-relevant. The model's fine resolution accurately captures local hydrodynamic and dispersion processes, making it appropriate at the local scale, while the two years of monthly data used in this study could feasibly be collected by a developing country devoid of historical datasets. We hypothesize that marine suspended sediment concentrations in the outer limits of Cartagena Bay can be maintained below coral reef ecosystem thresholds by reducing fluvial suspended sediment loads in the bay's principle freshwater source, the Dique Canal. Our specific research question thus asks: what fluvial suspended sediment load is needed to effectively ensure that the selected marine ecosystem target is not exceeded?

5.3 Materials and Methods

5.3.1 Study Area

The tropical semi-closed estuary of Cartagena Bay is situated in the southern Caribbean Sea on the north coast of Colombia (10°20' N, 75°32' W, Fig. 5.1). The bay has an average depth of 16 m, a maximum depth of 32 m and a surface area of 84 km², including a small internal embayment situated to the north. Water exchange with the Caribbean Sea is governed by wind-driven circulation and tidal movement through its two seaward straits (Molares & Mestres, 2012a): "Bocachica" to the south and "Bocagrande" to the north. Movement through Bocagrande strait is limited by a defensive colonial seawall 2 m below the surface. Bocachica strait consists of a shallow section with depths of 1-3 m, including the Varadero channel, and the Bocachica navigation channel which is 100 m wide and 24 m deep (Tuchkovenko & Lonin, 2003; Lonin et al., 2004). The tides in the bay have a mixed, mainly diurnal signal with a micro-tidal range of 20-50 cm (Molares, 2004).

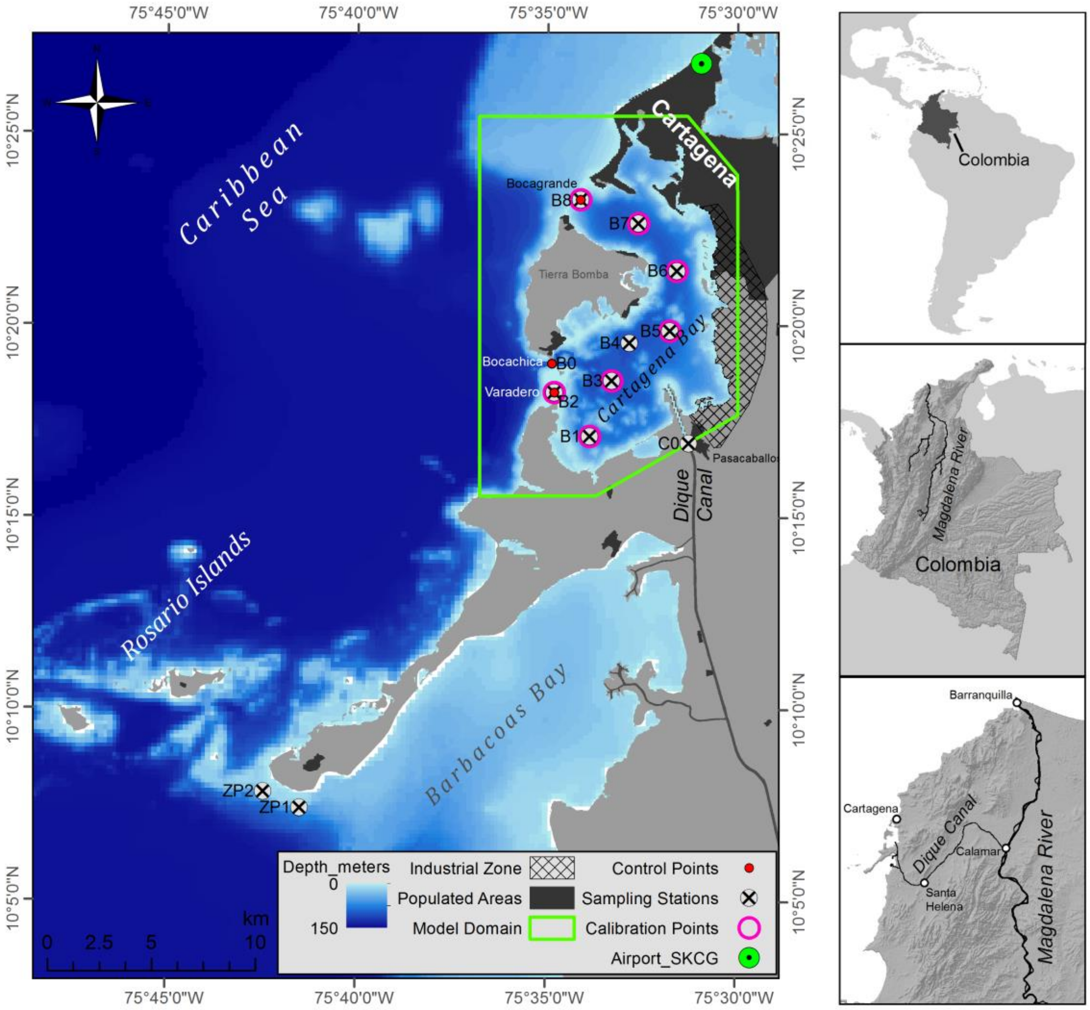


Figure 5.1. Principal panel: study area showing sampling stations, model calibration points, control points, weather station (SKCG), bathymetry and model domain. Secondary panels: location of Colombia (upper panel); location of the Magdalena River (middle panel); flow of the Magdalena into the Caribbean Sea and along the Dique Canal into Cartagena Bay (lower panel).

Estuarine conditions in the bay are generated by discharge from the Dique Canal which diverges from the Magdalena River at Calamar, 114 km upstream of Cartagena Bay. The Dique Canal discharges approximately 50-250 m^3/s of freshwater into the bay (Tuchkovenko & Lonin, 2003; Tosic et al., 2018a), the variability of which is strongly related to seasonal runoff from the Magdalena watershed. With an area of 260,000 km^2, this basin covers approximately 25% of the country's land area and is the main contributor of fluvial fluxes to the Caribbean Sea (Restrepo & Kjerfve, 2000; Restrepo, 2008). The Dique Canal transports a total sediment flux of 1.9 Mt/y (Restrepo et al., 2018) which has increased over the past decade (Restrepo et al., 2015). When compared with the average discharge during the 2000-2010 period, the canal's sediment load is projected to increase by as much as 317% by the year 2020 (Restrepo et al., 2018). The sediments discharged into Cartagena Bay have been characterized as fine lithoclastic clays with low $CaCO_3$ content (Franco et al., 2013) and deposition rates that vary between 1.2 – 5.0 cm/y (Restrepo et al., 2013). The flow of freshwater and sediments into the bay generate a highly stratified upper water column with a pronounced pycnocline in the upper 4 m of depth, above which turbid freshwater is restricted from vertical mixing and fine suspended particles tend to remain in the surface layer (Tuchkovenko et al., 2000, 2002; Tuchkovenko & Lonin, 2003).

The Dique Canal's highest discharges typically occur from October to November, while its lowest levels occur from January to February. Winds are strongest and predominantly northerly from January to April due to the trade winds which coincide with the strengthening of the southern Caribbean upwelling system and the cooling of surface water temperatures (Andrade & Barton, 2005; Rueda-Roa & Muller-Karger, 2013). These conditions result in seasonal variability of the bay's water quality (Tosic et al., 2017) and hydrodynamics (Tosic et al., 2018b). During the dry/windy season (Jan.-Apr.), vertical mixing increases and water quality improves. This is reversed during the transitional (May-Aug.) and rainy (Sept.-Dec.) seasons when vertical stratification increases and water quality worsens. The transitional season has the highest temperatures, lowest levels of dissolved oxygen and a dominant westward surface water flow towards Bocachica strait. In contrast, the rainy season has a larger northward surface water flow towards Bocagrande and higher concentrations of suspended sediments, nutrients and pathogenic bacteria. Restrepo et al. (2016) also showed that the dry season's strong northerly winds result in greater sediment deposition in the southern part of the bay, while during the rainy season there is a more homogenous distribution of sediments, the extent of which is proportional to discharge from the Dique Canal.

Marine ecosystems in this coastal zone have been degraded by poor water quality and other impacts, including the depletion of seagrass beds, coral reefs and benthic suspension feeding invertebrates (Restrepo et al., 2006, 2016). On the seaward side of Bocachica strait lies the Varadero reef, which has a high coral cover of 45.1% despite its proximity to Cartagena Bay (Pizarro et al., 2017), likely due to the strong stratification of water flowing out of the bay which maintains sediments and associated contaminants in the surface layer until they are transported further offshore. Further offshore are the previously described reefs of the Marine Protected Area of the Rosario Islands (Fig. 1), which are chronically exposed to river sediment plumes (Restrepo JD et al., 2016).

5.3.2 Data Collection

Water quality was monitored monthly in the field from Sept. 2014 to Nov. 2016 between the hours of 9:00-12:00 (Karydis & Kitsiou, 2013). Measurements were taken from 11 stations (Fig. 5.1), including one station in the Dique Canal (C0), eight stations in Cartagena Bay (B1-B8) and two stations at the seaward end of Barú peninsula (ZP1-ZP2). At all stations, CTD casts were deployed using a YSI Castaway measuring salinity and temperature every 30 cm of depth. Grab samples were taken from surface waters while bottom waters (22 m depth) were sampled with a Niskin bottle. Surface samples were collected in triplicate at station C0. A triplicate sample was also taken at a different single station in the bay each month to estimate sample uncertainty. Samples were analyzed at the nearby Cardique Laboratory for total suspended solids (TSS) by standard methods (APHA, 1985).

At station C0 in the Dique Canal, discharge was measured with a Sontek mini-ADP (1.5 MHz) along a cross-stream transect three times per sampling date from Sept. 2015 to Nov. 2016. Bathymetric data with 0.1 m vertical resolution were digitized from georeferenced nautical maps (#261, 263, 264) published by the Colombian Navy's Centre for Oceanographic and Hydrographic Research (CIOH-DIMAR). In the 3x2 km area of Bocachica strait, the digitized bathymetry was updated with high-precision (1 cm) bathymetric data collected in the field on 17 Nov. 2016.

Hourly METAR data of wind speed, wind direction, air temperature and relative humidity were obtained from station SKCG at Rafael Núñez International Airport (approximately 10 km north of the bay; Fig. 5.1). Tidal components were obtained for numerous locations offshore of the bay from the finite element solution tide model FES2004 (Lyard et al., 2006) using the MOHID Studio software. Hourly measurements of water level at a location within the bay were also obtained from the Colombian Navy's Centre for Oceanographic and Hydrographic Research (CIOH-DIMAR).

5.3.3 Model Application

The hydrodynamics of Cartagena Bay were simulated using the MOHID Water Modelling System (Leitão et al., 2008; Mateus & Neves, 2013). The MOHID Water model is a 3D free surface model with complete thermodynamics and is based on the finite volume approach, assuming hydrostatic balance and the Boussinesq approximation. It also implements a semi-implicit time-step integration scheme and permits combinations of Cartesian and sigma coordinates for its vertical discretization (Martins et al., 2001). Vertical turbulence is computed by coupling MOHID with the General Ocean Turbulence Model (GOTM; Burchard, 2002).

Model configuration for Cartagena Bay was based on an equally-spaced Cartesian horizontal grid with a resolution of 75 m and a domain area of 196 km^2 (Fig. 5.1). This included an offshore area extending 2.3 km off the bay, though only the results inside the bay are considered within the limits of the monitoring stations used for calibration. A mixed vertical discretization of 22 layers was chosen to reproduce the mixing processes of the highly stratified bay by incorporating a 7-layer sigma domain for the top five meters of depth and a variably-spaced (depth-incrementing) Cartesian domain below that depth. An optimal time step of 20 seconds was chosen.

The collected data of temperature, salinity, canal discharge, bathymetry, tides, winds and other meteorological data were used to configure the hydrodynamic model. This process included a sensitivity analysis to identify the system's most effective calibration parameters (horizontal viscosity and bottom roughness), and subsequent calibration and validation of the model. For more details on the hydrodynamic model's configuration, calibration and results, see Tosic et al., (2018b).

MOHID Water was also utilized to model the dynamics of suspended sediments in Cartagena Bay's surface waters, as the model has previously been shown to successfully reproduce sediment dynamics in estuaries (e.g., Franz et al., 2014). The modelling focused on the rainy and transitional seasons, which both yield high TSS concentrations in the bay though with distinct flow characteristics. For this purpose, two periods of a lunar-month duration in the year 2016 were simulated: 28 June – 26 July (transitional season) and 19 Oct. – 17 Nov. (rainy season). Start and end times for each simulation coincided with the dates of monthly field sessions. Initial conditions of suspended sediment concentration were defined by TSS measurements made on the corresponding start date. To avert numerical instabilities, a spin-up period of one day was applied to gradually impose wind stress and open boundary forcings (Franz et al., 2016). Outputs from the simulations were compared with field measurements of surface TSS on the respective end date at seven calibration points in the bay (Fig. 5.1).

The model's suspended sediment block included flocculation processes. Calibration focused on the parameterization of the settling velocity (W_s) of suspended sediments. Constant and variable settling velocity configurations were tested, with a better fit yielded by the formulation proposed by Nicholson & O'Connor (1986): $W_s = K \cdot C^m$, where K is a coefficient relating to sediment mineralogy, C is the TSS concentration and the exponent m relates to particle size and shape. As suspended sediment modelling was focused only on surface waters, the effect of hindered settling in bottom waters beyond a threshold concentration ($C > C_{HS}$) was ignored. Various combinations of K and m values were tested to identify the configuration that best reproduced the TSS field observations, as quantified through model performance statistics.

Model performance statistics included the mean of observed ($\bar{O}$) and predicted ($\bar{P}$) values, and the sample standard deviation of observed (s_O) and predicted (s_P) values (Willmott, 1982). The average error ($AE = \Sigma(P_i - O_i)/N_s$), with N_s being the sample size, was calculated as a measure of aggregate model bias in order to identify the model's tendency to over- or under-estimate observed values, while keeping in mind that positive and negative discrepancies can cancel one another (Zhang & Arhonditsis, 2008). The magnitude of the model's prediction accuracy is shown by the mean absolute error ($MAE = \Sigma|P_i - O_i|/N_s$) and the root mean squared error ($RMSE = (\Sigma(P_i - O_i)^2/N_s)^{0.5}$), the latter of which is sensitive to the inaccuracies of outliers (Willmott & Matsuura, 2005). The relative error ($RE = \Sigma|P_i - O_i| / \Sigma O_i$) was calculated in order to express error as a percentage of the observed values (Arhonditsis & Brett, 2004), noting that this measure is dependent on the magnitude of the variable itself. The index of agreement ($IOA = 1 - [\Sigma(P_i - O_i)^2 / \Sigma(|P_i - \bar{O}| + |O_i - \bar{O}|)^2]$) was also analyzed as a standardized measure of the model's error between 0 and 1 (Willmott, 1982).

5.3.4 Target Setting Approach

This study selected a TSS concentration of 10 mg/l as an ecosystem threshold for coral reef health. TSS threshold values for coral reef conservation can be highly controversial as factors such as local hydrodynamics, solar radiation, coral physiology and sediment characteristics can all interact to cause varied effects (Brodie et al., 2009). The threshold value of 2 mg/l is used for the open coastal waters of the Great Barrier Reef (GBRMPA, 2010), while 5 mg/l is used in the Caribbean waters of Barbados (Barbados Government, 1998) and 50 mg/l has been reported as a threshold in Western Australian reefs (Gilmour, 1999). The value of 10 mg/l was selected because it has been reported to be a tolerance threshold for chronic stress on coral reefs in the Caribbean (Rogers, 1990) and is a value that has also been accepted by other authors (Fabricius, 2005; Erftemeijer et al., 2012; Bartley et al., 2014), including previous studies of the Rosario Islands (Restrepo JD et al., 2016).

Three control points were selected to define the spatial limit beyond which the selected ecosystem threshold of 10 mg/l should not be exceeded. The three control points were established in Bocagrande strait, Varadero canal and the Bocachica navigation canal (Fig. 5.1). By establishing the control points in the straits that separate Cartagena Bay from the marine waters outside the bay, this approach effectively defines the bay as a mixing zone in which TSS concentrations can acceptably exceed the ecosystem threshold value. This spatial delineation could be considered logical as prominent coral reefs no longer exist within the bay, while the functioning reef of Varadero inhabits the area just outside the bay. This delineation also responds to previous research which showed that TSS concentrations above 10 mg/l regularly extended outside Cartagena Bay, contributing to the chronic stress of the Rosario Islands (Restrepo JD et al., 2016).

Time series of the results generated by the model simulations were extracted at the three control points to analyze the dynamics of TSS concentration. This modelling approach optimized the available information as it permitted the analysis of data at a fine spatial (every 0.7 m depth from the surface) and temporal (hourly) resolution over the entire month-long period of the simulation, as opposed to the monthly snapshots at fixed depth of the monitoring program. The time series were analyzed to assess whether the water quality at a control point exceeded the threshold value. This step required the definition of "exceedance", an important criterion with respect to the overall time period. For example, compliance with the EU's Water Framework Directive is reached when the median concentration of a parameter over a 5-year observation period (based on annually aggregated data) remains below the threshold concentration (Schernewski et al., 2015). However, this actually implies that the parameter's concentration could be above the threshold value for up to 50% of the time, which does not support the concept of ecosystem relevance. Exposure to above-threshold concentrations for extended periods of time would constitute a chronic disturbance, which restricts ecosystem recovery (Connell, 1997). On the other hand, it may be impractical to use a criterion that requires the concentration to be below the threshold for 100% of the time, as occasional pulse events can be expected to occur that result in threshold exceedance, though such an event could be considered an acute disturbance from which an ecosystem can recovery (Connell, 1997). In this consideration, this study established the criterion of 10% as the maximum allowable time of threshold exceedance and thus extracted the 90th percentile value from the time series results to compare with the threshold value.

Multiple scenarios of reduced sediment load were simulated with the model in order to evaluate the effect of load reductions on TSS concentrations at the control points. In order to not disturb the prevailing hydrodynamics, reduced sediment load scenarios were configured by reducing TSS concentrations in the Dique Canal, while the canal's freshwater discharge was left constant. The simulated scenarios of reduced TSS in the canal included concentrations of 225, 150, 100, 50 and 25 mg/l. The time series results of TSS concentration at control points in the bay under reduced TSS load scenarios were analyzed for both the rainy and transition seasons. Plots of the 90th percentile value versus mean TSS load in each scenario permitted the identification of the end-of-river target load. The corresponding reductions in TSS load required to meet the end-of-river target were then calculated as a percentage of the current load.

5.4 Results

5.4.1 Monitoring Results

Monthly mean TSS concentrations in the Dique Canal ranged from 30 to 400 mg/l with a pronounced seasonal variation and higher concentrations occurring from April to June and Oct.-Nov. (Fig. 5.2). While low concentrations were generally observed in the months of Jan., Feb., July and Aug., distinctly lower concentrations were found in the canal during the months of July-August 2015 and Jan.-March 2016, likely related to the El Niño phenomenon occurring during this period. This period of decreased flow yielded lower discharge values of 30-60 m^3/s in Sept. 2015 and Jan.-Feb. 2016 (Fig. 5.2). Peak discharge values of 215-230 m^3/s were found in Oct.-Nov. 2016, while discharge during the rest of the study period ranged between 110-175 m^3/s. These measurements yielded a wide range of TSS load estimates, from 89 t/day in Jan.-Feb. 2016 to 6.4 $x10^3$ t/day in Oct.-Nov. 2016 and a mean load ($\pm$ standard deviation) of 3.2 $\pm$ 2.2 $x10^3$ t/day over the one-year period from Dec. 2015 to Nov. 2016.

TSS concentrations in the bay ranged from below detection limits (D.L. <4.2 mg/l) to 76.0 mg/l (Fig. 5.2) with a mean of 20.1 mg/l, a median value of 15.0 mg/l and a standard deviation of 14.1 mg/l. Greater TSS concentrations were generally found in bottom waters than in surface waters. Station B5 directly north of the canal outlet yielded higher concentrations than other stations, particularly during the wet season, while similar concentrations were found at station B3 (west of the canal) during the transitional season. High TSS concentrations were also observed during the dry/windy season in the southwest part of the bay (stations B1, B2). Mean TSS concentrations in the straits of Bocachica (B2) and Bocagrande (B8) were 16.3 $\pm$ 13.0 mg/l and 17.0 $\pm$ 11.6 mg/l, with ranges of <4.2-60.0 mg/l and 5.3-51.6, respectively. High TSS concentrations were also found outside the bay at the offshore stations ZP1 and ZP2, with means of 22.1 $\pm$ 18.7 mg/l and 22.4 $\pm$ 20.1 mg/l, respectively, demonstrating the influence of sediment plumes in the Rosario Islands.

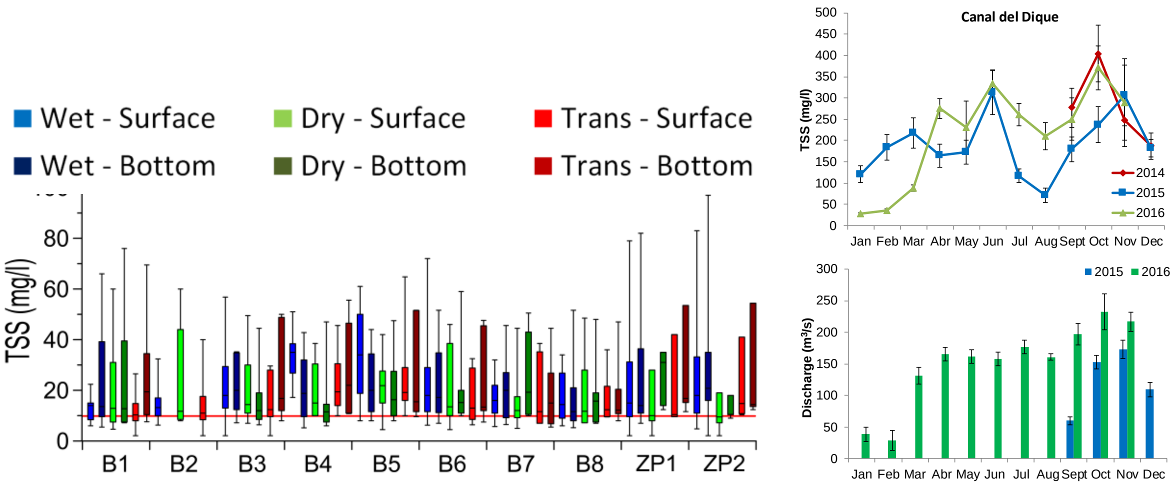


Figure 5.2. Left: Box plot of total suspended solid concentration (mg/l) measured monthly from Sept. 2014 to Nov. 2016. Colours differentiate the wet (Sept.-Dec.), dry (Jan.-Apr.) and transitional (May-Aug.) seasons as well as surface and bottom waters (see legend). The red line drawn across the plot represents the threshold value of 10 mg/l. Station locations are shown in Figure 5.1. Right: Mean TSS concentration (above) and mean discharge (below) measured in the Dique Canal. Error bars show standard deviations.

5.4.2 Modelling Results

The calibrated model adequately reproduced field observations of surface water TSS concentrations during the 2016 rainy (Oct.-Nov.) and transition (June-July) seasons, as shown by the different performance statistics analyzed (Table 5.1; Fig. 5.3). Mean TSS values observed during the two simulation months were in close agreement with mean values produced by the model, yielding differences of just 0.1-0.2 mg/l. Greater standard deviations were found among values predicted by the model than those of observed data, showing that the model augmented the spatio-temporal variability of TSS in comparison to observations. However, the additional variability generated by the model was equally distributed above and below the means, as shown by the very small values of average error (AE: -0.2, 0.1 mg/l) which suggest that the model showed almost no bias in over- or under-estimating TSS concentrations. MAE and RMSE values of 2.8-3.5 mg/l and 3.9-4.1 mg/l, respectively, suggest adequate model performance as these error values are within the range of the laboratory detection limit (4.2 mg/l) and the natural variability shown by triplicate sampling (standard deviations between 0.2-9.6 mg/l). However, the values of RE are rather high (21-32%), though this may be expected given the magnitude of uncertainty due to natural variability and laboratory limits. It must also be taken into consideration that observations are made at single points in time while model values are obtained at hourly intervals for comparison, which could also contribute to the large RE and greater variability of predicted values. The Index of Agreement (IOA) value of 0.9 (out of a maximum index value of 1.0) also supports the assessment that the model performs adequately.

Table 5.1. Statistics of model performance in predicting suspended sediment concentration: mean and standard deviation of observed ($\bar{O}$, s_O) and predicted ($\bar{P}$, s_P) values, average error (AE), mean absolute error (MAE), root mean squared error (RMSE), relative error (RE), and index of agreement (IOA).

Parameter	Season	$\bar{O}$	$\bar{P}$	s_O	s_P	AE	MAE	RMSE	RE	IOA
TSS (mg/l)	Transition	12.6	12.4	3.8	6.5	-0.2	3.5	3.9	32%	0.90
	Rainy	15.6	15.7	2.4	5.6	0.1	2.8	4.1	21%	0.90

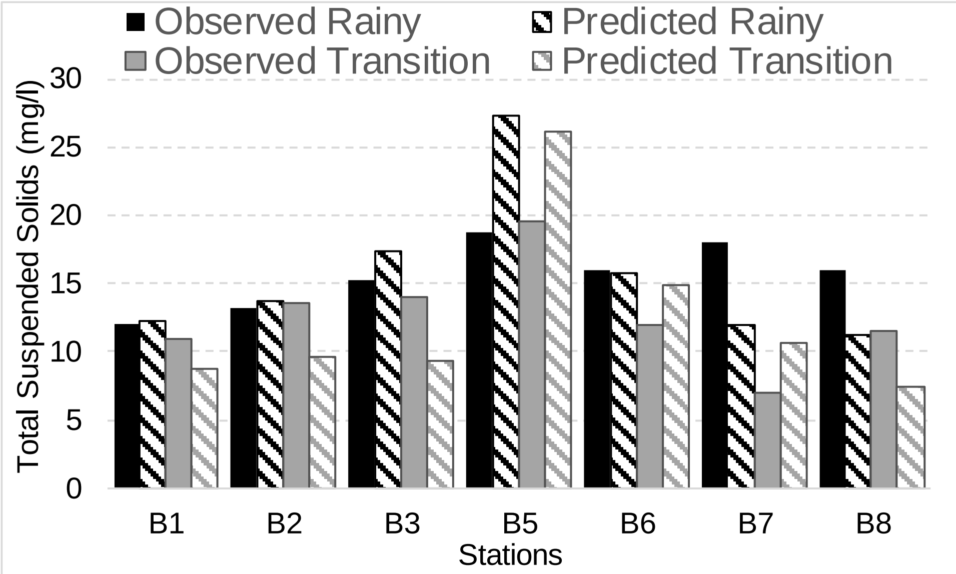


Figure 5.3. Predicted model results compared to measured observations of total suspended solid concentration in surface waters at the end of two simulations (rainy and transition seasons) at calibration stations in Cartagena Bay (see Fig.5.1).

Time series analyses of model results at the designated control points (B0, B2, B8) in the bay's straits showed that TSS concentrations were frequently above the threshold value of 10 mg/l (Fig. 5.4). At station B2 in Varadero canal, TSS was consistently above the threshold value in both the rainy and transition season simulations with maximum values greater than 20 mg/l. At station B0 in Bocachica canal, TSS was also consistently above the 10 mg/l threshold during the rainy season, particularly in the top 3 m of surface water. During transitional season, TSS values at B0 oscillated around the threshold value, with a mean concentration of 9.9 mg/l. At station B8 in Bocagrande strait, concentrations were consistently below the threshold during the transition season, while rainy season values were consistently above 10 mg/l in the top 3 m of surface water.

Figure 5.4. Time series of total suspended solid concentration (mg/l) during simulations of present load conditions of the transition (above) and rainy (below) seasons at control points B0 (left), B2 (center) and B8 (right). Coloured lines represent results at different depths from the surface (see legend). The red dashed line highlights the threshold value of 10 mg/l.

5.4.3 Target Setting

The 90th percentile values of TSS concentration at the three control points during simulations of the 2016 rainy and transition season were all above the 10 mg/l threshold value, with the exception of station B8 during the transition season (Fig. 5.5). Mean TSS loads during these months were approximately 6,373 t/d (rainy) and 4,295 t/d (transition). Simulated scenarios under conditions of decreased sediment load showed that load reductions gradually lowered the 90th percentile values of TSS at the control points, with a more pronounced effect at loads below 2,000 t/d (Fig. 5.5).

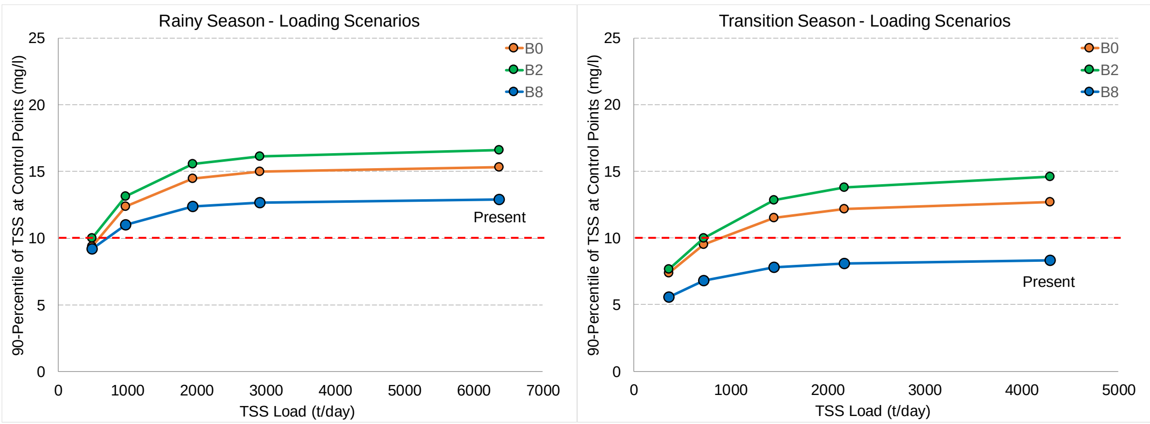


Figure 5.5. TSS loading scenarios of the Dique Canal into Cartagena Bay in the modelled rainy (left) and transition (right) seasons. The present and reduced TSS load scenarios (t/day) in the Dique Canal are shown along the x-axis. The 90th percentile values of TSS concentration (mg/l) at three control points (B0, B2, B8; see Fig. 5.1) during the simulated 1-month time series (see Figs. 5.4 & 5.5) are shown along the y-axis. The red dashed line highlights the threshold value of 10 mg/l.

Under the transition season scenarios, compliance of the 90th percentile value with the threshold value at all three control points was eventually reached at a TSS load of 723 t/d. Under the rainy season scenarios, even further reductions to a TSS load of 486 t/d were required to achieve compliance of the 90th percentile value with the threshold, likely due to the greater discharges during the rainy season generating faster surface currents and a broader dispersion of sediment plumes. Figure 5.6 demonstrates the effectiveness of these reduced loads as TSS concentrations are shown to be consistently below the 10 mg/l threshold value at all three control points throughout the duration of both the rainy and transition season simulations.

Figure 5.6. Time series of total suspended solid concentration (mg/l) during simulations of reduced load conditions of the transition (above) and rainy (below) seasons at control points B0 (left), B2 (center) and B8 (right). Coloured lines represent results at different depths from the surface (see legend). The red dashed line highlights the threshold value of 10 mg/l.

The identified end-of-river target loads reveal the need for significant reductions in the upstream watershed. Under rainy season conditions, decreasing the load from 6,373 t/d to 486 t/d represents a reduction of 92% (Table 5.2). Transition season conditions would require a reduction of 83% to decrease loads from 4,295 t/d to 723 t/d. If the present flow conditions in the watershed were to be maintained, to achieve these load reductions with the discharge levels observed in 2016 would require a considerable decrease in canal TSS concentrations, down to 25 and 50 mg/l in the rainy and transition season, respectively.

Table 5.2. Total suspended sediment (TSS) loading of the Dique Canal into Cartagena Bay under present conditions and the reduced target load scenario. Present discharge and concentration values are based on monthly measurements in the Dique Canal in the year 2016.

	Rainy Season		**Transition Season**	
Canal Parameters	**Present**	**Reduction**	**Present**	**Reduction**
TSS Load (t/day)	6373	486	4295	723
Discharge (m^3/s)	225	225	167	167
TSS Concentration (mg/l)	328	25	297	50
Percent Reduction		92%		83%

5.5 Discussion

5.5.1 Feasibility of Load Reductions

Of course, reducing TSS concentrations in the canal without reducing its freshwater discharge (Table 5.2) is an abstract concept, as naturally the discharge and TSS concentration are related. The reduced sediment load scenarios of this study were configured by only reducing TSS concentrations in the canal while leaving its discharge constant in order to not disturb the bay's prevailing hydrodynamics. Despite this simplification, the recommendation is to focus mitigation on the reduction of TSS loads in general.

The task of reducing suspended sediment loads by 80-90% would seem daunting to any environmental manager or decision-maker, especially considering the enormous 260,000 km^2 area of the Magdalena Watershed. However, the drastic reductions needed to ensure adequate marine water quality are simply a reflection of ongoing issues of watershed management. Between the years 2000 and 2011, the Magdalena streamflow and sediment load experienced increases of 24% and 33%, respectively, compared to the 1972-1999 year period (Restrepo et al., 2018). These trends reflect poor land management practices, as 79% of the catchment is classified to be under severe erosional conditions, and an intense rate of deforestation, as more than 70% of natural forests were cleared between 1980 and 2010 (Restrepo et al., 2015).

One of the solutions proposed to decrease these flows of freshwater and pollution into Cartagena Bay is a hydraulic intervention that plans to construct hydraulic doors along the Dique Canal (Fondo Adaptación, 2018). However, Tosic et al. (2018) have shown that reducing freshwater discharge into the bay would result in slower water renewal time scales, which would actually worsen the bay's water quality issues as local pollution sources would then persist with less seawater renewal. Regardless, the closing of the Dique Canal would still result in the Magdalena River's full flow volume discharging to the coast of Barranquilla to the north (see Fig. 1), from where sediment plumes can still impact the reefs of the Rosario Islands (Restrepo et al., 2006). It is therefore imperative that efforts to reduce fluvial sediment loads be focused in the watershed itself.

The implementation of Best Management Practices (BMP) to the wide range of activities in the Magdalena Watershed would be a recommendable initiative to improve erosion rates. For example, BMP implementation on farming practices in the US State of Ohio, such as conservation tillage or no tillage, decreased river TSS annual loadings by 50% over about 5 years (Karr & Yoder, 2004). A large proportion of load reductions can also be achieved quite economically, as Roebeling et al. (2009) show that BMPs can reduce TSS loads by 35% and dissolved inorganic nitrogen (DIN) loads by 50% at no additional cost (and potential benefit) to farmers.

Considerable load reductions, such as the 80-90% target of the present study, have also been proposed in other watersheds. Recommended reductions for the watersheds adjacent to the Great Barrier Reef (GBR) include targets of 50-63% for TSS (Brodie et al., 2009, 2017) and 80-90% in DIN loads (Wooldridge et al., 2015). The most substantial of these reductions is needed for the Burdekin River watershed, which has an area of 130,000 km^2, is the largest single exporter of suspended sediment (~4 $\times 10^6$ t/year) to the GBR and contributes 25% of the total average annual load exported from the GBR catchment area (Kroon et al., 2012). While the Magdalena watershed is just twice the size of the Burdekin watershed, the Magdalena's average annual sediment load of

188.2 ×10^6 t/year (Restrepo et al., 2015) is nearly 50 times greater than that of the Burdekin, underlining the immense amount of work needed to improve watershed management in Colombia.

Without improved management in the Magdalena watershed, future sediment loads are expected to increase. Projections of the catchment's hydrological trends show that water discharge and sediment flux from the Canal del Dique will increase by ~164% and ~260%, respectively, by the year 2020 when compared with the average discharge of the 2000-2010 period (Restrepo et al., 2018). These trends correspond closely with land-cover changes in recent decades (Restrepo et al., 2015) while future precipitation increases due to climate change and the intensification of the El Niño-Southern Oscillation could also be expected to further heighten freshwater flows (IPCC, 2014; Paeth et al., 2008; Restrepo et al., 2018). Extrapolation of the relationships observed in Figure 5.5 suggests that such increases will worsen seawater TSS concentrations and thus exacerbate impacts on the coral reef system.

5.5.2 Feasibility of Achieving Ecosystem Improvement

This study showed that TSS load reductions of 80-90% are needed to ensure that sediment plumes do not extend outside of Cartagena Bay in concentrations exceeding the ecosystem threshold value of 10 mg/l. While this management strategy focuses on sediment plumes dispersing from Cartagena Bay, it should be noted that there are also other sediment sources affecting the Rosario Islands, including plumes from Barbacoas Bay to the south and the Magdalena River's principal outlet to the north at Barranquilla (Restrepo et al., 2006, 2016). Nevertheless, all three of these sources originate in the Magdalena watershed (see Fig. 5.1) and so improved land use management would effectively reduce the loads of all three sources.

Of more concern to the feasibility of achieving ecosystem improvement is that the reefs of the Rosario Islands may be impacted by a variety of stressors other than just sediment plumes (Restrepo et al., 2006; Tosic et al., 2017). Additional stressors such as excessive nutrients, rising sea temperatures or irresponsible tourist activities could likewise impair the marine ecosystem regardless of reductions in sediment plumes (Fabricius, 2005; Wild et al., 2011). Cumulative impacts of multiple stressors make the ecosystem particularly vulnerable (Rogers, 1990; Connell, 1997; Erftemeijer et al., 2012; Wooldridge et al., 2015), which further supports the importance of mitigating sediment loads as one of the principal stressors.

5.5.3 Benefit of the Approach

The demonstrated approach of setting targets for coastal water quality may be considered practical as its implementation does not require a long-term database. While the collection of two years of monitoring data and calibration of a coupled hydrodynamic-water quality model are labour-intensive activities to undertake, the method could still be carried out in a relatively short amount of time when compared to the time scales of policy development. The modelling approach also optimizes the amount of available data as the time series analysis provides more information than the snapshots of monitoring data. In this sense, the practicality of this method makes it accessible to countries like those of the WCR that are still developing their water quality policies.

Also important are the facts that this approach is science-based and ecosystem-relevant. Without considering nearshore hydrodynamics or ecosystem thresholds, the process of developing coastal effluent limitations can be relegated to an *ad-hoc* process of determining limits without the scientific knowledge needed to predict the consequences on coastal water quality and marine ecosystems. Of course, decision-makers may still prefer to utilize economically-based methods (e.g. Best Available Technology Economically Achievable), but the presented approach can at least inform decision-makers with science-based knowledge on the ecological relevance of a proposed target.

Furthermore, the presented method is appropriate for setting coastal water quality targets at the local-scale. Characteristics such as hydrodynamics, morphology and pollution sources vary greatly between different coastal zones, which makes each stretch of coastline, bay, lagoon or estuary unique. This variability is recognized by instruments such as the WCR's Cartagena Convention, which entails the distinction between Class 1 and Class 2 waters, or the GBR's Water Quality Guidelines (GBRMPA, 2010), which classify trigger values by water type (enclosed coastal, open coastal, midshelf, offshore). Eventually, site-specific targets should be developed for individual coastal water bodies (Schernewski et al., 2015) and by capturing local hydrodynamic and dispersion processes with a calibrated model, the presented approach can accomplish this.

The presented method could also be applied to other parameters or discharge targets. For example, control points could be established outside the swimming areas of Cartagena's beaches and microbiological parameters (e.g. E. Coli, Enterococcus) could be modelled to set targets relevant to the recreational use of beach waters. Similarly, targets for industrial effluents could also be set by this method using discharge and water quality data from a coastal industry. Of course, when multiple pollution sources are present in a single coastal water body, a method of load allocation would need to be applied to set targets for the multiple pollution sources and satisfy the total load target (Deng et al., 2010; Han et al., 2011). In the present study of TSS loads in Cartagena, however, load allocation was not necessary as it has been established that the Dique Canal is the principal source (99%) of sediments in the bay (Tosic et al., 2018a).

5.5.4 Limits of the Approach

As the approach does not incorporate long-term hydrological data, it does not provide knowledge on the relevance of the simulated discharge conditions in comparison to maximum flood plume conditions. The maximum discharge value used in this study (233 m^3/s) is similar to the high range value of 250 m^3/s reported by Tuchkovenko & Lonin (2003), but unfortunately, the present study's discharge data comprise the only monthly dataset in the Dique Canal outlet published to date. The relevance of the simulated rainy season conditions with respect to maximum flood conditions could be better verified with a long-term data set of daily discharge from upstream. As such, the potential for higher flood conditions could result in more threshold exceedance than predicted by this study's simulations of the year 2016, making the target loads and reductions a conservative estimate. However, considering the substantial TSS load reductions of 80-90% needed based on the modelled conditions, it is unlikely that refined information based on maximum flood conditions would be relevant in endorsing the need for improved watershed management.

A potential criticism of this method could also be in its selection of an ecosystem threshold value, rather than establishing the value by conducting an ecological study. While it is true that biological assessments provide better information than physical-chemical indicators (Karr & Yoder, 2004), it could also be argued that the urgent need to establish water quality standards justifies the use of the information available. The range of identified threshold values for a given parameter in a given ecosystem or water use may be refined in time as further research establishes a better understanding of the parameter's impact on the ecosystem or water use. Though conversely, it is also possible that the range of identified threshold values becomes broader in time, as further research could find a greater variability in ecosystem response to the combined effects of water quality and other stressors. Deliberation over the specific causes of coral reef degradation or the threshold value at which degradation occurs can require a lengthy process to come to a consensus. Such deliberation can deter action, and may be one of the reasons that action was not taken prior to the wide-scale decline of the world's coral reefs (Risk, 1999). What is important is not to delay water policy until ecosystem thresholds are defined with undisputed certainty, as this day may never come. Improved policy on controlling land-based discharges is urgently needed now, as current policy is inadequate, and the process of improved watershed management can be very long. If ecosystem thresholds become more refined in the future, the policy on land-based discharge limits can be updated accordingly.

5.6 Conclusions

In demonstrating the presented method to the example of Cartagena Bay, it was shown that TSS concentrations could be maintained below ecosystem thresholds within the extent of the bay by reducing TSS loads in the Dique Canal. To effectively ensure that the coral reef ecosystem threshold of 10 mg/l is not exceeded outside the bay, current load estimates of 6.4 $x10^3$ t/d (rainy season) and 4.3 $x10^3$ t/d (transition season) would need to be reduced to target loads of approximately 500-700 t/d, representing load reductions of approximately 80-90%. This considerable reduction needed in TSS loading reflects ongoing issues in the Magdalena watershed which has experienced severe erosional conditions and intense deforestation over the past four decades. The implementation of Best Management Practices in the Magdalena Watershed is recommended as this could contribute to substantial load reductions at no additional cost (and potential benefit) to farmers.

Policy and management actions to mitigate the impacts of sediment plumes and other types of pollution on coral reef ecosystems are urgently needed as these systems are particularly vulnerable to multiple stressors. The influence of sediment plumes in the Rosario Islands is evident as field observations show that TSS concentrations there were consistently above the threshold value, as were concentrations in the straits of Cartagena Bay. Without improved management, sediment impacts on the coral reef system will likely worsen due to future increases in TSS load caused by ongoing land-cover change and climate change. Current policy on coastal water quality in Colombia is inadequate to mitigate this issue as it does not include marine ambient water quality standards nor end-of-river targets, while discharge limits for coastal wastewater effluents were determined without established marine ecosystem thresholds or consideration of nearshore dispersion processes.

Coastal water quality policy in Colombia and other Caribbean countries could be improved with science-based, ecosystem-relevant methods such as that presented in this study. By optimizing monitoring data with a calibrated model that adequately reproduces TSS field observations, this method can be applied to effectively set targets for coastal land-based discharges without the need of a long-term database. This approach is thus accessible to countries like those of the Caribbean that are still developing their water quality policies. The presented method could also be applied to other parameters or discharge types. Such methods are beneficial to environmental management for the development of site-specific targets, which are needed in consideration of the natural and anthropogenic variability between different coastal zones and water bodies.

CHAPTER VI

General Discussion

6.1 Social Implications

While the previous chapters demonstrated various impacts to the natural environment, these impacts in turn have numerous social implications, many of which were documented through different research components of the multidisciplinary BASIC project (Restrepo & Tosic, 2017; Espitia et al., 2015; Garzón et al., 2016; Castillo, 2016; Ruiz-Díaz et al., 2017; Escobar et al., 2018). As shown in chapter II, the current state of the bay is likely to have impacts on the marine ecosystem as the majority of water and sediment quality parameters analyzed in this study were found to exceed national and international threshold values. The social implications of these ecosystem impacts are most likely felt strongest in the artisanal fishing communities that dwell in Cartagena's coastal zone.

Direct impacts on the fisheries may be expected in consideration of the oxygen levels found in the bay. Concentrations of dissolved oxygen, BOD and O_2 saturation were all found to be inadequate in comparison with ecosystem thresholds defined by Colombia (MinSalud, 1984), Cuba (NC, 1999) and Newton & Mudge (2005), respectively. Indirect impacts may also be expected on the fisheries due to the effect of degraded water quality on coral reefs, which provide fish with important habitats for feeding, refuge and reproduction. Throughout most of the year, measurements of total suspended solids, turbidity, phosphate, total phosphorus, nitrate and chlorophyll-*a* exceeded coral reef thresholds (Barbados, 1998; ANZECC, 2000; Fabricius, 2005; U.S.EPA, 2015). Considering the occasional exceedance of temperature thresholds as well (Wilkinson & Souter, 2008; Vega et al., 2011), the offshore coral reef ecosystems of the Rosario Islands are very likely impacted as these systems are particularly vulnerable to multiple stressors. The exception is the healthy Varadero reef just outside the bay (Pizarro et al., 2017), which appears to be less affected by the bay's sediments and associated contaminants due to the strong stratification of outflowing waters that maintain these substances in the surface layer until they are transported further offshore.

Research done by Escobar et al. (2018) on fish populations in the study area demonstrated the ecological impact that this pollution may have. The study showed that genetic diversity in fish populations was high, which supports the sustainability of the resource. However, the authors also found that the mean body length for all species was significantly smaller than body length at maturity. While this latter finding clearly indicates a need for improved fishery management, as fishermen should not be consciously capturing immature fish, it also implies that fish stocks in this coastal zone are low. Given the findings of this thesis and those of Escobar et al. (2018), the cause of this low abundance in the fish stock is likely due to both pollution and overfishing.

Reduced fishing returns for communities that traditionally depend on this resource will expectedly have a socio-economic impact. Social research within these communities by Castillo (2017) demonstrated that the fishermen are indeed aware of the negative impact of overfishing, as they themselves participate in fishermen associations that attempt to regulate this risk. However, it was also found that the economic pressures of reduced fishing returns often drive the fishermen to incompliance with their own regulations.

One of the solutions proposed by Castillo (2017), in collaboration with the communities themselves and other participating researchers, is the development of eco-tourism activities with local artisanal fishermen. Snorkeling at nearby reefs or visits to protected areas, for example, could help to balance the fishermen's income while also alleviating pressure on the fisheries. Though it would be essential that these activities be developed in a sustainable manner. The economy of these communities has already shifted from fishing towards tourism, in large part due to the growth of tourist activities at the nearby beach Playa Blanca (Restrepo & Tosic, 2017). While chapter II showed that water quality was mostly adequate at this beach, with only an occasional exceedance of the guideline value for enterococcus in recreational waters (WHO, 2003), there has been a steep increase in visitor numbers at Playa Blanca in the years since. These increases eventually reached a point when local newspapers reported visitor numbers tripling the beach's carrying capacity, forcing local authorities to implement occasional beach closure. This boom in tourism has surely been beneficial to the local communities, though the potential environmental impacts of exceeding the beach's carrying capacity could eventually make this economy unsustainable, and so the recommended initiatives to develop eco-tourism activities with fishermen would hopefully take this risk into consideration.

Medical research also demonstrated health impacts in coastal communities due to the decline of artisanal fisheries. Surveys of nutritional status in the community of Ararca found signs of a change in diet. In response to the scarcity of fish, local communities have been forced to compensate their diet with economic alternatives, such as fried chicken for example. Medical surveys found that the population exhibits nutritional alterations in comparison with a typical rural fisherman community. This finding is based on cases of poor nutrition, such as obesity, which signal altered metabolic profiles that increase the chances of cardiovascular diseases (Restrepo & Tosic, 2017).

Further to reductions in the quantity of available fish, there are also impacts due to the changes in the quality of fish caused by pollution. As shown in chapter II, the sediments of Cartagena Bay contain concerning amounts of mercury (Hg), nickel (Ni), copper (Cu) and chromium (Cr), with concentrations above the Threshold Effects Level (TEL) used to indicate potential risk (Buchman, 2008). Cadmium (Cd) concentrations also exceeded the TEL threshold during the 2014 rainy season, while lead (Pb) was below the TEL, and mercury even surpassed the Probable Effects Level (PEL) on one occasion in Nov. 2014. Some lesser examples of research have chosen to ignore the TEL guidelines and focus analyses solely on the PEL thresholds (INVEMAR, 2018), presumably under the assumption that "potential risks" are not of concern and only "probable risks" should be reported. However, ecotoxicological studies in Cartagena Bay prove this approach to be inadequate, as many of the metals found below the PEL values for sediments, including Cd, Cr, Pb and Hg, have been found at high concentrations in the bay's organisms.

Ecotoxicologaical studies carried out in the BASIC project analyzed the tissue of fish collected by artisanal fishermen in the coastal communities of Ararca, Barú, and Caño del Oro (Restrepo & Tosic, 2017). Fish tissue was analyzed in a total of 90 fish samples, including 3 different species (snapper, jacks, catfish) collected from 3 different fishing zones in March and November 2015. Laboratory analyses showed that the fish had accumulated Cr, Hg and Pb in their tissues, while Ni and Cu were not detected. They found that concentrations of Cr and Hg exceeded the maximum limits for human consumption recommended by the United Nations Food and Agriculture Organization and World Health Organization (FAO & WHO, 1995). Concentrations of Pb exceeded the maximum limit of consumption recommended for children used by the European Union (EU, 2006). These ecosystem impacts also concur with previous findings of high metal concentrations accumulated in the food chain of this coastal zone, including Cd in oysters (Manjarrez et al., 2008); Cd, Pb and Hg in corals (Torres & Torres, 2004); and Hg in fish, crabs, birds and humans (Alonso et al., 2000; Olivero-Verbel et al., 2008, 2009, 2013; Cogua et al., 2012).

The contamination of the bay's biological resources, such as fish, likely results in health impacts in the coastal communities and also holds economic relevance. Socioeconomic studies show that the fishermen would be willing to sacrifice significant income (271 Colombian pesos per kg of fish) to reduce mercury contamination in the fish by 1% (Garzon et al., 2016; Restrepo & Tosic, 2017). In terms of health, the most recent study in these communities found the presence of mercury in hair samples (Restrepo & Tosic, 2017). Five cases of congenital anomalies were also observed in members of the Ararca community (Espitia-Almeida et al., 2015) which cannot be linked directly with metal contamination but does raise concerns. Health issues in these communities are further compounded by poor water supply and education level, which have been associated with water-related diseases (Ruiz-Díaz et al., 2017). Ongoing research in the second phase of the BASIC project (2018-2020) will attempt to alleviate some of these health impacts through community education, support to health services and pilot projects for household systems of water distribution and sewage collection.

6.2 Management Recommendations

Given the wide range of pollution issues and social implications found in Cartagena Bay, management would surely benefit from many recommendations for improvement. Based on the specific findings of this thesis (chapters II-V), the following recommendations are made with a focus on each of the three principal sources of pollution identified in chapter III.

6.2.1 Watershed Runoff via the Dique Canal

The most significant progress in managing Cartagena's coastal zone could be made through lobbying the Colombian national government towards improved management of the Magdalena watershed. There is already an urgent need to control the fluvial fluxes of freshwater, sediments and nutrients flowing from this watershed into Cartagena Bay, and projections show these fluxes are increasing (Restrepo et al., 2018). Yet no current initiative for the joint management of this watershed is being effectively implemented. This is both a daunting and challenging task due to the basin's large scale (260,000 km^2), widespread population (80% of Colombia) and diverse anthropogenic activities, and so integrated management of this watershed would require collaboration between a great number of municipal and regional governments and environmental authorities. But the process of integrated watershed management must be initiated eventually and the results of the BASIC project present Cartagena's authorities with the opportunity to justify this need at the national level. Without improved watershed management, pollution impacts on Colombia's coral reef ecosystems will likely worsen due to the projected future increases in fluvial fluxes caused by ongoing land-cover change and climate change.

Watershed management should include objectives for downstream pollution loads. However, current policy on water resources in Colombia is inadequate in this regard as it does not include marine ambient water quality standards nor end-of-river targets, while coastal wastewater effluent limits were determined without consideration of nearshore dispersion processes. Science-based, ecosystem-relevant methods, such as that presented in chapter V, should be applied to establish effluent limits and end-of-river targets. This method showed that to effectively ensure that coral reef TSS thresholds are not exceeded outside Cartagena Bay, current TSS loads from the Dique Canal need to be reduced by approximately 80-90% to target loads of approximately 500-700 t/d. The demonstrated method used to quantify TSS load reductions could also be applied to other parameters, such as nitrogen and phosphorus, in order to establish similar objectives for the watershed.

While these reductions may seem drastic, they simply reflect the poor level of watershed management to date, as 79% of the catchment is classified to be under severe erosional conditions and intense deforestation has cleared over 70% of the basin between 1980 and 2010 (Restrepo et al., 2015). Comparable load reduction targets of 63% TSS (Brodie et al., 2009, 2017) and 90% DIN (Wooldridge et al., 2015) have been recommended for the Burdekin River watershed (130,000 km^2) adjacent to the Great Barrier Reef in Australia. Significant sediment and nutrient load reductions of up to 50% could be achieved at no additional cost (and potential benefit) to farmers through the implementation of Best Management Practices, such as conservation tillage or no tillage, in agricultural areas (Karr & Yoder, 2004; Roebeling et al., 2009). Though clearly, the important issues of intense deforestation, mining and urbanization would need to be addressed as well.

The alternative solution of constructing hydraulic doors along the Dique Canal (Fondo Adaptación, 2018) was shown to be ineffective in chapter IV. Firstly, closing the Dique Canal would still result in the Magdalena River's full flow volume discharging to the coast of Barranquilla to the north, from where sediment plumes can still impact the reefs of the Rosario Islands (Restrepo et al., 2006, 2016). Furthermore, the hydrodynamic model assessment of chapter IV showed that reducing freshwater discharge into the bay would result in slower water renewal time scales, which would actually worsen the bay's water quality issues as local pollution sources would then persist with less seawater renewal. While this hydraulic intervention may be recommendable to reconsider in the future, it would be crucial to first focus efforts on watershed management and the reduction of local pollution sources.

6.2.2 Nearby Sources of Industrial Wastewater

Pollution control of local effluents requires significant improvement as chapter III showed that nearby sources of industrial wastewater are a primary source of nitrogen and organic matter, which contribute to the bay's problems of eutrophication and hypoxia. Of even more concern could be the uncertainty concerning a myriad of other potential contaminants for which there is no available data. Industrial discharges in this coastal zone remain largely a mystery as policy until now has only required industries to report TSS and BOD concentrations in their discharges twice a year, while the industries themselves are responsible for sampling and the data is not publicly available. New policy has recently been developed for point-source discharges in Colombia's coastal zone (MADS, 2018) though this policy is inadequate for the management of pollution issues as it does not consider marine water quality thresholds or nearshore dispersion processes, and thus lacks the science-based foundation needed to ensure marine ecosystem relevance. Furthermore, this policy does not oblige industries to report volumetric discharge, making a calculation of pollutant loads impossible.

Management of industrial effluents would be recommended to, firstly, establish local policy for stricter monitoring regulations for industrial discharges. Without obliging industries to measure volumetric discharge and monitor water quality parameters with sufficient frequency (e.g. monthly), coastal zone management will lack the knowledge necessary to identify primary polluters. Secondly, the establishment of maximum discharge limits using science-based, ecosystem-relevant methods, such as that presented in chapter V, would be recommended to ensure that industrial discharges effectively permit estuarine waters in the bay's to comply with water quality thresholds. Given that at least 50 industries are present in the bay's coastal zone, a method of load allocation would also need to be applied to set targets for the multiple pollution sources and satisfy the total load target (Deng et al., 2010; Han et al., 2011).

6.2.3 Nearby Sources of Domestic Wastewater

Considering the substantial challenges of mitigating pollution from the upstream watershed and nearby industries, it would seem that the nearby sources of domestic wastewater are of least concern to coastal zone management in Cartagena. This is in part thanks to the city's recently improved sewage system which replaced the bay's submarine outfall with a new one discharging far north of the city. A focus on managing domestic wastewater can be expected considering Cartagena's economic dependence on tourism. While pollution-related ecosystem degradation presents significant impacts to perhaps thousands of artisanal fishermen and some twenty-five thousand coastal community members, city authorities are likely more concerned by the hundreds of thousands of tourists that visit Cartagena's beaches annually.

However, chapter II's findings of increased concentrations of fecal coliforms and enterococcus during the rainy season show that a potential sanitary risk to bathers persists in the bay and chapter III confirmed nearby sources of domestic wastewater to be the principal source of these contaminants. In fact, coliform loads from domestic wastewater are probably even higher than this study's estimate because of the added load from the city sewage system's occasional overflow during rainy conditions, which was not accounted for but evidenced by the coliform increase near the city during this season. A recommendation to mitigate the issue of occasionally inadequate recreational waters would be to improve the capacity of the city's sewage system.

Perhaps the most effective mitigation action would be to install a wastewater treatment plant in Pasacaballos (~10,000 people), which is the largest of the non-serviced communities populating the coastal zone (approximately 33,381 ± 10,274 people in total). Pasacaballos is located near the Dique Canal's outlet where wastewater impacts were evidenced by increased coliform concentrations during the dry season when the community's untreated wastewater discharge receives less dilution due to low-flow conditions in the canal. In order to ensure compliance with recreational water quality standards in the city's beaches, the target setting method of chapter V could be applied by calibrating the bay's hydrodynamic-water quality model for microbiological parameters such as fecal coliforms, enterococcus and E. coli.

6.3 Regional Application

Many of the pollution issues and management needs described in this thesis are not new to the Wider Caribbean Region (WCR). The Convention for the Protection and Development of the Marine Environment in the WCR ("Cartagena Convention") was adopted in 1983 to protect the Caribbean Sea. The need for this convention and its protocol concerning pollution from land-based sources and activities (LBS Protocol) was partly in response to the risks of water pollution, as most wastewater is discharged to the Caribbean coast without treatment (UNEP, 2004). Projects funded through the Cartagena Convention have undertaken coastal zone diagnostics, such as the heavily contaminated bays project (UNDP-UNOPS, 1999) which conducted studies in Cartagena Bay, Havana Bay (Cuba), Puerto Limón (Costa Rica) and Kingston Harbour (Jamaica). International collaboration also generated a regional assessment of land-based sources of pollution in the Caribbean (UNEP-UCR/CEP, 2010). The establishment of water quality standards and effluent limits is also outlined in the LBS Protocol, which entails the distinction between Class 1 and Class 2 waters, while recent work to prepare a State of the Caribbean Report (SOCAR) seeks to update these guidelines with more detailed water quality threshold values.

The experiences, findings and methods established through this thesis could contribute towards the protection of the Caribbean Sea by extending similar research to other countries in the region, many of which face problems and limitations similar to those of Cartagena. The advanced diagnostic carried out in chapter II would provide a good example for monitoring programs in other Caribbean countries. For example, this study demonstrates the importance of monthly monitoring due to seasonal variability and the need to monitor the full water column as opposed to just surface waters, aspects which are not always considered in national monitoring programs.

The integrated approach to assessing land-based pollutant loads presented in chapter III would also serve to improve the WCR's previous regional assessment (UNEP-UCR/CEP, 2010) which was limited in its load comparison by a lack of knowledge on the relative uncertainty and variability of each load estimate. The inclusion of confidence intervals in the analysis of chapter III would provide further clarity to analysts in the WCR, allowing them to better portray differences in results and identify where additional data gathering is needed. By plotting load estimates with confidence intervals, decision-makers are permitted to confidently base their actions on numerically supported conclusions, which would be beneficial to coastal management in any country.

The hydrodynamic modelling approach demonstrated in chapter IV would also be appropriate to apply in similar tropical bays throughout the WCR where pollution impacts pose an extensive threat to coastal waters, ecosystems, tourism and human health. This approach was shown to be successful in assessing water renewal times and in evaluating the effectiveness of proposed pollution mitigation scenarios. This represents important knowledge needed by environmental authorities and decision makers in any coastal zone and the calibration of a hydrodynamic model also provides coastal managers with a valuable tool that can be further applied to many other management purposes.

The establishment of land-based discharge limits would also be supported by the target setting method developed in chapter V. Such methods are needed in the WCR where countries are still developing their water quality policies and have a strong dependency on coastal resources and tourism. In addition to the method being focused on effectiveness for the receiving marine environment, given that it is science-based and ecosystem-relevant, it is also practical for developing countries as it does not require a long-term database. While the collection of two years of monitoring data and calibration of a coupled hydrodynamic-water quality model are labour-intensive activities to undertake, the method could still be carried out in a relatively short amount of time when compared to the time scales of policy development. The approach also optimizes the amount of available data as modelling analyses provide more information than the snapshots of monitoring data. In this sense, the practicality of this method make it accessible to countries like those of the WCR which are often limited by data availability. Furthermore, such methods are needed for the development of local-scale water quality targets, in consideration of the natural and anthropogenic variability between different coastal zones and water bodies. While this variability is recognized by the LBS Protocol in its distinction between Class 1 and Class 2 waters, local-scale targets will eventually be needed as well for different stretches of coastline, bays, lagoons and estuaries (Schernewski et al., 2015). By capturing local hydrodynamic and dispersion processes with a calibrated model, this can be accomplished by the presented approach.

6.4 Further Research

The findings of this research raised various other questions that would merit further research. Some of these questions will be addressed in the second phase of the BASIC project (2018-2020) which recently received continued funding through another grant (#108747-001) from the International Development Research Centre (IDRC) of Canada. Unfortunately, constraints in overall research capacity and funding will not permit investigation of all of the following research aspects within "BASIC-2", but these recommendations are made nonetheless for future researchers.

6.4.1 Water and Sediment Quality Monitoring

Results of the monthly monitoring program presented in chapter II permitted an analysis of 14 physical, chemical and biological parameters of water quality along with seven metals analyzed in sediment samples. Most of these parameters were found in concentrations considered inadequate for the natural marine environment, providing important knowledge for coastal management. Additional knowledge could be generated by analyzing other contaminants not assessed in this study.

Further research on metal pollution would be merited considering the results of chapter II, which analyzed Cd, Cr, Cu, Hg, MeHg, Ni and Pb. These were selected based on local laboratory capacities and an initial screening analysis done at Hill's Laboratory in New Zealand for a wider range of metals which also included Aluminum (Al), Arsenic (As), Cobalt (Co), Iron (Fe), Tin (Sn), Strontium (Sr), Vanadium (V) and Zinc (Zn). Though low concentrations of Fe and Sr were found in these preliminary results, various other metals were found above the TEL guidelines including Al, As, Co, Sn, V and Zn. In fact, As and Sn concentrations were found well-above TEL guidelines but their continued analysis was not possible due to limitations in local lab capacities. Meanwhile, other unmeasured metals such as Barium (Ba), Manganese (Mn) and Selenium (Se) could also be of interest considering the petro-chemical industries adjacent to the bay.

Two other important types of contaminants to consider are pesticides and hydrocarbons. Laboratory analyses of these contaminants are not commonly available and otherwise can be quite costly. However, pesticides have been found in the waters, sediments, and biota of Cartagena Bay (Castro, 1997; INVEMAR, 2009; Jaramillo-Colorado et al., 2015) and high levels of hydrocarbons have been found in the sediments and marine organisms (Garay, 1983: Parga-Lozano et al., 2002; Johnson-Restrepo et al., 2008). With sufficient resources, these previous studies could be advanced with more detailed monitoring programs that permit knowledge on the spatio-temporal variability and sources of these contaminants. Though pesticides are most likely sourced in the Magdalena watershed's agricultural areas, there are also some industries around Cartagena Bay, such as chemical and food processing plants, that may also contribute these substances to the coastal waters. Hydrocarbon pollution has long-been a known issue in the bay, given that petro-chemical industries form the majority of local industrial activities and the bay supports an active commerce of maritime transport.

There are also various chemicals that may be utilized in the Cartagena's industrial sector. Iodine (I) is applied in a wide range of industrial processes. Cyanide (CN^-) is also used in different chemical plants. Sulfides are likewise utilized in various petro-chemical industries as well as leather processing. However, none of these potential contaminants have been researched in this coastal area.

Eutrophication issues in the bay were quite evident considering the monitoring results of chlorophyll-a, oxygen, nitrate, phosphate and total phosphorus. It was also shown that algal blooms are most limited by light in this system, rather than by nutrients. Measurements of silicate, ammonium, total nitrogen, phytoplankton and zooplankton would all help to further comprehend the processes of primary productivity in the bay. Data on these parameters would also be useful for the calibration of a eutrophication model in the bay that could be applied to setting targets for nutrients in land-based discharges.

6.4.2 Pollution Source Assessment

6.4.2.1 Industrial Activities

There is much research still to be done at the local scale of Cartagena Bay. First and foremost, data is needed on the specific industries in this coastal zone. Chapter III identified industrial wastewater as a significant source of BOD and nitrogen loads to the bay based on data from previous studies. However, the lack of available data on these discharges did not permit a reliable assessment of the primary sources of metal pollution. As outlined in the previous section (6.4.1), there are various other potential contaminants that could be contributed by the industrial sector as well.

However, approximating individual loads in this area would be an enormous challenge because: i) what little data exists on discharges (biannual measurements of TSS and BOD concentrations) are not publicly available, and ii) the sector includes at least 50 different companies, which change periodically. Uncertainty surrounding this sector is so great that, regrettably, this research could not even confirm a definitive list of the industries currently operating in the bay with the corresponding environmental authorities. Obtaining the available discharge data and a confirmed georeferenced list of industries would be a great initial accomplishment in this regard.

Generating more data on the various other potential contaminants discharged from this area would be an excellent next step. There may be progress in this respect as Colombia's new legislation on land-based discharges in the coastal zone (MADS, 2018), which enters into force in early 2019, will oblige industries to monitor more than just TSS and BOD. Nevertheless, the policy does not address measurements of volumetric discharge, making load estimates impossible, while sampling will continue to be the responsibility of the industries themselves, making data quality doubtful.

As sample collection on the property of over 50 private companies would not be feasible, the monitoring program of the second phase of the BASIC project will focus on a series of sampling stations in coastal waters adjacent to this industrial sector. Though this method will not permit load calculation nor identify specific polluters, increased contaminant concentrations at a specific location could at least serve to identify sub-sectors that may be polluting more than others. Hopefully, monitoring data from the industries themselves will become available in the near future through the new legislation (MADS, 2018) which could then be combined with nearshore monitoring data for an improved assessment.

Moreover, knowledge of the general characteristics of industrial discharges could be used to narrow down the potential sources. A previous study by INVEMAR-MADS (2011) showed that among the types of industrial activities found in Cartagena, food processing and chemical plants are the most likely to contain high levels of organic matter and nutrients in their discharge. Therefore, pollution control efforts could also focus on the food processing and chemical industries for mitigation of the bay's hypoxia.

6.4.2.2 Urban Stormwater

As explained in chapter III, the domestic wastewater load received by the bay is likely even higher than this study's estimate due to occasional overflows of the city's sewage system. The flow through this system occasionally exceeds capacity, particularly during rainy conditions when it combines with stormwater runoff, resulting in direct discharge into the bay through the bay's old submarine outfall and backup outlets along the coast (personal communication with the city water authority, AcuaCar). This likely explains the rainy-season increases in coliform concentrations, particularly to the north near the city, shown in chapter II.

Different approaches could be taken to investigate this issue in more detail. For example, the available geographical data on population distribution, sewerage cover, topography and rain data could be utilized for a preliminary risk assessment in order to identify and prioritize potential risks in a spatial context. Nearshore water quality data could also be related to corresponding rain data for the development of a stochastic model that predicts the probability of coastal contamination. Eventually, a deterministic model of stormwater runoff could be developed with precise information of the water levels and discharges of the city's drainage network in order to quantify pollution loads under different conditions and prioritize mitigation. Some of these exercises will be attempted in further research through the phase-2 BASIC project.

6.4.2.3 Marine Sources of Pollution

Another local source of pollution, which is little-studied yet largely important, is the bay's maritime activities. Cartagena Bay is one of the most important ports of the Colombian Caribbean, including more than 57 docking areas (UNDP-UNOPS, 1999). Some of these docks are for touristic or private recreational purposes, while others are used for industrial facilities. Cargo can include a wide range of products, including oil, fuels, chemical products and general bulk cargo (Garay & Giraldo, 1997).

The main port of Cartagena has an average of 5,000 dockings per year, of which 22% corresponds to services, 29% to cargo, 32% to international navigation, and 17% to cruise ships (Garay & Giraldo, 1997). These maritime activities naturally pose a potential source of pollution to the bay in terms of the ships' ballast water discharges, the discharge facilities of the ports and docks, operational discharges of liquid cargo, and accidental cargo spills which would reflect the wide range of substances utilized in the industrial sector. There have been previous studies on these subjects (Lonin, 1999; Rendón et al., 2003; Lonin & Parra 2005; Arregocés & Cañón, 2015) which could be updated and taken further.

6.4.2.4 The Magdalena Watershed

As described in section 6.2.1, an important recommendation for management would be an initiative for the integrated management of the Magdalena watershed. The identification and prioritization of principal pollution sources within this watershed should naturally form an important part of this initiative, however quixotic the task may seem. As the watershed covers approximately 25% of Colombia's land area and includes 80% of the population, this would certainly be a national initiative requiring participation of numerous environmental authorities and researchers. This matter is further complicated by the fact that Colombia continues to transition from a long-term armed conflict which restricts access and information in many parts of the watershed. Nevertheless, such an initiative is needed and would hopefully be made possible in time. In the meantime, researchers could make progress through studies of specific types of pollution sources, such as agriculture, mining or urban areas, in attempt to produce general macro-analyses that would subsequently form the basis of more detailed research.

6.4.3 Hydrodynamic and Water Quality Modelling

The analyses conducted in the different chapters of this thesis could be improved by relating the discharge values measured in the Dique Canal with long-term data of upstream discharges. At the time of writing, upstream discharge data was not yet publicly available for the year 2016 but this data will eventually be made available and could be utilized to consider the present results in the context long-term hydrological variability and maximum flood conditions. As upstream discharge data at Calamar is collected daily, this information could also potentially be correlated with measurements made at the Dique Canal's outlet in order to create a long-term time series of freshwater discharge entering the bay. Such a time series could improve the model simulations carried out in chapters IV-V and permit hindcasting of the bay's hydrodynamics.

The hydrodynamic model's performance could be further improved. By incorporating additional sources of freshwater not considered in the model (e.g. smaller coastal canals, urban runoff), inconsistencies in the model's reproduction of surface salinities could be resolved. Further investigation of temperature dynamics during the warmer transitional season could help to improve the model's predictions of thermal stratification, which were less pronounced than observed temperature profiles. The coupled hydrodynamic-water quality model of chapter V could also be used to compute values of the short-wave light extinction coefficient (K_d) for each cell and at each time-step, thus incorporating the spatio-temporal variability of water transparency into the model's computations of heat fluxes. Calibration of the hydrodynamic model could benefit from continued evaluation of the more sensitive parameters, horizontal viscosity and bottom roughness. Velocity fields generated by the hydrodynamic simulations could also be verified in comparison to velocity measurements in the bay. Coefficients for particle turbulence in the model's Lagrangian transport could be optimized through calibration with field measurements.

The simulated scenario of increased discharge in chapter IV showed that the bay's water renewal would improve under such conditions, though recognizing that the increased pollution carried with heightened freshwater discharge was not included in this study. Further research with the TSS-calibrated water quality model could include projected increases in sediment fluxes in order to evaluate the balance between increased water renewal and increased sediment loads. This assessment could also be extended to other contaminants by calibrating the corresponding water quality parameters.

The calibration of other water quality parameters would also permit a broader application of the target setting method demonstrated in chapter V for suspended solids. The same method could be applied to other parameters or discharge targets, such as microbiological targets needed for the protection of recreational beaches or nutrient targets needed for coral reef conservation. This method could likewise be applied to set targets for industrial effluents in a similar manner to the demonstrated example of end-of-river targets.

The current model could be built upon towards the development of an operational model. Such a tool could provide environmental authorities and stakeholders with a real-time early warning system for different pollution issues in Cartagena Bay. Work on this topic is currently underway within the second phase of the BASIC project.

CHAPTER VII

Overall Conclusions

The following conclusions integrate the central findings of each of the previous chapters and reflect the aims, objectives, hypotheses and research questions posed in section 1.3. The format of this section is designed as a policy brief and thus intends to translate the technical results of this research into commonly understood language in order to convey the central concepts of these findings to decision-makers. For the scientific basis of these conclusions, researchers are referred to the individual manuscripts of chapters II-V.

7.1 The Present State and Seasonal Variability of Water and Sediment Quality

Pollution issues in Cartagena Bay are evidenced by water and sediment quality that is considered inadequate in comparison to national and international reference values. Monthly measurements of 14 physical, chemical and biological parameters of water quality along with an analysis of seven metals in sediments during the year 2014-2015 yielded results showing that values of nearly all of these parameters were inadequate for marine conservation or recreation. In the case of water quality, the bay has a pronounced seasonal variability controlled by different environmental factors. Rainy season conditions are characterized by peak freshwater discharge from the Dique Canal and occasional overflows of the city sewage system. These conditions generate increases in total suspended solids, turbidity, biological oxygen demand, nitrate, phosphate, total phosphorus, phenol, faecal coliforms and enterococci, particularly to the north near the city and in the central part of the bay where freshwater plumes tend to disperse during this season. Insufficient oxygen concentrations are found in the bay's lower depths throughout most of the year, though this problem is worst during the transitional season when temperatures are highest. During the dry/windy season, sediment plumes recede and water transparency improves in the bay, though this is accompanied by blooms of algal growth in the water column, which suggests that primary productivity in this system is limited by light rather than nutrients. The bay's sediments contain mercury, cadmium, chromium, copper and nickel concentrations that are indicative of potential impacts on marine life. Considering associated ecotoxicological studies that found mercury, chromium and lead in the tissue of the bay's fish, as well as findings of mercury in human hair from fishing communities, it should be recognized that the impacts of metal pollution are not limited to the bay's sediments. These findings present coastal management with a justification for mitigation actions and policy development, and it would be recommended to conduct further research on other potential contaminants, such as arsenic, tin and hydrocarbons.

7.2 Integrated Assessment of Land-Based Sources of Pollution

The primary land-based pollution sources responsible for the issues of hypoxia, turbidity and unsanitary conditions in the waters of Cartagena Bay were identified through an integrated assessment of pollutant loads. By combining various methods to compare the loads of coastal domestic wastewater, coastal industrial wastewater, and upstream continental runoff, and calculating confidence intervals for each load value, this approach can confidently inform decision-makers on the following primary pollution sources. Turbidity issues in Cartagena Bay can clearly be related to the Dique Canal, which discharges sediment and phosphorus loads that outweigh the local wastewater loads by 2-3 and 1-2 orders of magnitude, respectively. The principal source of coliforms is coastal domestic wastewater, as the load coming from non-serviced coastal populations is several orders of magnitude higher than that of the industrial sector and the Dique Canal. This conclusion is further evidenced by increased coliform concentrations found in the canal during the dry season, when low-flow conditions provide less dilution to the wastewater discharged by the adjacent Pasacaballos community, and considering that this estimate does not include domestic wastewater loads from the city's sewage system that occasionally overflows, which would explain the increased coliform concentrations found to the north in the rainy season. The issue of hypoxia may be associated with all three of the sources evaluated (the canal, industrial and domestic wastewater) which all contribute significant loads of nitrogen and organic matter to the bay. It would be recommended to mitigate the identified sources through integrated watershed management, industrial effluent control and improved capacity of the city sewage system. Future research should focus on identifying the principal sources of metal contamination, assessing pollutant loads from individual industries and an initial approximation of pollution contributed by maritime activities.

7.3 Water Renewal Times and the Effect of Upstream Hydrological Changes

A semi-enclosed coastal water body, like Cartagena Bay, has a capacity to receive pollution that is reflected by its time scales of water renewal. As these time scales are sensitive to hydrological changes, coastal management needs knowledge on a water body's hydrodynamic processes in order to assess how future hydrological changes could affect its capacity to receive pollution. Calculations with a calibrated hydrodynamic model show that water flowing from the Dique Canal remains in the bay for an average of 3-6 days and a maximum of 10-20 days, the lower ranges of which occur when high discharge levels in the canal drive the water out faster. Meanwhile, the bay's entire water volume requires approximately 70-100 days for complete renewal, with the lower range corresponding to the windy season when greater vertical mixing increases seawater renewal. The assessment of future scenarios showed that increases in freshwater runoff caused by human development in the Magdalena watershed would result in faster water renewal in the bay. On the contrary, a decrease in freshwater discharge from the Dique Canal would result in slower water renewal, thereby reducing the bay's capacity to receive pollution. Therefore, mitigation plans to reduce the canal's discharge by constructing hydraulic doors upstream could in fact worsen the bay's water quality issues because local wastewater discharges would remain in the bay for longer time periods. While this hydraulic intervention may be recommendable to reconsider in the future, it would be crucial to first focus efforts on reducing local wastewater pollution and to advocate for improved management of the Magdalena watershed.

7.4 Targets for Coastal Water Quality Policy

Management of coastal pollution requires policy that regulates land-based discharges by establishing target loads for watershed outlets and coastal wastewater effluents. However, for these targets to be effective, they need to consider the coastal zone's nearshore hydrodynamics in order to verify that a given discharge target will result in adequate water quality in the receiving waters. Knowledge on coastal dispersion processes can be generated with a calibrated coupled hydrodynamic-water quality model, which was applied in Cartagena Bay to determine an end-of-river target for suspended sediment load from the Dique Canal. It was shown that suspended solid concentrations could be maintained below coral reef ecosystem thresholds within the extent of the bay by reducing suspended sediment loads in the Dique Canal by approximately 80-90% to target loads of 500-700 t/d. This considerable reduction reflects ongoing issues in the Magdalena watershed which has experienced severe erosional conditions and intense deforestation over the past four decades. The implementation of Best Management Practices is recommended in the watershed while a broader application of this method is recommended to set targets for other parameters and discharges. This study demonstrates how coastal water quality policy could be improved with science-based, ecosystem-relevant methods for target setting that are appropriate at the local-scale and applicable to the coastal zones of developing countries where historical datasets are unlikely to be available.

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