Sustainable Integrated Water Management Model with Public Health Strategies

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Abstract

Water management is a global challenge. Important facts of current concern in the water sector are: water scarcity threatened by the increasing consumption, safe drinking water supply resources threatened by climate changes and pollutants discharged from anthropogenic activities; and the accelerated urbanisation demanding adequate water supply together with the increasing wastewater generated by the growing urban population. These issues are becoming an imperative need that could be effectively addressed through adaptive water management strategies for the sustainable development of the societies worldwide.

Metropolitan areas exemplify the rapid increase of urban population within a relative small area, which consequently results in the overexploitation of water supplies. Together with this overexploitation, human health could be threatened due to the water-health nexus in terms of water quality and quantity. The specific case study of this research: the Metropolitan Area of Merida (MAM) in Yucatan, Mexico has been analysed in order to exemplify the use of a decision maker's tool to improve public health through the identification of major water pollutants and correlate them with waterborne diseases documented in epidemiologic statistics. The focus of this research was on two indicator contaminants: Faecal coliforms as microbial indicator of water quality, representing the non-conservative pollutants, and nitrate as chemical indicator of water quality, an example of a conservative pollutant that may persists in the groundwater for decades. Seven engineering interventions have been tested to identify most suitable management strategies through the following steps: 1. Quantify pollutants in the aquifer with the Sustainable Integrated Water Management Model (SIWMM), using a system dynamics approach; 2. Outcomes of the model served to quantify a) Public health risks posed from faecal coliforms through Quantitative Microbial Risk Assessment (QMRA), and b) Economic savings associated with pollutants reduction, 3. Develop cost benefit analysis of selected interventions, and 4. Identify the most suitable intervention in order to assist decision makers to cope with a sustainable supply of safe water and an integrated water management. The model framework developed in this thesis identifies the installation of soil absorption systems into septic tanks at household level, and installation of treatment plants for livestock wastewater as the most cost-benefit interventions of substantial positive impacts on groundwater quality and public health and, in addition, economic benefits.

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List of Acronyms and Abbreviations

AD/TF	Anaerobic Digester/trickling filter
AGR	Agriculture activity (sub-model acronym)
AQU	Aquaculture activity (sub-model acronym)
AS/EA	Activated Sludge/extended aeration
B/C	Cost benefit ratio
BMP	Best Management Practice
BOD	Biochemical Oxygen Demand
BOD_5	Five-day BOD (by standard test)
CBA	Cost-Benefit Analysis
CFU	Colony-forming units
COD	Chemical Oxygen Demand
DALYs	Disability-Adjusted Life Years
DU	Domestic urban activity (sub-model acronym)
DR	Domestic rural activity (sub-model acronym)
DW	Drinking Water
E. coli	*Escherichia coli*
EPs	Emergent Pollutants
ERA	Environmental Risk Assessment
EU	European Union
FC	Faecal Coliforms
GIS	Geographic Information System
GDP	Gross Domestic Product
HH	Household
IARC	International Agency for Research on Cancer
IND	Industry activity (sub-model acronym)
INS	Institution activity (sub-model acronym)
LIV	Livestock activity (sub-model acronym)
MAM	Metropolitan Area of Merida
MC	Monte Carlo
MCL	Maximum Contaminant Level
MPN	Most Probable Number
NO_3	Nitrate
OECD	Organisation of Economic Cooperation and Development
O&M	Operation and Maintenance
Pppd	Per person per day

PU	Public Urban activity (sub-model acronym)
QALYs	Quality-Adjusted Life Years
QMRA	Quantitative Microbial Risk Assessment
RA	Risk Assessment
RS	Ring of Sinkholes
SAGARPA	Ministry of Agriculture, Livestock, Rural Development, Fisheries and Food
SAS	Soil-Adsorption System
SDM	System Dynamics Modelling
SIWMM	Sustainable Integrated Water Management Model
ST	Septic Tank
UN	United Nations
UK EA	United Kingdom Environment Agency
(US)EPA	Environmental Protection Agency - USA
WDM	Water Demand Management
WHO	World Health Organization
WWTP	Waste Water Treatment Plant
YLL	Years of Life Lost
YLD	Years Lost due to Disability

List of Units and Definitions

CFU	Colony-forming unit (microbial unit)
ha	hectares
km	kilometres
l	liter
l/s	liter per second
lpppd	liter per person per day
m^3/s	Cubic meter per second
µg	microgram
mg	milligram
ml	milliliter
mg/l	milligrams per liter
MPN	Most Probable Number (microbial unit)

Aquifer	Is a geological formation where all the void spaces are filled with water (saturated).
Grey water	Wastewater from domestic activities such as laundry, dishwashing, and bathing
Groundwater	Water located beneath the ground surface in soil pore spaces and in the fractures of rock formations
Non-point pollution	Water body polluted from land runoff, precipitation, atmospheric deposition, drainage, seepage or/and hydrologic modifications
Point pollution	Water body polluted from well-located activities such as industry, institutions, domestic, etc.
Surface water	Water body collected on the ground such as a stream, river, lake, wetland, or ocean
Water use	Describes the total amount of water withdrawn from its source to be used. This helps to evaluate the level of demand from different users (i.e. industry, agriculture and domestic). It could take two forms – consumption or withdrawal.
Water consumption	It is the portion of water use that is not returned to the source after being withdrawn due to lost into atmosphere through evaporation or incorporated into a product or plant (i.e. corn stalk), and is no longer available for reuse.
Water withdrawal	Water diverted or withdrawn from a surface water or groundwater source. Most of this water is returned to the environment later after been used.

Chapter 1. Introduction

"Access to water supply and sanitation is a fundamental need and a human right"…"those without access are the poorest and least powerful. Access for the poor is a key factor in improving health and economic productivity and is therefore an essential component of any effort to alleviate poverty" (WHO/UNICEF, 2000).

"… intimate interconnections of exposure pathways and control mechanisms suggest that treating water, including supply and resource management, as an integral part of the risk factor unsafe WSH (Water, Sanitation and Hygiene) *is rational" (Ezzati et al., 2004).*

1.1. Global water crisis

Water is becoming an increasingly scarce resource. Even in the twenty first century, unclean water, inadequate sanitation and insufficient hygiene are the most significant risk factors of diarrhea, which is the world's second leading cause of death of children under five years (UNICEF, 2000; WHO, 2008; UNICEF/WHO, 2009). Across developing countries, there are more than 750 Million people without access to safe water, 2.5 billion people without access to sanitation and 1.8 million children die every year due to diarrhea and other water-related diseases (WWDR, 2014). Most of these diseases have been attributable to unsafe water, sanitation and hygiene. However, few countries consider water and sanitation as a political priority, in particular the Latin-American countries (Pruss-Ustun , 2008; Jefferies and Duffy, 2011; OPS, 2007).

Rapid population growth in urban areas has generated a new challenge in terms of water management: to provide sustainable water access for current and future population. A rational approach is needed to find a balance between too weak standards, resulting in the depletion of water resources, and too stringent standards which are needlessly costly to achieve. That right balance can be achieved through the implementation of integrated and sustainable water management strategies designed to identify and reverse trends of pollutant concentrations through global planning and monitoring systems (GCCC, 2009; IWRB, 2009; Wu, 2011; Padowski and Jawitz, 2012).

Water abstraction per capita has substantially increased over the last decades due to growth, globalization and virtual water trade. In Mexico, a member country of the Organization for Economic Co-operation and Development (OECD), water abstraction per capita increased from 730m^3 in 2002 to 740m^3 in 2006 (Figure 1).

It is important to notice that majority of this water abstraction increases is alongside the agricultural activity, since agriculture is the major water consumer in most developing countries, with an intrinsic relationship to the reliability of rainfall. For instance, within the OECD, Mexico is ranked among the countries with the highest water abstraction per capita, but considering other countries with comparable rainfall such as Spain and Korea, abstraction per capita in Mexico is comparable (Figure 1). In addition, one of the biggest challenges in Mexico is the sustainable management of the water cycle within urban areas (OECD, 1998b; San Martin, 2002; Robles-Morua, 2010; OECD, 2011; OECD, 2008; OECD, 2009).

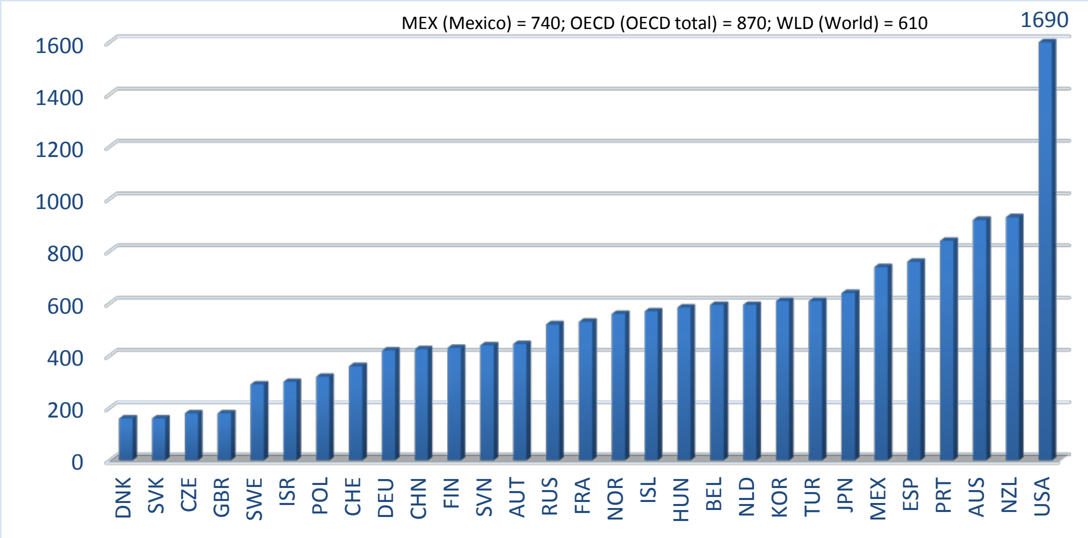


Figure 1 Per capita water withdrawals (m^3/year) for OECD countries. Source: OECD, (2011).

1.2. Water management challenges

Rapid global urbanization creates complex water management issues and the need to establish new adaptive approaches to environmental standards and regulations relating both to water quality and quantity, which respond to global and local challenges.

In terms of water quantity, large volumes of water are daily used for every social and economic activity at different levels. Major sectors demanding water for their particular uses are agriculture, domestic and industry, which are continuously increasing their water withdrawals (Figure 2). While industrial water use dominates water consumption in developed countries, the agriculture sector is the main water consumer in developing countries. Significant portions of the water consumed in different sectors returns to the environment as wastewater with very variable water quality characteristics.

Estimating volumes of wastewater per socio-economic activity and the corresponding concentration of pollutants per specific basin, catchment, district or country, can be a challenging task. The loads of water pollutants from municipal uses are particularly severe in developing countries due to the limited wastewater treatment, if any at all. For instance, a recent study in Bangladesh identified high concentrations of organic matter, total nitrogen and total phosphorus associated to discharges from septic tanks into water bodies (Tsuzuki et al., 2011). Another concern is the poor efficiency of existing wastewater treatment facilities, as reported in Thailand (Tsuzuki et al., 2009).

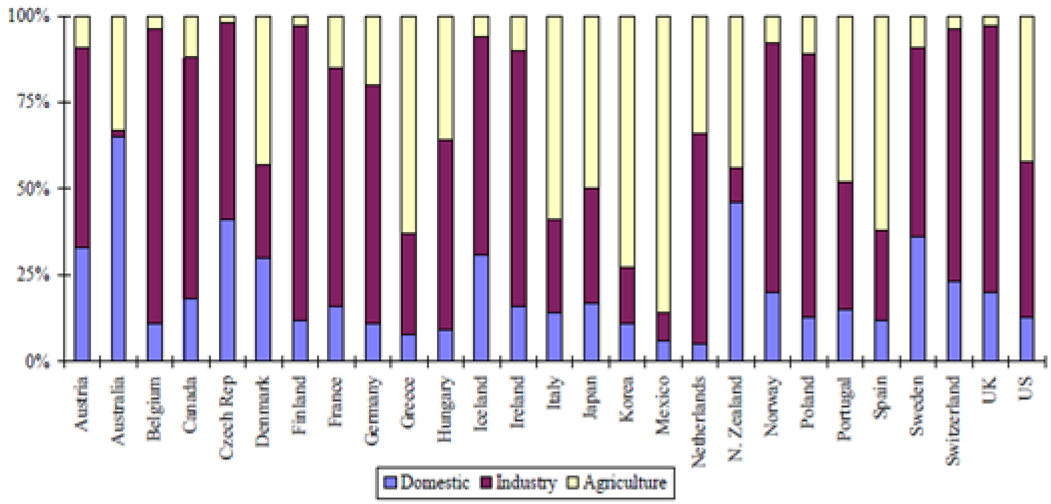


Figure 2 Water withdrawal by 3 major sectors in the OECD countries (%). Source: OECD, (1998a).

Even though, it is common practice to develop water quality legislation based exclusively on the most suitable water characteristics for specific uses (i.e. drinking water); however, new directives would need to take into account particular conditions related to local natural resources such as climate, catchment characteristics and changes in water flows, in order to establish more adequate water quality criteria. With this approach, sustainable management for further improvements could be achieved based on updated monitoring data and new technologies (Wu, 2011).

Because of urban areas generate large volumes of wastewater, high concentrations of contaminants such as organic matter, pathogens and heavy metals are expected in water resources, when poor sewage treatment is available in place. Therefore, there is a need to look at the whole system to propose integrated solutions (from the extraction point of the water supply chain, up to the final wastewater disposal), as well as sustainable solutions (forecasting future scenarios) for each individual problem to tackle (Leyva, 2010; Pokrajac, 1999).

Integrated and sustainable approaches have been used worldwide in order to develop water resource management and city water safety plans (Jefferies and Duffy, 2011). From an integrated approach, it is important to consider physical and hydro-geological conditions which describe natural resources at spatial and temporal levels. From sustainable approach, cultural, economic, environmental and political factors that influence access to natural resources should be considered in order to manage water supply, sanitation and public health by local governments and water supply/wastewater treatment providers (Jonch-Clausen, 2000). Therefore, a combined sustainable integrated approach could cover all possible aspects necessary for an improved water management.

Sustainable water management may require collaborative cross-sector networks and trans boundary partnerships. An example of this is the current dinaric project where governments of Albania, Bosnia & Herzegovina, Croatia, and Montenegro work together for the protection and sustainable use of the dinaric karst transboundary aquifer system (UNESCO-IHP, 2006; Nalecz, 2011; Stevanovic et al., 2012). It implies taking participatory interventions from practitioners and stakeholders looking at both beneficial and detrimental interactions at the human-water system nexus.

1.3. Water-health nexus

The International Drinking Water Supply and Sanitation Decade (1981-1990) emphasized a global awareness of the intrinsic nexus between water and public health. Increasingly noticeable is the lack of data available that could relate the public health impact of water quality and quantity. Emerging water-related diseases such as public poisoning with arsenic and cadmium in drinking water worldwide due to natural and anthropogenic sources have contributed to the initiation of research projects and studies in order to generate a more solid database in both developed and developing countries.

Positive and negative relationships can be found within the water-health nexus. For example, in Latin American and the Caribbean (LAC) countries (San Martin, 2002) reported negative impact in public health (i.e. incidence of cholera cases) due to the lack of water supply coverage. He also reported a positive impact in public health (i.e. nutrition indicators) due to higher coverage of water supply. These examples serve to indicate, that it is important to have and generate data related to the water-health nexus within geographical bases in order to identify current threats and hazards to be tackled, in short and long term prioritizing interventions.

1.4. Aims and objectives

Efficient water management strategies are required to rapidly adapt to current changes in urban areas. The aim of this research is to develop a framework-planning tool for urban water management in developing countries, as a strategy to assess public health improvements through cost effective interventions. This thesis will use a case study at the Metropolitan Area of Merida (MAM), in Yucatan Mexico, to develop such adaptive tool, and to propose potential strategic solutions.

In order to achieve this, the following specific objectives have been identified:

1. Estimate water quality and water quantity data in order to generate sensitive data input for the computer model named: Sustainable Integrated Water Management (SIWM) model.

2. Forecast water pollution concentration of selected water quality indicators by modelling them with the SIWM model, in order to identify main pollutants that could result in public health risks within a spatial location.
3. Develop Quantitative Microbial Risk Assessment (QMRA) with outputs from the SIWM model, in order to identify the most significant health risks associated to water supply.
4. Evaluate through cost-benefit analysis, potential engineering interventions to improve public health in order to support decision-makers in water resources management.
5. Evaluate to which extent the model framework developed for the MAM case study is applicable to other comparable case studies, and what modifications would be needed for less comparable case studies

1.5. Thesis outline

This thesis has been divided into nine chapters.

Chapter 1 gives an introduction and describes current water management challenges and outlines the aims and objectives of this research project.

Chapter 2 presents the state of the art for this research topic through a review of relevant published literature, to conceptualize water management practices in the past, present and future perspective, as well as a deep description of the water-health nexus and the importance of controlling current water pollutants of major concern. This chapter also introduces key components to develop the "Sustainable Integrated Water Management" (SIWM) model and the system dynamics modelling approach used.

Chapter 3 describes the case study of the Metropolitan Area of Merida (MAM), which was selected for simulation by the system dynamics model developed for this research. It includes current water management practices in the area, water policies and accessibility for the eight main socioeconomic activities demanding water in both urban and rural areas. It also explores current water threats of public health concern, in terms of pollutant's concentration in the aquifer.

Chapter 4 presents the first part of the methodology of this research, including three fundamental steps for the development of the system dynamics model (SDM): conceptualization, mathematical model and computer model (system dynamics model). Model assumptions, objectives and boundaries are also presented. It also describes the sub-models that integrated the SIWM model. In addition, the description of data input estimated for the model in terms of: water and wastewater flows, and pollutants loads.

Chapter 5 presents the second part of the methodology, which describes the process of modelling the two pollutants selected for the present research: Faecal coliforms (FC), and nitrate (NO_3), together with the description of the seven engineering interventions. At the end of this chapter the methodology of the cost-benefit analysis (CBA) undertaken with the results of the inventions is described in order to get familiarized with the potential outcomes of the CBA.

Chapter 6 presents the first part of the results: the results of pollutant concentrations predicted in the aquifer, which is the sole water supply for the MAM case study, together with the effect of the overall seven engineering interventions.

Chapter 7 presents the second part of the results: the results of the cost-benefit analysis. Costs are measured with available data from local infrastructure companies. Benefits are estimated through assessing changes in health risks associated with water supply quality, which is posed by the selected pollutants. Quantitative Microbiological Risk Assessment (QMRA) and, cost-saved for nitrate removal when excessive concentration is in the aquifer of the case study, in order to continue using as main water supply for the MAM case study.

Chapter 8 presents a discussion of the novel model framework developed in this research, including its applicability beyond the study area as exemplified by a case study of a karstic aquifer in Germany.

Chapter 9 presents the final conclusions and main recommendations for further research.

Chapter 2. Literature review

…"Researchers and practitioners alike are becoming more aware of the importance of whole-system integration, both at a disciplinary level and geographical scale" (Hannah et al., 2008).

This chapter introduces four key topics which serve as basis for the present research:

- Water resource management practices: sustainable and integrated approaches to water resource management are described, along with approaches to groundwater management, which are particularly relevant to karstic aquifers, that provides a solid background supporting the case study of this research.
- Water-health nexus: water pollutants (both microbial and chemical), and waterborne diseases are discussed in order to identify the impacts of water quality and quantity to public health. This section lays the foundation to understand the importance of engineering interventions in improving public health, sanitation and hygiene practices.
- Modelling and predicting contaminant concentrations in groundwater, with particular emphasis on karstic aquifers: Examples of modelling approaches related to two major chemical and microbial pollution indicators (i.e. nitrate and faecal coliforms) in vulnerable karstic aquifers are discussed.
- System Dynamics Modelling (SDM): the application of Vensim (i.e. a computer modelling software developed by Ventana Systems Inc.) in the water sector to improve current understanding and implementation of water management is described, together with its potential impact to improve public health.

2.1. Water management

Halving the proportion of people without access to safe drinking water and improved sanitation is one of the Millennium Development Goals set by the United Nations (WHO/UNICEF, 2006). To achieve this goal, it is important to recognise that the solution to similar needs may require different approaches, even within the same geographical area (San Martin, 2002; Taylor, 2011; Associates, 2008; Padowski and Jawitz, 2012). In order to tackle water issues within a specific area, it is important to identify the variety of pollutants discharged from different sources in a given zone, within a given water basin.

Perry, (2013) suggests that setting a relevant framework is the most effective foundation for addressing both current and future water resource management. Such framework was named as the *ABCDE + F* of water resource management, which stands for: Accounting, Bargaining, Codification, Delegation and Engineering (ABCDE) plus the corresponding Feedback (F) among these components. In particular, urban water management was related not only to surface water but also to groundwater, especially for those countries that rely mainly on groundwater such as Denmark (90%); Portugal (94%); Italy (89%); and Mexico (75%). Urbanization impacts the water balance in different ways such as: operation of water supply infrastructure, on-site sanitation, storm water management, and sewerage treatment plants (Marsalek et al., 2006).

The present research is based on a broad range of water management approaches from the literature, which were classified into two groups: sustainable and integrated approaches. Conceptual and computer models developed in this research are based on the combination of these approaches, thus it is named “Sustainable Integrated Water Management (SIWM) model”. Scientific contributions to the SIWM model are discussed below.

2.1.1. Sustainable approach

The sustainability approach for this research focuses on the four sustainability pillars: social, economic, political and environmental, interacting among the water cycle and human activities. Table 1 shows examples and the contributions of this approach to the present research.

Table 1 Examples of relevant sustainable water management approaches

Author	Description	Contribution
San Martin, (2002)	Findings of the Inter-American Development Bank’s Annual Meeting related to main impairments to promote improvements in water management for Latin America	Three water sustainability factors: environmental, economic and social were used in 2 area: agriculture and industry
Santana-Medina et al., (2013)	64 sustainable development (SD) indicators were classified. Qualitative analysis through active social participation result in a most inclusive and realistic approach	Participatory approach of authorities in water, health, agriculture, livestock, aquaculture and urban public and rural development
Howe et al., (2012)	SWITCH Project (Sustainable Water Management in the City of the Future) was an international research project aimed to tackle current water scarcity in 13 urban scenarios	Interconnection among water system and anthropogenic activities affecting the former to identify key human-water interactions
Brandes, (2003)	Water Sustainability Project (WSP) as part of the POLIS Project, uses “from supply to demand-side approach” for understanding of structure and dynamics of urban water	Water management proposed for this research is based on water supply and demand in a dynamic system

2.1.2. Integrated approach

The integrated approach is defined for this research as the water management that focuses on all the physical factors affecting the water cycle considering spatial and temporal conditions. Relevant literature (OECD, 1998a; Pokrajac, 1999; Jonch-Clausen, 2000; Liu et al., 2008; Arellanos, 2009) suggests that an integrated water resource management (IWRM) should involve stakeholders and scientists to fulfill human-water system as follows:

- Integration of freshwater and coastal zone management (saline intrusion)
- Integration of land use and water management (human development)
- Integration of alternative water sources (recycled water)
- Integration of green and blue water (freshwater)
- Integration of surface and groundwater (natural-environmental system)
- Integration of quantity and quality of water resources (watershed)
- Integration of upstream and downstream water (catchment integration)
- Integration of science and decision-making strategies (practical science)

Examples of Integrated Water Management (IWRM) approaches are: the administration for the River Delta in Iran, and the Nile River in Egypt, where national water resource management plans have successfully implemented an integrated approach from 2005 to 2007. Table 2 shows examples of integrated management schemes together with their relevance to the present research.

Table 2 Examples of integrated water management approach

Author	Description	Contribution
McFarlane D.J., (2005)	Integrated Water Supply Scheme (IWSS) is a demand-supply approach applied in three Australian case studies with specific water stress. Includes a water conservation optional system through behavioral changes.	Methodology used to: 1) identifying the gaps (direct and indirect factors influencing water demand); 2) the options, and 3) selection of options
Bueno et al., (2006)	Water Demand Management (WDM) approach reducing water withdrawals or consumption to protect or enhance water quality, by increasing both supply capacity and storage capacity	It served as a guide for the analysis of water engineering interventions
Brandes O.M., (2007)	Water soft path planning (WSPP) also known as “Back of the envelope” attempts to achieve a more comprehensive integrated scenario-based approach. It was applied in Canada with a Back-casting Framework (BEBF) at multi-scale scenario	Back-casting technique used to optimize a desired future (i.e. increase water metering, pollution-fee policy, incentives for water use reduction)
Turner, (2008); Turner et al., (2010)	Integrated Resource Planning (IRP) for urban water resources in Australia to reduce water stress due to climate change, analyzing demand and supply options. It includes five steps: 1. Data collection; 2. Demand forecasting; 3. Optional analysis; 4. Implementation; and 5. Program evaluation.	This methodology served to develop: 1) Overall process; 2) Current situation (baseline); 3) Developing interventions; 4) Implementing interventions; 5) Monitoring results.
Pahl-Wostl, (2009)	Adaptive Integrated Water Resources Management (AIWM), an interdisciplinary water management project based on complex socio-ecological system in line with the Water Framework Directives	Methodology used on the modelling process by testing and improving interventions
Jefferies and Duffy, (2011)	Decentralized Systems (DS) are designed to look all 7 components of management within the water cycle to optimize its management: 1) storm-water management 2) water conservation measurements 3) treatment for potable and not potable uses 4) energy recovery 5) nutrient recovery 6) source separation, and 7) landscape	The seven components were considered together with the multi-stakeholders approach for the inclusion of socioeconomic activities and the corresponding authorities in the model
Australian Government, (2011)	Integrated Resource Planning with Sustainability (IRPS) assessment for Urban Water Project with broader understanding of urban water demand analysis known as demand forecasting	Water demand forecasting was used to estimate flows from different users and to identify main pollutant sources

To summarise, both approaches offer advantages and disadvantages, which are presented in Table 3, and the decision of using one among the other relies mainly on the specific water management issues faced. The major advantage of the approach taken in this thesis is the combination of the sustainable and integrated approach, providing a broader view of the particular scenario under study and facilitating decision making to tackle specific concerns. In particular, the unified approach proposed in this research aims to help decision makers to tackle public health concerns.

Table 3 Advantages and disadvantages of sustainable and integrated approaches

Author	**Advantages/Disadvantages**	**Improvement of SIWM Model**
	Sustainable approach	
San Martin, (2002)	Adv. Water is treated as both environmental and economic resource, considering all sustainability pillars. Disadv. Limitation to current water management issues	The present model identifies potential engineering interventions and allows to forecast future water management issues
Santana-Medina et al., (2013)	Adv. The participatory approach allows a holistic identification of local sustainability indicators. Disadv. Subjective evaluation of indicators by participants, not essentially objective	Inclusion of all stakeholders of the water management provides a qualitative basis for sustainable development, but in addition, decisions will be based on quantitative considerations using the SIWMM
Howe et al., (2012)	Adv. Creates stakeholder platforms to solve local problems by sustainability-oriented solutions. Disadv. Lacking synchronisation between project stages	Synchronisation between project stages is facilitated by the integrated approach and the extended modelling time scale of the SIWMM
Brandes, (2003)	Adv. Demand-site management rather than increases in water supply capacity was used. Disadv. This approach focuses on urban water demands whereas in many developing countries the major water-user is agriculture	This soft-path approach for sustainable water management at urban level was extended in the present research by integrating both urban and rural areas for a more holistic solution
	Integrated approach	
McFarlane D.J., (2005)	Adv. Allows a consistent comparison between water supply and demand management options. Disadv. Sustainability issues remain to be implemented	SIWMM combines sustainable and integrated approaches to enhance decision support
KayagaBueno et al., (2006)	Adv. Water supply and demand have a common metric for strategic planning Disadv. Fully relies in stakeholders participation, and limited number of available data for decision making	Lack of data compensated by estimating non-available data, using suitable literature and local databases (over a 20 years time period)
Brandes O.M., (2007)	Adv. It looks at three levels to compare future scenarios with back casting. Disadv. Scenarios analysis is based on water reduction assumptions, taking water-use of a specific household type as basis for these calculations	The present thesis integrates a broader variety of water-users from urban and rural areas for a more comprehensive description of the real scenario
Turner et al., (2006), (2010)	Adv. It equally considers demand and supply-side options to close the supply-demand gap. Disadv. Mainly based on accurate demand forecasting which requires highly skilled users; does not consider adaptations	Application of the SIWMM does not require advanced knowledge. The model is flexible and allows adaptations, relating for instance demand forecast even in extreme situations

Pahl-Wostl, (2009)	Adv. Water management adaptations are considered within the complexity of the water system, with indirect consideration of sustainability. Disadv. Approach is at the local level; extension to regional levels would require modifications of the model framework	SIWMM provides a holistic approach on regional scale, considering management policies in line with national and international regulations, for present and future water management policy options
Jefferies and Duffy, (2011)	Adv. It tackles long-term issues considering the complex water system. Disadv. Main focus is improving urban water systems and solutions are based on cases studies that might not be generalizable	SIWMM incorporates various groups of stakeholders within the 8 different socio-economic sectors of the water system.
Australian Government, (2011)	Adv. It considers reduced demand and increased supply, including alternative sources to overcome increases in demand and to enhance supply-demand balance. Disadv. The focus is on urban systems to optimize water management; sustainability and cost-effectiveness analysis remain to be implemented	SIWMM includes sustainability assessment and cost-benefit analysis as part of the model framework

In terms of water management, the aim of this research is the development of a model capturing water-human interactions that considers local factors affecting water flow such as rainfall and recharge, and current water management practices, such as wastewater treatment infrastructure and level of treatment in water supply systems. In particular, challenges related to water management in Latin American countries were documented by Anton, (1993) as follows:

- *Lack of financial resources*: inadequate investment and maintenance of existing water supply networks; inadequate investment in water and sewage treatment in spite of increasing risk of contamination; less investment in replacement of obsolete systems and expansion of networks; increased water services cost due to elimination of subsidies.
- *Structural adjustment and the need for self-financing*: reduced expenditure on water management, supply and sanitation; poorest with no water access due to increased connection cost or discontinued services due to lack of payment.
- *Population growth*: as consequence of a decrease in death rates during most of the 20th century, and birth rates remains high, global population is still increasing. Even though birth rates in urban areas have decreased during last decades, water needs has increased due to urbanization and increasing per capita demand.
- *Inadequate protection of water resources*: hydrological catchment systems and recharge areas affected by unregulated water abstractions and wastewater discharges.
- *Lack of knowledge of existing resources*: inaccurate information for long-term decisions in urban water supply, such as types of local water resources, available

volumes, water renewal, vulnerability to contamination, and measures to develop, manage and protect water resources.

- *Inadequate water resources management*: limited operational management due to lack of qualified technical personnel, lack of coordination between decision-makers and technical experts, lack of comprehensive analysis of management practices.
- *Wasteful practices*: inappropriate consumption practices, lack of awareness of water value and water-saving technologies, pricing policies which do not promote conservation, lack in metering.
- *Inefficiency, politics, and bureaucracy in water management*: shortage of financial resources in water institutions often related to large amounts of money spent on salaries for politicians shifted to administrative charges in the water sector. Growth on administrative personnel is often paralleled by shrinking of technical personnel, with shortage of money for the latter.
- *Corruption*: water companies collect large amounts of money which they report for purchasing expensive materials which in fact are "commissions" to suppliers to profit from their administrative position in the public water company. In addition, ambiguous procedures for new household connections or maintenance promote the use of bribes to speed up the process.
- *Shortage of trained professionals*: as a consequence of shortage in salaries, experienced professionals prefer to leave water companies, lack of professional training due to low salaries and insufficient incentives.

Consideration of these challenges helped to inform the design of water management interventions for the case study of this research, in which public health improvements could be achieved by protecting water supply sources and considering the most cost-benefit strategy.

2.1.3. Groundwater management

Globally, groundwater is the largest available reservoir of water on Earth. 65% of groundwater abstracted is used for drinking purposes, 20% for irrigation and livestock, and 15% for industry and mining (Zektser and Everett, 2006). Millions of people in urban areas depend on groundwater as their principal source of drinking water. For example, in urban Nigeria almost 60% of the population use local wells.

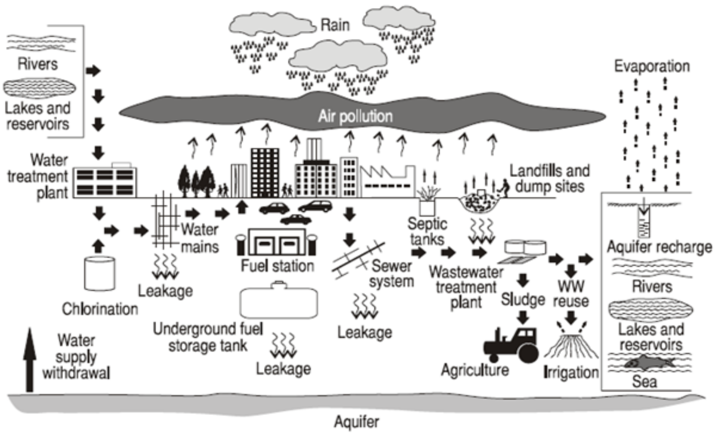


Figure 3 Urban water cycle. Source Marsalek et al., (2006)

Even though many countries rely on groundwater as the only drinking water source (Kabo-bah et al., 2014), Figure 3 shows variety of anthropogenic activities within urban areas that strongly affect groundwater systems (Marsalek et al., 2006). Thus, it is important to establish adequate treatment for wastewater and drinking water supply (Saleem et al., 2011; James and Martha, 2002; Bolger and Stevens, 1999). An example in this respect is the European Water Framework Directive (WFD), which has set groundwater quality standards that cope with up to date emergent pollutants identified (Zwahlen, 2003; Nalecz, 2011; UNESCO-IHP, 2006).

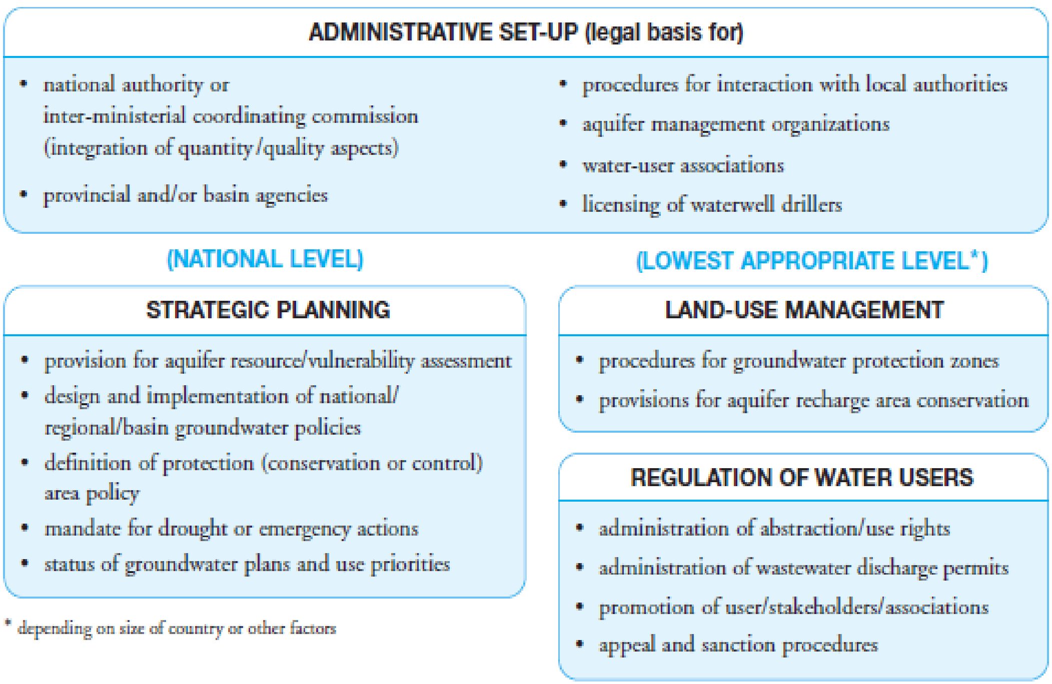


Figure 4 The role of public administration to facilitate groundwater management. Source: Nanni et al., (2003)

Moreover, sustainable groundwater management is not clearly or noticeably appreciated in city planning and decision-making because some countries consider groundwater as a hidden (and free) resource, hence it is barely or not accounted for at all in national statistics (Grönwall, 2010). Consequently, it is common to underestimate groundwater consumption as many indirect groundwater uses are barely identified (Hernandez et al., 1995; Grönwall, 2010).

Groundwater status & required services	**Resource Assessment** • Describe local hydrogeology in regional context with simplified maps and representative profiles • Calculate aquifer water balances, including surface water interactions • Appraise hydrogeological uncertainty and groundwater-level historical trends • Describe links to surface water and wastewater as potential resource and threat (Briefing Note 12)
	Quality Characteristics • Assess annual quality variations(Briefing Note 14) and presence of brackish/saline groundwater • Evaluate evidence for extent and possible causes of current pollution • Assess potential pollution risks from land use & aquifer pollution vulnerability (Briefing Note 8)
	Required Services • Discuss alternative socioeconomic scenarios with political leaders and water users • Predict future demands over planning period (10 or more years) • Assess aquifer target yields, allowing for environmental discharges • Draft options for aquifer stabilization or rational mining with exit strategy (Briefing Note 11)
Current management arrangements	**Institutional Provisions** • Appraise legal framework, customary arrangements and water permit system (Briefing Notes 4 & 5) • Assess responsibilities of all relevant organizations • Identify groundwater allocation criteria and priorities • Review resource-fee policy and enforcement
	Water Allocation & Usage • Summarize current position by sector graphs of historical trends in water use • Establish water-user profiles and water well inventory
	Monitoring Networks • Status of abstraction metering and estimation • Status of wastewater discharges affecting groundwater • Arrangements for aquifer water level and water quality monitoring
	Institutional Capacity • Assess 'enforceability' of water, land use and environmental law • Scope of user and other key stakeholder participation
Future management options	**Economic Analyses** • Estimate groundwater economic value (Briefing Note 7) • Assess feasibility of implementing direct and/or indirect groundwater pricing • Assess consequences of modifying macro-economic policies • Undertake systematic cost-benefit analysis of short listed options
	Definition of Options • Describe management options to achieve stated aquifer services (Briefing Note 3) • Consider conjunctive use and compare demand management options to supply-side augmentation • Appraise need to integrate groundwater and Surface water planning • Conclude on preferred option to pursue • Identify key tasks, responsible institutions, financial needs and implementation timetable
Implementation program	**User/Stakeholder Participation** • Appraise improvements in user/stakeholder participation required (Briefing Note 6) • Define action plan for their engagement • Prepare program for training, communication & publicity
	Monitoring & Review Requirements • Define improvements in monitoring needed for new management plan (Briefing Note 9) • Install improved management monitoring network • Propose timetable and process for internal/external evaluation of plan effectiveness

Figure 5 Checklist for the elaboration of groundwater management plans, from the World Bank. Source: Garduno et al., (2006)

Worldwide increased use of groundwater at the urban level has been common in recent years. This has cause sinking water tables as the aquifers have become overexploited. Abstraction rates compromise feasibility of aquifer recharge, by concentrating pollutants in reduced aquifer volumes (Kabo-bah et al., 2014). In order to avoid the incremental growth of groundwater contamination and the depletion of water table, public water administration should ideally be based on recommendations such as those made by the World (Figure 4) (Nanni et al., 2003).

To illustrate this we can consider some of the issues documented by Febles and Hoogesteijin, (2008) in the management of the aquifer underneath of the Metropolitan Area of Merida in Yucatan, these are: the lack of control of water pollutants, volumes of water abstracted and water quality of all types of wastewater discharged.

Following the checklist actions from the World Bank shown in Figure 5, a starting point could be the development of a framework of main and potential pollutant sources, taking into account current scenarios, as described in Chapters 4 and 5. Groundwater pollutants and their relations to public health are described in the following sections.

2.1.4. Groundwater critical hazards

Groundwater is contaminated by different sources such as acid rain, farming activities, contiguous aquifers or surface-water bodies with high concentrations of pollutants. Some of the major groundwater pollutants of human health concern are pathogenic bacteria and nitrate originated from human activities and animal waste; also heavy metals such as lead and copper from household corroded plumbing or from mining and construction activities.

For instance, a major source of nitrate is the spread of fertilizers used in agriculture (Akinyemi et al., 2014; USEPA, 2012b; Siarkos et al., 2014). Anton, (1993) has reported high concentration of nitrate in groundwater beneath agricultural areas, explained as a consequence of intensified fertilizer use since 1960’s, and documented a list of main sources of urban and rural pollutants (Table 4).

Nonetheless, groundwater has also a natural attenuation capacity to reduce pollutants. For instance, discharges from domestic sources accumulate disease-causing agents (i.e. faecal coliforms) that could be removed rapidly because of the filtering capacity of the aquifer. Although, such capacity depends on the aquifer geological formation, while some are very effective (silty sandstone) others not, such as karstic aquifers with more permeable structure that allows contaminants to move quickly through the aquifer conduits.

Table 4 Sources and potential groundwater contaminants

Source	Potential contaminants
Accidental spills	Various inorganic and organic chemicals
Acid rain	Oxides of sulphur (SOx) and nitrogen (NOx)
Agricultural activity	Fertilizers, pesticides, herbicides, and fumigants
Animal feedlots	Organic matter, nitrogen, and phosphorus
De-icing of roads	Chlorides, sodium, and calcium
Deep-well injection of waste	Inorganic and organic compounds, radioactive materials, and radionuclides
Hazardous waste disposal sites	Variety of inorganic and organic compounds (i.e. pesticides)
Industrial-liquid-waste storage ponds and lagoons	Heavy metals and various cleaning solvents and degreasing compounds
Landfills, industrial	Wide variety of inorganic and organic compounds
Landfills, municipal	Gases, organic and inorganic compounds (i.e. chlorides)
Land disposal of liquid and semisolid industrial waste	Organic compounds, heavy metals, and various cleaning solvents and degreasers
Industrial wastes leakage	Petroleum and derivate, toxic metals
Land disposal of municipal wastewater and waste	Organic and inorganic compounds (i.e. ddetergents and solvents), and pathogenic microbial, etc.
Mining	Minerals and acid mine drainage
Rainfall	Chloride, sulphate, organic compounds, etc.
Saltwater intrusion	Inorganic salts
Septic tank leaching fields	Organic matter, nitrogen, phosphorous, bacteria, etc.
Storage tanks, underground	Organic cleaning and degreasing compounds, petroleum products, and other hazardous wastes

Source: Adapted from Anton (1993); Bear and Verruijit (1987), Tchobanoglous and Schroeder, (1987); Ritter et al., (2002); Drew and Hotzl, (1999).

Furthermore, contamination due to overexploitation to supply urban and rural activities are increasing (Rodes et al., 1998). For instance, in 2012 in the United States an increasing number of groundwater sites (>120,000) do not meet the drinking water standards. Estimated costs for reaching the standards are 110 - 127 billion USD because of expensive treatment processes (Program and Council, 2012).

In Europe, three conventions are relevant to groundwater management: the UNECE Water Convention (Convention of the Protection and Use of Trans boundary Watercourses and International Lakes), the Convention on Biodiversity (CBD), and the UN Framework Convention on Climate Change (UNFCCC). Examples of groundwater projects undertaken in line with these conventions are: the groundwater monitoring system in Minsk Belarus 2007, and technical aspects of groundwater monitoring in Ukraine 2008, which generated valuable field data (Nalecz, 2011). Particularly, the INTERREG III C project from the Western Bug River Basin, identified that the critical water pollutant sources across Ukraine and Poland were industrial plants and uncontrolled sewage discharges (Nalecz, 2011).

From a political perspective, it is necessary to take into account the lack of regulation related to the discharge of wastewater into groundwater resources, which may be difficult to achieve due to the “hidden” character of groundwater resources.

2.1.5. Karstic aquifer

Karstic aquifers supply drinking water to 25% of global population (Leibundgut, 1998; Pulido-Bosch, 1999). A conceptual model to represent karstic aquifers was developed by the European Commission in 1995 (Figure 6).

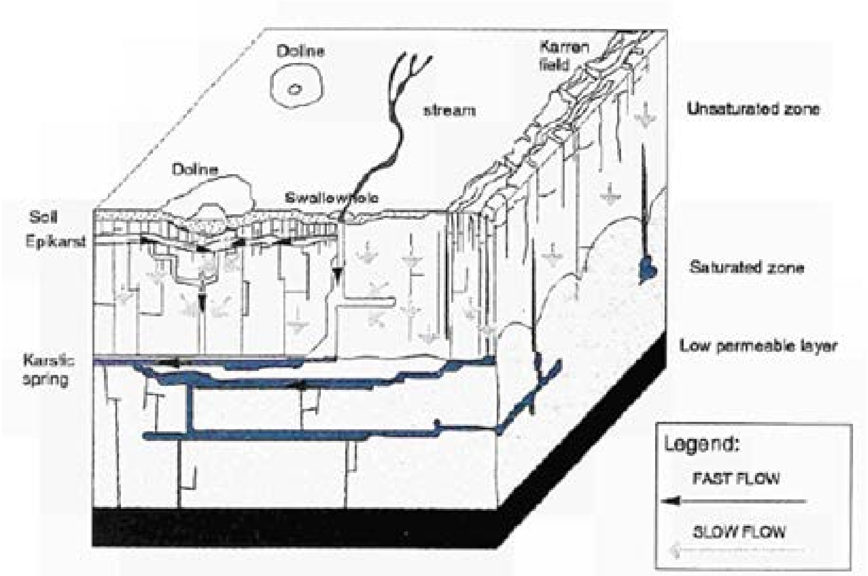


Figure 6 Conceptual model of a karstic aquifer. Source: European Commission, (1995)

Karstic aquifers have been defined as '*those that contain dissolution-generated conduits that permit rapid transport of groundwater, often in turbulent flow*' (Beck et al., 1999). For instance, karst term in the MAM case study describes the limestone nature of the aquifer where the water body is composes mainly of soluble rock that conduct water principally through porosity media formed by the dissolution of the rock. These are characterized by (European Commission, 1995):

- Absence of permanent surface flow, and presence of shallow holes and closed depressions
- Presence of caves and large underground passages
- Existence of large springs frequently located at the base of the carbonate sequence

Karstification is a process starting with the rainwater that becomes acid in contact with carbon dioxide in the atmosphere and the soil. Then it drains into the fractured rock of the aquifer creating a network of channels and fissures continuously eroding and enlarging these channels. As it drains, it transport larger amount of water together with pollutants present (Gunn, 2004).

Intrinsic properties used to describe karstic aquifer are: porosity, permeability and hydraulic conductivity. Porosity is expressed as a percentage of 'the ratio of the volume of voids in the rock mass to the total bulk volume'. Permeability depends on hydraulic conductivity which is expressed as the rate at which a volume of fluid can pass through a cross-sectional area of rock in units of length per unit of time (Beck et al., 1999; Waltham, 2013). Thus, such easy infiltration could lead to public health threats due to waterborne diseases as discussed below.

2.2. Engineering interventions

Engineering interventions are defined here as the actions taken in the water sector related to the improvement of water and wastewater treatment and management in order to improve the overall impact in population’s health. Mara, (2006) is one of the pioneers to develop engineering interventions related to water supply, sanitation and hygiene for the developing world. One clear example in WASH engineering interventions was the condominium sewerage, an off-site system of lower cost compared to conventional supply ($19/connection vs $88/connection), created to provide adequate water supply and sanitation to urban and high-density peri-urban areas (Mara, 2008; Mara, 2009). The main aim of those engineering interventions was to slow down the diarrhoea disease incidence (DDI), specially for the 0-4 years age-group, which is of major concern in developing countries (DDI= 2.4-5.2).

According to Harvey et al., (2002), in order to determine the most appropriate engineering interventions for wastewater management, the following factors must be considered:

- Ground conditions
- Groundwater level
- Topography
- Location and type of water sources
- Quantity and quality of wastewater generated
- Climate conditions
- Socio-cultural conditions

A report from the European Commission distinguishes a variety of engineering interventions based on the “typical response interventions” as illustrated in Table 5 (European Commission, 2005).

According to Table 5 most of the engineering interventions in the present research are classified as water and environmental sanitation interventions because they focus on the improvement or development of water and wastewater infrastructure to improve the management of the water supply system.

More specifically, engineering interventions for preventing diarrhoea have also been reviewed by Clasen et al., (2007) concluding that interventions at household level are more effective to prevent diarrhoea than interventions at water source level. Thus, the selection of interventions in this thesis would be considered at household level.

Table 5 Examples of engineering interventions in the water sector

Water	
• Development of new or expansion of existing water systems • Water resources assessments • Expansion and/or improvement of the water system	• Provision of items to host communities or water institutions • Water tankering • Redirection of supplies to drinking water
Environmental Sanitation	
• Sanitary assessment • Implementation and improve facilities at water points • Construct private toilet/latrine facilities • Implement other sanitation facilities	• Expansion of existing sanitary systems • Construction of temporary toilet and other sanitary facilities • Distribution of sanitary kits • Vector control

Source: Modified from European Commission,(2005)

2.3. Water and public health

Human health has been affected along the years by pollutants released in the environment. One of the most relevant interconnections between environmental pollutants and public health is in terms of water quality, a vital element for every society which has been changed over time due to traditional and modern anthropogenic activities. Therefore, establishing adequate water quality and public health monitoring data is fundamental to prevent and control present and future water-related diseases (Briggs, 2003; Fawell and Nieuwenhuijsen, 2003).

Water quality is interconnected with the public health status of the supplied area. For example, in the U.S. alone, more than 80 percent of antibiotics produced are used in livestock activities for pigs, cows, chicken, and turkeys. In terms of wastewater generated from livestock, the U.S. Food and Drug Administration (FDA) recognized misuse and overuse of antibiotics in a wide range of livestock activities. These issues have created a specter of untreatable infections due to superbugs, which are defined as antibiotic-resistant bacteria capable of infecting people. The amount of these antibiotics released in wastewater is difficult to quantify considering the massive livestock production not only in the U.S. but worldwide (Kar Avinash, 2011).

Historical evidences of water-related disease outbreaks have been documented worldwide. Bitton, (2010) classified 502 causes of disease outbreaks from 1971 to 1985 into four categories. Category 1 (49%): untreated or inadequately disinfected groundwater; Category 2 (24%): untreated or inadequate disinfected or filtered surface water; Category 3 (16%): due to distribution or storage deficiencies; and Category 4 (11%): represents miscellaneous. Even though, dirty water kills people, especially children not only in outbreaks but in everyday life.

2.3.1. Drinking water contaminants

Mexico drinking water regulations shown in Table 6 are based on the USEPA (U.S. Environmental Protection Agency) in line with the Maximum Contaminant Level (MCL) set by WHO.

USEPA has set primary and secondary drinking water regulations (NPDWR or primary standards and NSDWR or secondary standards). In general, primary standards are legally enforceable while secondary standards are not.

Nevertheless, in Mexico only those in Table 6 are officially regulated; of special concern is microbial control which is regulated by a single indicator: faecal coliforms (full list -Appendix A). Pollutants can be classified in two main groups: chemical and biological, which are discussed in the following section, along with emergent pollutants that comprise current pollutants of major concern worldwide.

Chemical pollutants. Some sources of chemical pollutants in water are: natural occurrence, and anthropogenic activities such as industrial, agricultural and livestock production. It is challenging to monitor the broad variety of pollutants discharged from all sources, thus some pollutants of major occurrence and significant health and/or environmental impacts have been selected as chemical indicators (Table 7).

Biological pollutants*.* Current WHO guidelines for drinking-water quality (fourth edition, 2011), classified 29 pathogens transmitted through drinking-water supply into four categories: 19 bacteria, 8 viruses, 11 protozoa, and 4 helminthes (Table 8).

Major pathogenic bacteria and protozoa transmitted through water are listed in Table 9 shows a list of pathogenic microbial, with transmission routes, symptoms and diseases.

The use of faecal coliform as microbial indicator for monitoring water quality has been widely promoted by national and international authorities. In particular *E. coli*, which accounts for up to 97% of total coliforms is used for different purposes (Dufour, 1997; Allen and Edberg, 1995). It assumes absence of similar organisms as the indicator used (WHO, 2011).

Table 6 Maximum Contaminant Levels (MCL) for drinking water

Pollutant	Type*	Unit	USEPA	EU	WHO	Mexico
Microorganisms						
Total Coliforms	1	%	≤5			0
Faecal coliform (*E. Coli*)	1					0
Turbidity	1	NTU	≤5			5
Disinfection Byproducts						
Total Trihalomethanes (TTHMS) [2]		µg/l	80	10	1	200
Disinfectants						
Chlorine as CL_2	1	µg/l	4000		5000	200-1500
Chloride	2	mg/l	250	250		250
Chemicals						
Aluminum	2	mg/l	0.05-.2	2		2
Ammonium		µg/l		500		500
Arsenic	1	µg/l	10	10	10	50
Barium	1	mg/l	2		0.7	0.7
Cadmium	1	µg/l	5	5	3	5
Chromium (total)	1	µg/l	100	50	50	50
Copper	2	mg/l	1	2	2	2
Cyanide (as free cyanide)	1	µg/l	200	50	70	70
Fluoride	2	mg/l	2	1.5	1.5	1.5
Iron	2	mg/l	0.3	0.2		0.3
Lead	1	µg/l	15	10	10	25
Manganese	2	mg/l	0.05	0.05	0.4	0.15
Mercury	1	µg/l	2	1	1	1
Nitrate (as Nitrogen)	1	mg/l	10	50[1]	50[1]	10
Nitrite (as Nitrogen)	1	mg/l	1	50	20	50
Silver	2	mg/l	0.1			
Sodium	1	µg/l		200,000		200,000
Sulfate	2	mg/l	250			
Zinc	2	mg/L	5			5
Aldrin and dieldrin[4]	1	µg/l			0.03	30
Chlordane[4]	1	µg/l	2		0.2	300
2,4-D (Dichlorophenoxy acetic acid)	1	µg/l	70	30		50,000
DDT (Dichlorodiphenyl trichloroethane)		µg/l		1		1000
Heptachlor epoxide	1	µg/l	0.2			300
Hexachlorobenzene	1	µg/l	1			10
Lindane[4]	1	µg/l	0.2		2	2000
Metoxichlorine		mg/l				20
Phenol		µg/l				1
Radionuclides						
Alpha particles	1	pCi/l	15			0.1[3]
Beta particles and photon	1	mRem/l	4			1[3]
Others						
Color	2	units	15			20
Hardness (CaCO3)		µg/l				500,000
Foaming Agents	2	mg/l	0.5			0.5
Odor	2	Unit	3			good
pH	2	Unit	6.5-8.5			6.5-8.5
Total Dissolved Solids	2	mg/l	500			1000

[1]as nitrate; [2]The sum of benzo(b)fluoranthene, benzo(k)fluranthene, benzo(g,h,l) perylene, and indenol(1,2,3-cd)pyrene; [3]unit in Bq/l; [4]pesticides. *Type refers to primary (1) or secondary (2) standard. Source: USEPA, (2009); WHO, (2011); NOM-127-SSA-1994; Sullivan et al., (2005).

Table 7 Chemical pollutant indicators for water quality monitoring

Pollutant	Origin	Human/environmental impact
Nitrate (NO_3)	Fertilizers, human and animal wastes	Wastewater from livestock, agriculture, domestic urban and rural (septic tanks, landfills), and industries
Phosphorus	Fertilizers, detergents	Eutrophication
Nitrogen	As nitrite and nitrate: runoff from fertilizer use; leaching from septic tanks; sewage; erosion of natural deposits	Infants <6 months who drink water with nitrate above MCL* (10 mg/L) could become seriously ill and, if untreated, may die. Symptoms include shortness of breath and blue-baby syndrome
Fluorides	Additive for strong teeth; erosion of natural deposits; discharge from fertilizer and aluminum factories	Bone disease (pain and tenderness of the bones); children may get mottled teeth
Pesticides	Runoff from agriculture	Cardiovascular system or reproductive system problems
Lead (Pb)	Corrosion of households plumbing systems; erosion of natural deposits	Children: delays in physical or mental development; deficits in attention; adults: kidney problems; high blood pressure
Cadmium (Cd)	Corrosion of galvanized pipes; natural deposits erosion; metal refineries discharge; batteries and paints runoff	Kidney damage
Copper (Cu)	Corrosion of household plumbing systems; erosion of natural deposits	Short-term: gastrointestinal distress. Long-term: liver or kidney damage
Chromium (Cr)	Discharge from steel and pulp mills; erosion of natural deposits	Allergic dermatitis
Nickel (Ni)	Production of stainless steel and nickel alloys. Naturally occurs in groundwater.	Nickel allergy. Inhaled nickel compounds are carcinogenic to human.
Mercury (Hg)	Erosion of natural deposits; refineries and factories discharges; runoff f	Kidney damage

Source: Adapted from USEPA, (2009); Avalos, (2009).

Table 8 Waterborne pathogens transmitted through water

Type of pathogen	Pathogen	Human risk	Pathogen	Human risk
Bacteria	*Acinetobacter*	High	*Leptospira*	High
	Aeromonas	High	*Mycobacteria*	Low
	Pseudomona aeruginosa	High	*Bacillus*	High
	Burkholderia pseudomallei	High	*Salmonella*	High
	Campylobacter	High	*Shigella*	High
	Enterobacter sakazakii	High	*Staphylococcus aureus*	High
	Escherichia coli – Pathogenic	High	*Tsukamurella*	High
	Helicobacter pylori	High	*Vibrio cholerae*	High
	Klebsiella	High	*Yersinia*	High
	Legionella	High		
Viruses	Adenoviruses	Moderate	Hepatitis A and B viruses	High
	Astroviruses	Moderate	Orthoreoviruses	High
	Caliciviruses	Moderate	Rotaviruses	High
	Enteroviruses	High	*Entamoeba histolytica*	High
Protozoa	*Acanthamoeba*	High	*Giardia intestinalis*	High
	Balantidium coli	High	*Isospora belli*	High
	Blastocystis	High	*Microsporidia*	High
	Cryptosporidium	High	*Naegleria fowleri*	High
	Cyclospora cayetanensis	High	*Toxoplasma gondii*	High
Helminths	*Dracunculus medinensis*	High	*Fasciola spp.*	High
	Free-living nematodes	High	*Schistosoma* spp.	High

Source: Adapted from WHO, (2011); Jimenez, (2008)

Table 9 List of pathogenic bacteria and protozoan, transmission route, and symptoms

Pathogen	Symptoms/transmission of infection	Illness
	Bacteria	
Escherichia coli	Diarrhoea, hemolytic uremic syndrome, vomiting.	Intestinal infections, gastroenteritis
Campylobacter jejuni	Diarrhoea, vomiting, fever, muscle pain, Guillain-Barre syndrome	Campylobacteriosis. Diarrhoea
Yersinia enterolitica	Fever, abdominal pain, dysentery	Yersinosis, diarrhoea
Salmonella typhi	Acute gastroenteritis with diarrhoea, abdominal cramps, fever, nausea, vomit, headaches and, in severe cases, death	Salmonellosis or typhoid fever
Shigella	Acute gastroenteritis with diarrhoea, abdominal pain, migraine and faeces with blood and mucous infectious	Bacillary shigellosis or dysentery; more virulent in old people and children
Mycobacterium tuberculosis	Cause diseases in people who swim in contaminated water	Gastrointestinal alterations
Vibrio cholerae	Usually affects children, causes liquid diarrhoea with hydro electrolytic losses and severe dehydration, and vomiting	Gastroenteritis by ingestion of polluted water or irrigation
Helicobacter pylori	Transmission not well-known; possible by unsanitary conditions and polluted food.	Gastritis, duodenal ulcer, carcinoma, diarrhoea
Legionella pneumophila	Acute respiratory illness	Legionellosis
Leptospira	Jaundice, fever (Well's disease)	Leptospirosis
	Protozoan	
Enterocytozoon bieneusi	Diarrhoea, malabsorption	*Enterocytozoon bieneusi* diarrhoea
Balantidium coli	Diarrhoea, dysentery	Balantidiasis
	Watery diarrhoea	Isospora belli diarrhoea
Entamoeba histolytica	Gas, abdominal pain, fever, invade the large intestine.	Amoebiasis, (amoebic dysentery); hepatic dysentery.
Naegleria fowleri	Fatal disease; inflammation of the brain	Amoebic meningoencephallitis
Cryptosporidium	Diarrhoea, stomach cramps, nausea, dehydration and headaches.	Cryptosporidiosis Infective dose is 1-10 cysts.
Giardia lamblia	Very liquid, odorous and explosive diarrhoea, and loss of appetite	Giardiasis, particularly affects undernourished children
Ascaris	Persistent cough, wheezing, nausea, vomit, abdominal pain, and diarrhoea	Ascariasis
Hymenolepis	Diarrhoea, itching, loss of appetite	Hymenolepiasis
	Viral	
Hepatitis A	Fever, nausea, diarrhoea, inflammation of liver, necrosis, sclera icterus	Hepatitis A; Hepatitis E
Enterovirus (67 types)	Heart anomalies, meningitis	Gastroenteritis
Poliovirus	Weakness, muscle pain, fatigue	Poliomyelitis
Rotavirus	Vomiting, diarrhoea, fever, dehydration	Diarrhoea, gastroenteritis
Adenovirus (31 to 51 types)	Stomach and intestines inflammation, watery diarrhoea, vomiting, and fever	Adenoviral diarrhoea
Reovirus	Vomiting, diarrhoea	Gastroenteritis

Source: Marsalek et al., (2006) based on – for bacteria: Craun, (1988); Sansonetti, (1991); Thomas et al., (1992); Hopkins et al., (1993), Lima and Lima, (1993); Nachamkin, (1993); Jawetz et al., (1996); Johnson et al., (1997); - for protozoan: Salas et al., (1990); Gray, (1994); Goldstein et al., (1996); Tellez et al., (1997); WHO, (1997); Cifuentes et al., (2002); USEPA, (2010); Jimenez, (2008)based on California Department of Health and Cooper, (1975); Asano et al., (1998); Kadleck & Knight, (1995).

Emergent pollutants. These are defined as a broad set of synthetic or naturally occurring chemicals or microorganisms of increasing concern, which are not commonly monitored and may be harmful to human health and the ecosystem (Survey, 2013). Examples of emergent pollutants are:

- Heavy metals that remain unchanged for years and thus may pose a threat to both human health and the ecosystem.
- Endocrine Disruptors (EDs). Since 2012 there is a list of 435 endocrine disrupter priority substances of adverse health effects and of environmental concern (Petersen G., 2007).
- Methyl tert-butyl ether (MTBE). The use of MTBE-oxygenated gasoline poses an imminent threat to public health because it dissolves easily in water, infiltrating faster in the ground than other gasoline components. Data support the conclusion that MTBE is a potential human carcinogen at doses higher than 20-40 ppb.
- Diethylhexyl-phthalate (DEPH). Mainly used as additive in plastics for flexibility purposes, therefore it has widespread use in industry, consumer and medical products. DEHP is well absorbed by the body when swallowed or breathed in. Some health effects in animals are kidney adverse effects, impacting male sexual development, and causing tumors.

2.3.2. Transmission of water-related diseases

Transmission routes for water-related diseases include gastro-oral, oral-oral, breastfeeding, and faecal-oral. Most water-related diseases such as diarrhoea is mainly transmitted through faecal-oral route shown in Figure 7, which has been further differentiated as the 5F's by (Mara et al., 2010) in Figure 8. Health impacts of these water related diseases can be measured through Quantitative Microbial Risks Assessment (QMRA), as discussed below.

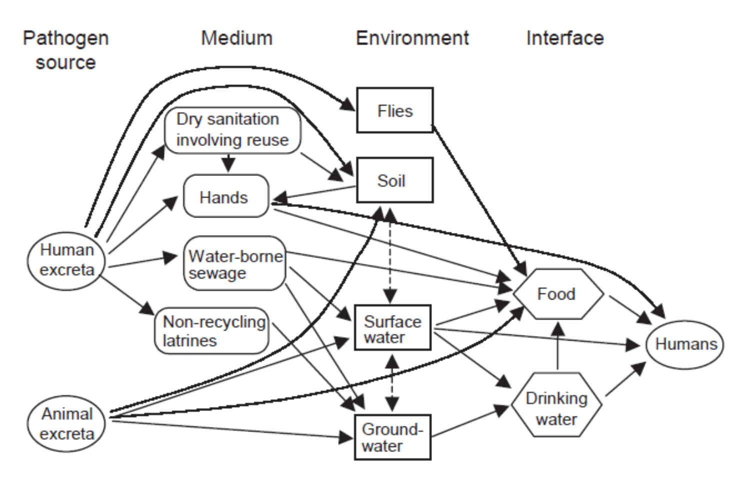


Figure 7 Faecal-oral transmission route. Source: Ezzati et al., (2004)

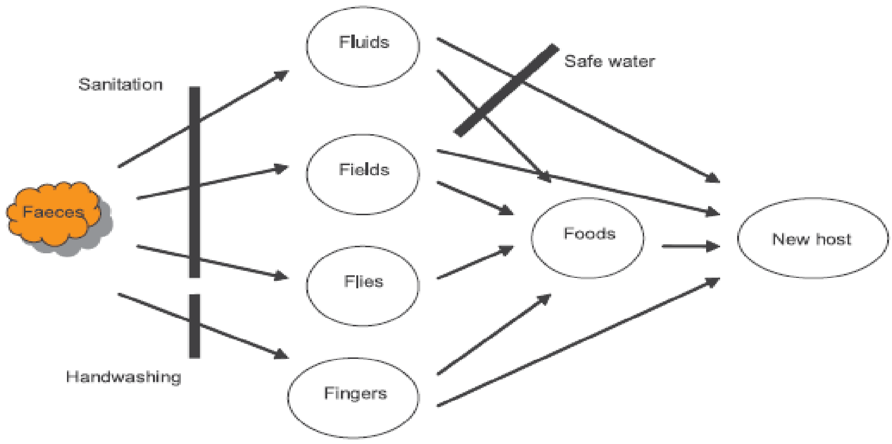


Figure 8 The 5F's Faecal-oral transmission pathways. Source: Mara et al., (2010)

2.3.3. QMRA

QMRA stands for "Quantitative Microbiological Risk Assessment". The first step to develop a QMRA is identifying the risk(s) through the collection of data. There are different data collection methods that could be implemented such as those used in previous studies, international guidelines, or field data collection. Commonly used parameters for risks identification are: access to water supply by service level, use of water sources, selection of water sources and households for testing quality and quantity data, sanitation condition, continuity, cost (affordability) and leakage. All these factors may be collected, to integrate all available data sources (official and non-official) and all the stakeholders involved (Howard et al., 2006).

According to the WHO guidelines for drinking water, it is important to consider a tolerable disease burden of ≤ 10^6 DALY (Disability Adjusted Life Year), which may be difficult to achieve for developing countries. There is a need of a better approach than a single value as a measure to evaluate risk acceptability. Besides, available dose-response database is also needed in order to implement Quantitative Microbial Risk Assessment (QMRA) at a case study level (Drechsel et al., 2010).

The 2006 WHO guidelines for wastewater reuse have established the method QMRA-MC (Quantitative Microbial Risk Analyses-Monte Carlo simulations) as a more rigorous method to estimate annual infection risks. This method represents the daily variation in infection risk to determine the annual risk by repeated calculation through n-Monte Carlo simulations. Monte Carlo Analysis incorporated into the QMRA (QMRA-MC) has the advantage to calculate and include the worst case scenario into the risk assessment and is a useful tool for decision-makers and city planners.

In general, Quantitative Microbiological Risk Assessment (QMRA) is divided into four main stages (WHO, 2011):

1. Hazard Identification

2. Exposure Assessment
3. Hazard characterization
4. Risk Characterization

In terms of chemical risks assessment, some models assess ecological risks from toxic chemicals in the environment (Bartell et al., 2003). Environmental Risk Assessment (ERA) is used to trace the environmental fate, human health effects, and ecological effects of different pollutants. ERA models aim to represent key ecological processes to quantify adverse effects of pollutants on organisms (Hunka et al., 2013; Schmolke et al., 2010a; Wang and Luttik, 2012; Forbes et al., 2011).

For example, risk assessment for contaminated groundwater has been performed in Denmark by Troldborg, (2010). Results allowed the identification of pollutant sources and reveal the importance of local rather than regional or national conditions.

2.4. Modelling contaminant concentrations in groundwater

This research focuses on nitrate as a persistent (conservative) chemical indicator pollutant, and faecal coliforms as non-persistent (non-conservative) microbial indicator pollutant – both of relevance to groundwater quality, modelled with the MAM case study data.

2.4.1. Nitrate

Nitrate is a major persistent contaminant in groundwater. For instance, in the United States this is the number one drinking water contaminant. Various models have been developed to describe and predict nitrate concentration in groundwater in relation to nitrogen input from agriculture and other anthropogenic activities in aquifer recharge zones. There is no general or widely applicable “nitrate model” available – modelling concept, input parameters and assumptions have to be adapted to local characteristics with respect to climatic conditions, aquifer hydrogeology, pollutant sources, population dynamics, etc.

The generation and fate of nitrate in aquifers is affected by a number of factors which bring uncertainties to its simulation. In the specific case of karstic aquifers, nitrate modelling is often based on simplifying approaches that appear to be reasonable approximations (Bear and Verruijit, 1987; Pinder et al., 1993):

- High vulnerability of karstic aquifers in combination with the shallow depth of the water table (such as in the MAM study area) justifies the assumption that any nitrogen applied to the surface may leach into the groundwater.

- Nitrogen input from agriculture, livestock and other human activities is primarily in the form of organic and ammonia nitrogen. Conversion to nitrate depends on the efficiency of bacterial nitrification and is not easy to predict. In highly vulnerable, oxygen rich karstic aquifers, organic N and ammonium N are effectively converted to nitrate. This is exemplified by nitrogen load modelling for the Barton Spring Zone, a karst aquifer in central Texas (Mahler et al., 2011). Over an observation period of 2.5 years, a balance between total nitrogen input by stream recharge and nitrogen load at discharge sites was found. However, the portion of organic ammonia N was high in recharge water and becomes rather low in deep water supply wells or at discharge sites, indicating extensive conversion to nitrate.

- Nitrate does not show significant attenuation in porous karst limestone with conduit flow and easily reaches deeper sections of an aquifer. In a south-western Georgia (USA) karst aquifer, no correlation between nitrate concentration and sampling well depth at 0-80 m was observed (Katz et al., 2014).

- Once formed in an oxygen rich karstic aquifer, nitrate is expected to persist for decades since bacterial denitrification is considered negligible under these conditions (Tesoriero et al., 2007). In line with this, nitrate levels may respond to interventions with a significant delay. Katz et al., (2014) have simulated for a south-western Georgia (USA) karst aquifer the scenario “no further N input from 2001” and found that flushing-out of nitrate to half of its concentration takes several decades. A similar delay time is predicted by a numeric simulation of nitrate concentration 2008-2060 for a highly permeable basalt aquifer in south-central Idaho, USA (Skinner and Rupert, 2012). Considering the above, a simplified approach has been applied to nitrate modelling in the MAM groundwater; further details are described in Chapter 4.

2.4.2. Faecal coliform

Microbial contamination from livestock and domestic wastewater constitutes the most serious water quality problem in many areas of the world. Faecal coliforms are an example of a non-persistent pollutant, which is widely used as a biological indicator of water quality as described in section 2.2.1. In the modelling of transport and concentration of faecal coliforms in groundwater, two major differences to nitrate have to be considered: larger size (about 1 µM vs < 1nm for nitrate) and die-off (half-life typically a few days vs years or decades for nitrate). These principally increases the complexity of modelling approaches together to the filtering effects, which depends on soil type and pore size, and due to the significant influence of environmental parameters on microbial lifetime (John and Rose, 2005). Thus, under the specific conditions of highly vulnerable karstic aquifers, various studies (including tracer tests) suggest relatively simple microbial transport behaviour (Pinder et al., 1993):

- Microbes can be transported through fractures in karstic limestone even faster than small molecules (such as nitrate) since they are excluded on the basis of their size from fine porosity outside major flow paths (Harvey, 1997).

- A diffuse recharge from rainfall into a 30m thick unsaturated limestone layer in the Jura Mountains, Switzerland, was simulated by irrigation with tracer spiked water. The tracers, either small fluorescent dye molecules or mud particles, took only 1-2 hours to pass 2 m of soil and 30 m of limestone; particles arrived with remarkably little retardation - even faster than the dye molecules (Goldschneider et al., 2008).

- A tracer study with 1 µM fluorescent microspheres (simulating bacteria) accompanied by transport modelling in a karst conduit system of the Austro-German alps revealed a transport over 2.5 km with about 40% recovery. Peak "arrival times" of the microspheres were 18 h and 83 h under high and low flow conditions, respectively (Göppert and Goldscheider, 2008).

- Field data from Yucatan, Mexico (Osorio, 2009; Torres, 2010) report significant faecal coliform levels in deep water supply wells, suggesting effective vertical transport from the surface to deeper zones of the karst aquifer.

From the above mentioned, the modelling approach of this thesis relies on various simplifying assumptions for transport and fate of faecal coliforms in the vulnerable karst aquifer of the MAM study area, as discussed in Chapter 4. The following section describes the modelling software used in the present research.

2.5. System Dynamics Modelling (SDM)

System dynamics concept was first developed by Prof Jay W. Forrester in 1956 through modelling industrial dynamics at Sloan School in the USA. It was meant to represent complex systems, providing feedback, accumulations, delays and non-linear dynamics and thus simulate a realistic scenario analysis on the basis of a suitable conceptual model. Applicability of SDM was rapidly extended from economics to engineering, environmental science, ecology and transport (Grobler and Strohhecker, 2012). SDM integrates multiple data sources and methods to analyze and design policies, from the easiest system with a few components to a complex social, economic, managerial, and ecological system (Wei et al., 2012).

In recent years, SDM has been applied to a broad set of research areas. In particular, environmental examples include: climate change prediction (Modelling et al., 2012), engineering services (Lai et al., 2001), environmental impacts (Deaton and Winebrake, 2000), interrelationships between environmental, ecological and economic resources (Constanza et al., 1998), garbage disposal (Cai, 2006), land resources (Chen et al., 1999), waste management (Ciplak, 2013); sustainable development (Xu et al., 2002), transport systems (Shepherd, 2014), and land use changes (Parsons et al., 2011).

SDM and sustainability are interwoven because of its suitability to provide a holistic and more comprehensive structural representation with interdisciplinary methodological formulations of the real scenario under study. For example, Wei et al., (2012) illustrates how SDM allows easy adaptation to changes within all elements of the water system that were tested through computer simulation.

The modelling process includes a series of steps that needs to be addressed in order to measure the usefulness of the model. On the one hand, the modelers may implement, test, and analyses the structural model. On the other hand decision makers and stakeholders may get involved in problem formulation, assessment of uncertainties and results, and formulation of recommendations (Grimm and Schomolke, 2011). An illustrative example is the process used to standardize Transparent and Comprehensive Ecological modelling (TRACE), where the following steps were taken (Schmolke et al., 2010b):

a. Model development: problem formulation, design and formulation of conceptual model represented, model description, parameterization, and calibration.
b. Model testing and analysis: verification, sensitivity analysis, and validation.
c. Model application: results and recommendations.

2.5.1. System dynamic software

Commercial computer-based simulation software with significant applications to the environmental sector are: Vensim (Ventana Systems Inc.), Simile, Simantics, STELLA, iThink (from isee Systems), PowerSim, and AnyLogic (Sterman, 2000; Ford, 2009).

2.5.2. Vensim software

Vensim was created in 1985 initially for business and technical support. There have been six versions of this software; likewise, there are six configurations of Vensim namely: Vensim PLE (free version), PLE Plus (used in the present research version 5.9e), DSS, Pro, Read, and Venapp Runtime. Differences among these configurations depend on the users purposes (Ventana Systems, 2012).

Vensim is an interactive software, which allows development, exploration, analysis and optimization of simulated systems (Eberlein and Peterson, 1992). The model can increase robustness and quality with dimensional consistency and reality check application within the interface. In addition, it can be coupled with advanced simulation technologies to enhance optimization of the system (i.e. causal tracing of structure and behavior, subscripting array, Monte Carlo sensitivity analysis, and GIS - Geographic Information System).

Multidisciplinary, multi-scale and multi-stakeholder approaches were envisaged for the development of the Sustainable Integrated Management Model designed in this research project. Therefore, the versatile interface development tool of Vensim for creating a management system was one of the selection criteria.

2.5.3. Using Vensim in the water sector

A water balance model is a system of equations designed to represent the hydrological cycle. It aims to improve understanding of the critical processes that influence the hydrological cycle. Depending on the objectives of the study and the data availability, modelling can have different levels of complexity. A simple model may be suitable for some purposes; in other cases more complex models may be required. It is important to recognize that increasing model complexity does not necessarily improve accuracy (Walker and Zhang 2001).

Examples of SDM applied to the water sector include: global modelling of water resources (Simonovic, 2002), carrying capacity of water resources (Sun et al., 2007) water users accountability (Wei et al., 2012), water and food security (Khan et al., 2005), water pricing (Sahin et al., 2014); water resource planning (Zhang et al., 2008), water use issues (Fedorovskiy et al., 2004), and reservoir operations (Ahmad and Simonovic, 2000). SDM of the interplay between environmental factors, socio-economic impacts of anthropogenic activities and engineering interventions helps water practitioners to gain a more comprehensive understanding of the scenario (Blanco-Gutiérrez et al., 2011).

Examples of Vensim application to the water sector are limited to water demand management (Jefferies and Duffy, 2011; Bueno et al., 2006); water-quality modelling for pollutant abatement (Mirchi, 2013), and water scarcity assessment (Sušnik et al., 2012). Water Demand Management (WDW) is a Vensim-based model to move toward demand-driven urban scenarios (Jefferies and Duffy, 2011). It is a generic decision support tool to help decision-makers to compare cost-effectiveness options. Another example of a generic demand management model using Vensim was developed by Bueno et al., (2006) as a simple decision support tool for urban water engineers, planners and managers, which compares cost benefits over long time periods to support strategic decisions.

2.6. Research problem

Yucatan, Mexico, presents over 115,000 km^2 of tertiary and quaternary carbonates, characterized by shallow doline (a hollow or basin in a karstic region, typically funnel-shaped), low hills, and flooded cave systems (Gunn, 2004; Reddell, 1977). These makes the MAM case study located at the northwest of Yucatan, Mexico a very vulnerable karstic aquifer to contamination. Of particular concern are the waterborne diseases in this area, as this aquifer is the sole water supply for every activity in the growing urbanized area (Drew and Hotzl, 1999; BGS et al., 1995; Marin et al., 2003; Reddell, 1977). Therefore, it is of major importance to have an adequate strategy for a sustainable water management, which if possible, avoids rapid infiltration of contaminants from the surface to the aquifer, but also mitigate pollutants infiltrated in the aquifer through adequate wastewater management. Evidences of waterborne diseases are limited; nevertheless the “diarrhea season” has been historically documented by Dohering and Buttler, (1974); Lutz et al., (2000); Marin and Perry, (1994) which is expected as a consequence of the wash out effect of the aquifer.

2.7. Research gap

The approach taken in this research to improve public health due to waterborne diseases is integrating water and wastewater management practices. As the majority of wastewater in the case study is directly discharged into the groundwater with little or no treatment (treatment is less than 15% in the case study area), engineering interventions suggested in this research are needed in order to control and mitigate pollutants present in these wastewater.

In terms of the water human health nexus, it is important to establish a relation between pollutants concentration in the aquifer and local epidemiologic statistics. Thus, this research introduces the implementation of Quantitative Microbial Risk Assessment due to direct water consumption from the aquifer, which could evidence risks posed by the presence of microbial contaminants in the aquifer, as documented by Marin et al., (2000). Moreover, as a decision making tool, this research quantifies the economic benefit obtained by tackle these waterborne diseases (i.e. diarrhea), and the treatment cost-saved by reducing chemical pollutants (i.e. NO_3) in the aquifer. Overall, this case study is used as an example to illustrate the need to adapt policies and protocols in a broader context to provide a safe quality level of water supply through an adequate wastewater treatment system.

Chapter 3. Case study

"Balancing anthropogenic demands for water against the water needs of aquatic ecosystems is a pressing global issue". (Petts et al., 2006).

This chapter contains the case study data collected during the last three years of research and 2 fieldworks and aims to summarize the most relevant aspects of the Metropolitan Area of Merida (MAM). MAM is located in Yucatan, southeast Mexico, one of the 26 Latin American and Caribbean (LAC) countries. Two thirds of the LAC region is classified as arid or semi-arid, thus a quarter of its total population (over100 million people) lives in water stress areas (UNESCO, 2005). In the LAC countries, 68 million people live without access to improved water supply, and in terms of sanitation coverage, a total of 116 million people have no access to improved sanitation, (UNESCO, 2005). Mexico faces specific water issues; one of the most relevant is groundwater pollution due to improper use and disposal of heavy metals, synthetic chemicals and hazardous wastes. In addition, aquifer depletion and saline intrusion are increasing. In Mexico in 2000, 94% of urban population had access to improved water supply, but only 63% in rural area had this service (WHO/UNICEF, 2000).

Interest for the MAM case study is raised by the intrinsic current and future threats of groundwater contamination in the area, as a consequence of the rapid urban development and climate change. Water threats pose serious risks to human health due to water's quality and quantity impact in health, which was discussed on previous chapters. In this chapter, important characteristics of the MAM are described. These are: location, hydrogeology, water management, water supply and wastewater infrastructures, and water threats both natural and anthropogenic.

3.1. Location of the MAM

The Metropolitan Area of Merida (MAM) is situated in the northwest of Yucatan State, in the Yucatan Peninsula, southeast Mexico (Figure 9). Merida is the largest urban area of the Yucatan Peninsula (Drew and Hotzl, 1999). Yucatan Peninsula is of hydrological importance because it has a mean recharge of 25,316 hm^3, which is the 32% of the total mean recharge in Mexico (79,652 hm^3). This makes Yucatan Peninsula one of the most important reservoirs of groundwater in Mexico (Yucatan Government, 2013; Anton, 1993; CONAGUA, 2010a).

The MAM was created in 2008 by Merida City Council comprising 5 municipalities: Kanasin; Conkal; Ucú; Uman; and Merida (Figure 9). The municipality of Progreso was included two years later due to its proximity (~ 25km from Merida), and because of its importance as a major maritime port for economic and touristic development in Yucatan (Herrera-Silveira et al., 2004; OECD, 2008).

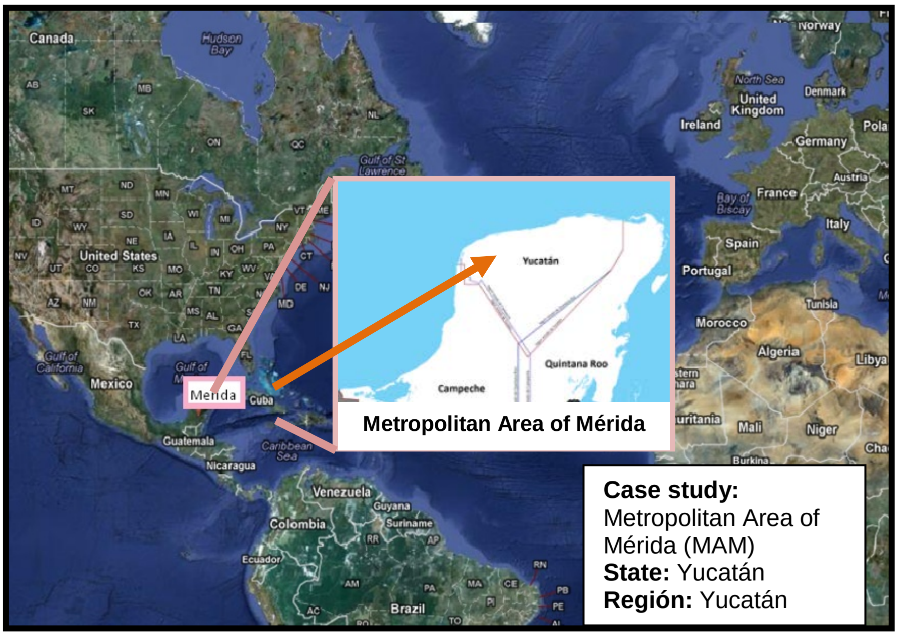


Figure 9 Case study location: Metropolitan Area of Merida (MAM), Yucatan, Mexico

For purposes of the present research, the MAM area includes the 6 municipalities of the MAM from Figure 10 along with 25 peripheral municipalities located between the Ring of Sinkholes (RS), as the south border, up to the coast of Progreso, as north border, which is further described in Chapter 4.

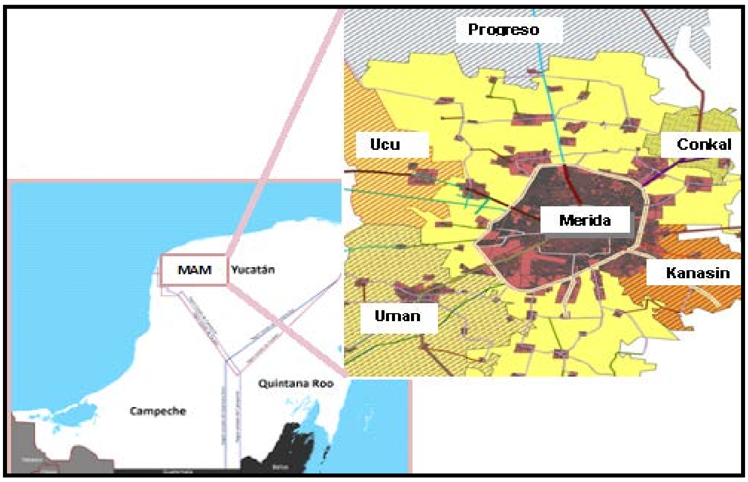


Figure 10 Municipalities of the Metropolitan Area of Mérida (MAM), Yucatan, México.

3.2. Hydrogeology

The Yucatan Peninsula is underlain by a highly permeable unconfined karstic aquifer of 48 meter depth in average at the MAM area, increasing to 80m at south and decreasing up to 0.5m depth at the north coast (González-Herrera et al., 2002). According to Morris et al., (1994), aquifers can be classified into two groups: consolidated and unconsolidated (deposits). Consolidated includes limestone, sandstone and some volcanic terrains. When fractured it is highly permeable. Some of the world's cities examples relying on this type of aquifer are Cebu City (Philippines), Jaffna (Sri Lanka), and Tai Yua (China). Examples of fractured volcanic aquifers are San Juan (Costa Rica), Guatemala City (Guatemala), and San Salvador (El Salvador). Unconsolidated aquifers are characterised by high porosity and great thickness. Some of the world's largest cities that rely on this type of aquifer are Mexico City, Beijing and Jakarta (See Appendix B of karstic aquifer worldwide). The MAM karstic aquifer is an unconsolidated aquifer of limestone and dolomites deposits (Teixeira, 2004; Sanchez y Pinto, 1999; RAMSAR, 2009; Yucatan Government, 2013).

The Ring of Sinkholes (RS) is a semi-circular, around 10 km broad and 240 km long band of natural waterholes (sinkholes) that spans from Celestun in the northwest cost to Dzilam at the northeast coast of the Yucatan Peninsula (Figure 10). Several researches describe the RS as a hydraulic barrier and highly conductive channel through which groundwater flows from the centre of Yucatan Peninsula to the sea (Perry et al., 1995; Holliday, 2007; CONABIO, 2010; SEDUMA, 2012). This might be questionable depending of local or regional levels of analysis. For instance González-Herrera et al., (2002) suggested that the RS has a hydraulic conductivity only slightly higher than the surrounding rock on the basis of groundwater flow modelling.

Geologically, the Yucatan aquifer was formed by a meteor collision ~65 Million years ago, which resulted on the development of the RS, where sinkholes are locally called "cenotes" from the Maya language "dzonot", meaning a cave with water (Lugo-Hubp et al., 1992; BGS et al., 1995; Urrutia-Fucugauchi et al., 2011; Marin et al., 2003; Gonzalez et al., 2007). Since 2009 the RS was declared as a wetland site of international importance due to its geological nature (Anton, 1993; RAMSAR, 2010).

The Yucatan aquifer is a mature karstic formation and it constitutes the only source of water supply for the MAM. Due to the karstic complexity with dissolution conduits, fractures and underground channels, water quality is difficult to monitor, thus a relatively simple model for "effective" groundwater flow has been suggested (González-Herrera et al., 2002), in which the karst aquifer was treated as homogeneous, granular-porous medium of very high hydraulic conductivity. There is limited evidence for the high permeability assigned to the "Sierrita de Ticul" (Figure 11), the highest area of Yucatan, which also has been modelled as a flow barrier (Perry et al., 2002; González-Herrera et al., 2002; Marin, 1990). Others more refined modelling approaches have tried to address regional-scale complexity of groundwater flow patterns, as documented in Bauer-Gottwein et al., (2011), but there is still not a consensus for modelling this system.

The Yucatan water table is less than 2m ±0.5masl (metres above mean sea level), with a freshwater lens of 1m thick at the north coast increasing to the south up to 80m

(Marin, 1990). On the basis of regional hydrological and geological characteristics, the national water authority CONAGUA (for its Spanish acronyms) has divided the Yucatan Peninsula in three regions (Figure 11): the coastal zone, the belt of cenotes, which includes the RS and the Sierrita de Ticul, and the inner plateau where the MAM is located (Anton, 1993; Sanchez y Pinto, 1999).

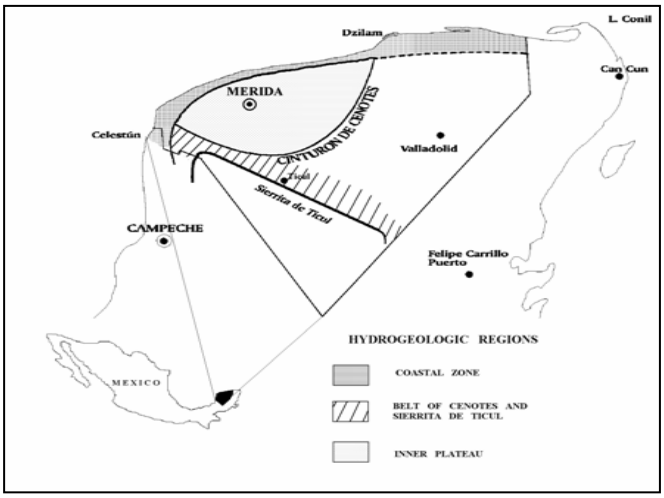


Figure 11 Hydrogeological regions of Yucatan aquifer. Source: Sanchez y Pinto, (1999)

Karstic aquifers can be defined as highly soluble freshwater bodies that conduct water principally via porosity by the dissolution of the rock. In the MAM area this dissolution phenomenon together with the high water table results in a high vulnerable aquifer (Holliday, 2007; Schmitter-Soto et al., 2002).

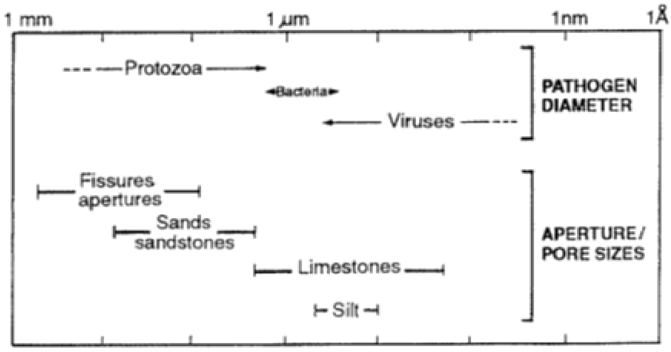


Figure 12 Diameter of microorganisms related to aquifers pore size. Source: Morris et al., (1994)

Even though natural attenuation, an intrinsic characteristic of the aquifers that often control contaminant loads through adsorption, dilution and filtration, in the MAM it has been reported to be limited. For instance, Figure 12 shows the nature of the unconsolidated limestone strata with fissures and apertures of larger pore opening than the size of the microbial pollutants (Morris et al., 1994).

3.3. Water management

Despite of the fact that Merida is the largest city in the southeast of Mexico, its urban development plan created every four years by Merida City Council has failed in terms of water management. The unsuccessful management may be due to political changes every three years at municipal level, every four years at Yucatan State level as well as every six years at National level, cutting off the creation of robust and uninterrupted action plans to protect and safeguard the natural water resources (Teixeira, 2004).

Febles and Hoogesteijin, (2008) analysed the water management regulations at the three government levels in Mexico: municipal, state and federal, in order to identify the cause of failure in terms of water management in Yucatan. After identifying many discrepancies in government jurisdiction and lack of congruence among the three authority levels, it was suggested the creation of a State law for drinking water and wastewater treatment, and the development of norms for a) pluvial water infiltration in green areas; b) design, construction and operation of drainage systems and treatment plants; c) infiltration of septic tanks discharges, and d) promotion of infiltration fields for septic tanks and treatment plants effluents.

Septic tanks (ST) are of particular concern in the MAM area. ST are designed with 2/3 of the tank volume set aside for sludge and scum accumulation and 24hr hydraulic detention time in the remaining 1/3 volume, which under effective operation leads to a total tank volume equal to 3 times the daily flow volume (Quintal, 1992; Gill et al., 2004). Nevertheless current status of ST in Yucatan do not comply with these specifications (Quintal, 1993).

One recent attempt to improve water management policies with a hydrological and political administrative approach has been the creation of a new decree by Yucatan Government in 2013. The aim of this new decree is to declare the Ring of Sinkholes as a protected reservoir area. This decree provided valuable data for this research in terms of water abstraction (Table 12) and wastewater discharge volumes (Table 13) within the study area (Yucatan Government, 2013).

According to the new 2013 decree, the area between the RS (as southern boundary) and the coastline area (as the northern boundary), is now a priority area to implement sustainable water management policies for the MAM (Figure 13). This priority area has been divided into four sub-zones A, B, C, D, and each subzone is again divided in: recharge, transition, and discharge zones according to its geographical location (Table 9) (Yucatan Government, 2013). A total of 53 municipalities are distributed as follows:

- *Sub-zone A: MAM and its periphery*. It comprises 17 municipalities: 7 are recharge zone (A1), 8 are transition zone (A2) and 2 are discharge zone (A3).
- *Sub-zone B: influence zone of the MAM*. It comprises 16 municipalities: 6 are recharge zone, 7 are transition zone and 3 are discharge zone.
- *Sub-zone C: eastern zone of the sinkholes ring influence*. It comprises by 14 municipalities: 11 are transition zone, and 3 are discharge zone.

- *Sub-zone D: western zone of the sinkholes ring influence*. It comprises 6 municipalities: 5 are transition zone, and 1 is discharge zone.

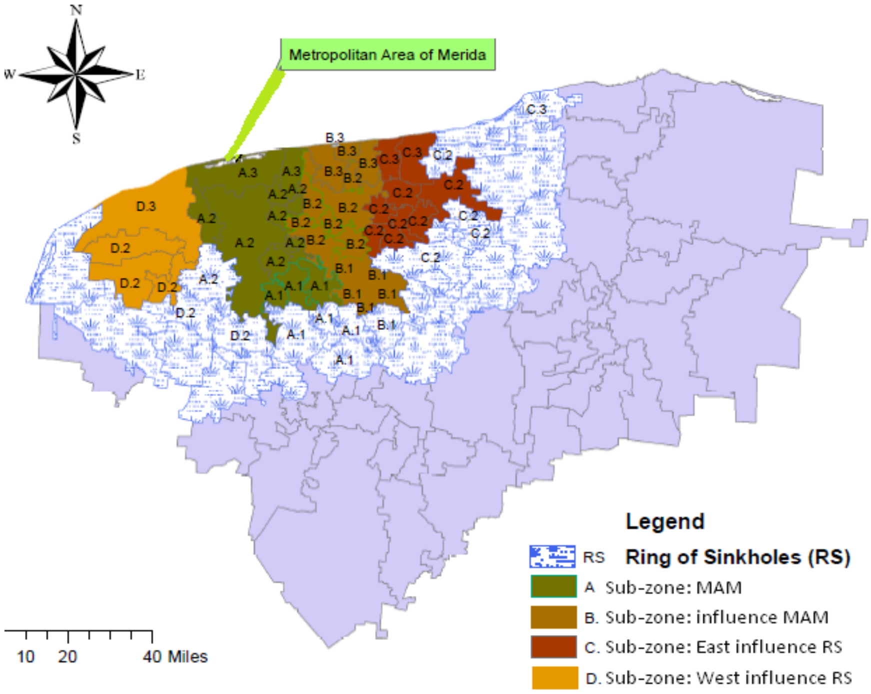


Figure 13 Priority area by Yucatan Government, (2013)

Twenty seven municipalities are comprised as RS municipalities (Figure 13), which are major recharge zone on this new decree (13 recharge municipalities from sub-zones A and B shown on Table 9). As an ecosystem, sinkholes are particularly vulnerable to drastic environmental changes (i.e. wastewater discharge from the MAM), thus these RS municipalities are of priority importance for this thesis (RAMSAR, 2010).

Additionally, the technical committee for water management of the MAM has reported that sub-zone A is of major concern for both water supply and wastewater disposal, because it encompasses 57% of total Yucatan population and 41% of total water abstraction (CCPY, 2012).

Due to the above, this new priority area was the basis to establish the study area of this research. The Sustainable Integrated Water Management Model (SIWMM) uses the data of the new ordinance, to serve as a strategy tool for the current and future sustainable water management of the MAM. Summarising, the selected area for the implementation of the SIWMM includes a total of 31 municipalities, 25 of which are within the priority area (Figure 13). The selection was based on two criteria, the geographical location of the MAM (municipalities highlighted on Table 10), and the regional groundwater flow direction, in order to assume a continuous water flow from one aquifer section to another (see Chapter 4 for further information).

Table 10 Priority area from government for water management in Yucatan

Sub-zone	Sub-zone A	Sub-zone B	Sub-zone C	Sub-zone D
Recharge	**A.1 Seye** **A.1 Acanceh** **A.1 Timucuy** **A.1 Homun** **A.1 Cuzama** **A.1 Tecoh** **A.1 Tekit**	**B.1 Tahmek** **B.1 Hoctun** **B.1 Xocchel** **B.1 Hocaba** **B.1 Sanahcat** **B.1 Huhi**		
Transition	**A.2Chicxulub Pueblo** **A.2 Mococha** **A.2 Merida** **A.2 Ucu** **A.2 Conkal** **A.2 Tixpehual** **A.2 Kanasin** **A.2 Uman**	B.2 Motul B.2 Telchac Pueblo B.2 Baca B.2 Muxupip **B.2 Yaxkukul** **B.2 Tixkokob** B.2 Cacalchen	C.2 Dzilam Gonzalez C.2 Temax C.2 Cansahcab C.2 Dzoncauich C.2 Suma C.2 Tepakan C.2 Teya C.2 Tekal de Venegas C.2 Tekanto C.2 Bokoba C.2 Izamal	D.2 Tetiz D.2 Samahil D.2 Kinchil D.2 Chochola D.2 Abala
Discharge	A.3 Ixil **A.3 Progreso**	B.3 Sinanche B.3 Telchac Pueblo B.3 Dzemul	C.3 Dzilam de Bravo C.3 Dzidzantun C.3 Yobain	**D.3 Hunucma**

Note: Municipalities highlighted are included in the MAM study area of the present research

3.4. Water supply infrastructure

In Mexico, each municipality is responsible to provide adequate water supply to its population, based on the Maximum Contaminant Level (MCL) standard, established on the national norm NOM-127-SSA1-1994. Since 1987, the water authority established a standard simplified water treatment process (potabilization system), for a maximum capacity of 250l/s to supply a population up to 75,000 inhabitants. Then in 2000 it was updated to a four-step treatment process due to new emergent pollutant. This comprises: flocculation (with $Al_2(SO_4)_3$ or polyelectrolytes), sedimentation, filtration, and disinfection (with chlorine gas), which is currently in use as generic treatment, adaptable to local pollutants of concern (CONAGUA, 2007).

Water supply coverage in Yucatan has been increasing from 74.8% of Yucatan population with access to water supply in 1990 up to 96.2% in 2009. The remaining people rely on private supply such as water wells in the backyards of households that have commonly been used until the 1960's before the authorities started the operation of a water supply system in Yucatan State (JAPAY, 2009). Water supply system provides chlorination as the only disinfection method. Nonetheless, according to national records, consumption of chlorine gas for water disinfection in Yucatan has decreased from 9220 L in 2000, up to 6290 L in 2009 (CONAGUA, 2010a). This may be because the majority of Yucatan water abstraction is used for agriculture activity, where water quality is not strictly regulated in contrast to drinking water.

Four main water well fields, which only use chlorination as treatment process, to supply an average of $4m^3/s$ of "tap water" to the MAM at 40m depth (Cuevas et al., 2003). These are: Merida I, Merida II, and Merida III and currently under construction Merida IV (Table 11). From these, Merida I is of major supply, and is the nearest to the recharge zone around the RS. Water quality of the RS has been reported as favourable for human consumption. Thus, proximity of water well fields to the RS guarantees good water quality (Pacheco et al., 2004b).

Table 11 Water supply fields of the Metropolitan Area of Merida

Well field	Operation start year	Supply area	Area (m^2)	Number of wells	Capacity (L/s)	(%) Population supplied
Merida I	1966	SE- Merida	6,250,000	24	1200	40
Merida II	1985	SW- Merida	720,000	10	500	14.46
Merida III	1993	North Merida	316,000	14 of 17	700	4.29
Merida IV[a]	2014	NW-Merida	413,610	26	1300	25
Interurban wells[b]		Around Merida		33	1650*	13.45
Progreso		Progreso		23	1150*	2.8

*Value estimated assuming 50 L/s for each well SE: Southeast; SW: Southwest. [a]Under construction, it is planned to supply 350,000 people; [b] data from SEDESOL. Source: adapted from Flores-Abuxapqui et al., (1995); JAPAY, (2013); CONAGUA, (2011). Chlorination in Merida IV has been reported to use 1680kg of chloride per month JAPAY, (2009)

Nevertheless, Flores-Abuxapqui et al., (1995) have reported different levels of contamination from these water well fields. Surprisingly, the highest contamination was within the peripheral area of Merida I (centre and south of Merida), which was associated to wastewater and faecal pollution. High concentration of mesophilic anaerobes in the influencing area of Merida III (northeast and east of Merida), was associated to the presence of wastewater treatment plant leakage or contamination within the distribution network. The best water quality was reported in the peripheral area of Merida II plant (west and north of Merida) for both: received water (at the household tap) and inter-households (at the distribution network).

In 2013 the decree set a total water abstraction for the priority area of 495 million m^3 (Table 12), which represent the 41% and 19% of the total abstraction in Yucatan State and Yucatan Peninsula respectively (CCPY, 2012).

Table 12 Water abstraction in the priority area of Yucatan in 2013 (m^3/year)

Sub-zone	Agriculture	Domestic	Aquaculture	Services	Industry	Livestock	Public Urban	Multiple	Total
A	3.7E+7	1.9E+4	0.0	7.2E+6	2.4E+7	2.0E+6	1.7E+8	5.9E+7	3.0E+8
B	1.7E+7	1.0E+3	3.6E+4	9.9E+4	1.3E+6	7.7E+5	8.8E+6	4.2E+7	7.1E+7
C	2.5E+7	0.0*	0.0	5.4E+4	6.4E+4	8.6E+5	6.6E+6	4.4E+7	7.7E+7
D	2.4E+7	0.0*	7.3E+4	1.1E+5	1.8E+4	3.2E+6	3.5E+6	8.0E+6	3.8E+7
A-D	1.0E+8	2.0E+4	1.1E+5	7.4E+6	2.5E+7	6.8E+6	1.9E+8	1.5E+8	4.9E+8
Rest of Yucatan	2.9E+8	5.0E+3	4.0E+3	3.8E+5	1.1E+7	9.1E+6	4.8E+7	3.2E+8	6.8E+8
Total	3.9E+8	2.5E+4	1.1E+5	7.8E+6	3.7E+7	1.6E+7	2.4E+8	4.8E+8	1.1E+9

*There was no data reported in the Yucatan ordinance, it is incomplete. Source: Yucatan Government, (2013).

Main users are agriculture and multiple uses with 257 million m^3 per year, public urban sector with 25.6 million m^3 per year, services sector with 7.4 million m^3 per year, and the livestock activity with 6.8 million m^3 per year (Yucatan Government, 2013).

Even though Table 12 shows the most recent statistic data of water abstraction in Yucatan, its reliability is questionable, since according to JAPAY (water authority for the Metropolitan Area of Merida) more than 40% of total extracted water is lost within the distribution system (JAPAY, 2009). Thus, volume of abstracted water documented might be overestimated for the MAM area. Additionally, unaccounted private water wells used for water supply all over the Yucatan State, particularly in rural areas, are not regulated by water authorities (Foster et al., 2001). Furthermore, private wells are not counted in the water abstraction statistics, which would result in an underestimation of the real abstracted volumes (Hernández-Terrones et al., 2010).

Similarly, wastewater discharge volume and pollutant loads reported in 2013, can be underestimated due to illegal discharges from industry and domestic users with water wells in the households backyard, inadequate septic tanks operation and maintenance (i.e. lower residential time than required), sewage leakage from septic tanks cracks, agriculture activity with unregistered fertilizers volumes, livestock with lack of wastewater treatments, and many small industries with inadequate disposal of wastewater (Osorio, 2009; Yucatan Government, 2013).

3.5. Wastewater infrastructure

Wastewater discharges to Yucatan aquifer have increased by 50% over the past 10 years. Even though sewerage in Yucatan has increased from 42.1% in 1990, up to 67.6% in 2009, only 2.4% of the wastewater in Yucatan is treated. It is important to notice that most of this wastewater is treated by wastewater treatment plants (WWTP) which are only located in new residential areas within the MAM area (CONAGUA, 2008; CONAGUA, 2010b; CONABIO, 2010).

In terms of domestic wastewater infrastructure, the majority of wastewater in Yucatan is collected in more than 200, 000 septic tanks, placed only 1 to 3 m above the water table. A removal of faecal bacteria by one log was reported for septic tanks (Morris et al., 2003). All septic tanks discharge direct or indirectly (through absorption wells) their effluents to the aquifer (Castillo et al., 2011). The majority of farmers and slaughterhouses do not apply any treatment at all and most of their wastewater effluents are directly discharged to the aquifer.

In 2010, a total of twenty-five WWTP were reported in the MAM that was treating only 2.4% of the total wastewater generated in Yucatan (Basulto-Solis, 2010; CONAGUA, 2010b). In 2011, it increased to 29 WWTP, even though Yucatan is still the third

Mexican State with the lowest sewerage coverage (CONAGUA, 2011). After treatment, effluents from WWTP are discharged via injection wells which are drilled down to saline interface. The residual sludge is spread on permeable areas, allowing the leachate seepage. Majority of these WWTP were built in the last two decades, as part of the new regulations for residential areas in Merida City as an attempt to reduce groundwater contamination (Yucatan Government, 1994). The rest of the MAM does not have any WWTP except one in Uman and one in Progreso (CONAGUA, 2009d).

Table 13 Wastewater discharge in the priority area of Yucatan (m^3/year)

Sub-zone	Aquaculture	Domestic	Services	Industry	Livestock	Total
A	7.28E+03	1.52E+04	3.93E+06	1.18E+07	1.44E+06	1.71E+07
B	0.00E+00	8.00E+02	1.33E+05	9.02E+05	7.50E+05	1.78E+06
C	7.15E+03	0.00E+00*	4.84E+04	3.84E+05	1.69E+05	6.08E+05
D	2.87E+04	0.00E+00*	6.24E+04	7.01E+04	6.27E+05	7.88E+05
A-D	4.31E+04	1.60E+04	4.17E+06	1.31E+07	2.99E+06	2.03E+07
Rest of Yucatan	0.00E+00	4.00E+03	1.78E+06	3.34E+06	1.69E+06	6.81E+06
Total	4.31E+04	2.00E+04	5.95E+06	1.65E+07	4.68E+06	2.71E+07

Note: there was not data reported for agriculture, domestic, public urban and multiple activities. Data reported *Data reported as in the Yucatan ordinance, it is incomplete. Source: Yucatan Government, (2013).

Priority area discharge a total of 128 million m^3 of wastewater per year, this is 70% of total wastewater of Yucatan (Table 13). From this 20 million m^3 per year were used for not municipal uses and 108 million m^3 per year for municipal uses, distributed as follows: 13.1 million m^3 from industry, 4.1 million m^3 from services sector, 2.9 million m^3 from livestock (porcine only), and 0.43 million m^3 from aquaculture (Yucatan Government, 2013). However, no data was reported for major water consumers such as domestic and agriculture sector. In case of agriculture, it is difficult to estimate due to its dependence on many factors such as effective water utilisation by the crops, water leakage and evaporated water from the soil. For this research, assumptions were made based on the water demand per each type of crop, and the fraction of water that is leaking into the soil as wastewater.

3.6. Water threats

In general, Yucatan Peninsula is considered as highly vulnerable to groundwater pollution based on the DRASTIC index, in particular for Yucatan State (Gijon, 2007). This index is a ranking methodology documented in 1987 by the U.S. Environmental Protection Agency and evaluates the most important hydrogeological factors that control the groundwater movement, and hence the pollution potential. These factors are the acronym of DRASTIC that stands for: Depth to water, (net) Recharge, Aquifer media, Soil media, Topography, Impact on the vadose zone (unsaturated zone), and hydraulic Conductivity of the aquifer. From the intrinsic vulnerability map obtained by

DRASTIC, and a map of land use, a contamination risk map was created by Gijon, (2007) to identify high risk areas, categorised by low, medium, high and extreme risk to contamination (Figure 14).

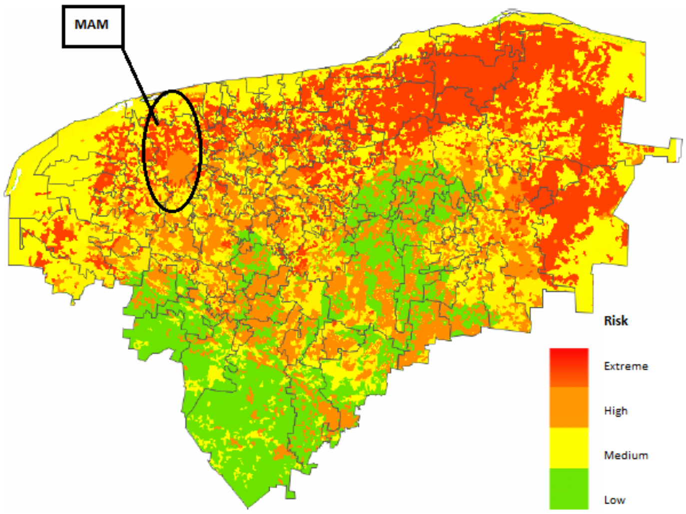


Figure 14 Groundwater contamination risk map of Yucatan. Source: Gijon, (2007)

Yucatan water quality degradation in the MAM is due to the following reasons: massive demographic growth (1.74 % of average annual population growth rate); saline intrusion; deterioration of wetlands, sinkholes and coastal areas; inadequate economic and environmental policies; inadequate water metering system and solid waste and wastewater disposals; flooding risk due to seasonal hurricanes; lack of regulation for water contamination control; lack of sustainable urban planning and management; and inadequate authority distribution among local, state and federal governments (Tello . and Alonzo, 2003; CONAGUA, 2009c). Of special concern are the frequency of hurricanes episodes occurring in the last decades, which can increase threatening to the MAM aquifer as documented by (Pacheco and Cabrera, 2013) after hurricane Isidore in 2002. Table 14 shows historic records of hurricanes that have affected Yucatan Peninsula.

Table 14 Historic record of hurricanes in Yucatan Peninsula

Name	Category*	Year	Wind speed (km/h)	Peninsula States affected
Gilbert	H5(H4)	1988	287	Quintana Roo, Yucatan
Diana	TT(H2)	1990	110	Yucatan, Campeche
Roxanne	H3(DT)	1995	185	Quintana Roo, Yucatan, Campeche
Dolly	TT(H1)	1996	110	Quintana Roo, Yucatan, Campeche
Isidore	H3	2002	205	Quintana Roo, Yucatan, Campeche
Emily	H3(H1)	2005	215	Yucatan, Quintana Roo
Wilma	H4(H3)	2005	241	Yucatan, Quintana Roo

Source: Alcica Construction, (2009); *Saffir/Simpson scale

The MAM case study has current and future anthropogenic threats (i.e. population growth, water demand and wastewater discharges) and natural threats (i.e. saline intrusion, hurricanes) to the water resources that have been historically documented since the Maya civilization (Back, 1995), and are topic of current and future concern due to metropolitan growth (Alcocer et al., 1999; Granel and Galez, 2002). The next sections describe major water threats to the MAM.

3.6.1. Population growth

The population of the MAM was 1,027,004 in 2010, almost double the population of 667,312 in 1990. Population density rise from 293 people per km^2 in 1990 to 450 people per km^2 in 2010. More than 50% of the total population of Yucatan State live in the MAM. The MAM is one of the faster growing regions from the 30 member nations of the OECD (OECD, 2008; COMEY, 2011). Based on CONAPO (National Council of Population in Mexico), projected population growth for 2030 in the MAM is 24%, therefore it is considered as a future challenge to develop a sustainable water supply system and sanitation coverage for adequate disposal of the wastewater discharged within this priority area (CCPY, 2012).

3.6.2. Groundwater contamination

Since 1999, the water underneath Merida city is considered inappropriate for human supply in line with the WHO standards. Such restriction is mainly because of leachates from solid waste and high concentration of chlorides from 100 up to 170 mg/L. Chloride contamination has been attributed to the infiltration of septic tanks and industrial chemicals such as chlorinated solvents (Alcocer et al., 1999). Additionally, dissolved solids concentrations are high in some areas associated to contamination from clandestine solid waste disposals, which has been a common practice for many years in all Yucatan. It was only since last decade, that three municipalities of the Yucatan State (Merida, Progreso and Uman) have built landfills with appropriate materials for solid waste disposal. Furthermore, open defecation, remains within the MAM area contributing as a source of contamination (Alcocer et al., 1999; Yucatan Government, 2013).

The major concern of water management due to the increasing population is in terms of water quality rather than water quantity. This is because the natural recharge capacity of the Yucatan aquifer is ten times higher than the current water extracted. Yucatan aquifer is considered an important national water reservoir, but of high vulnerability to contamination because of its karstic nature (COMEY, 2014; Hernández-Terrones et al., 2010).

Several studies suggest that groundwater mainly flows in a north-western direction across the Yucatan State (Figure 15), although karst fractures may change the groundwater direction at local level (Marin, 1990; Sanchez y Pinto, 1999; Steinich and Marín, 1997). Due to the nature of the aquifer, the groundwater is vulnerable to contamination from both natural and anthropogenic sources. Any pollutant that enters the groundwater system is rapidly spread out due to the high hydraulic conductivity, which implies rapid dilution at the local level but also rapid contamination of large water volumes (Felton and Currens, 1994). Historically, the main concern in terms of water quality has been the inability to provide an adequate sanitary drainage system in Yucatan. It is mainly because of two reasons: 1. the hardness and rock-like characteristics of the karstic soil and, 2. the surface-proximity of the water table underneath (5 meters in average) which does not allow drilling deep enough along all the MAM to build an adequate drainage system. Consequently, this has been driving the common practice of open defecation among rural areas.

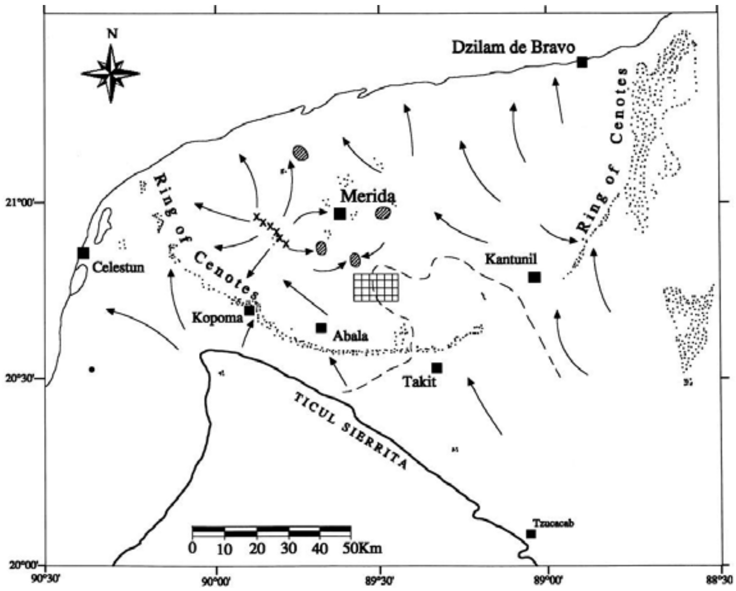


Figure 15 Natural water resource and groundwater flow direction in north-western Yucatan. Source: Escolero et al., (2000).

A potential solution but with good feasibility was suggested by Castillo et al., (2011) by implementation of a secondary treatment to the septic tank effluents. Special attention is needed in rural areas where most or even all of the water for human consumption and cultivation of crops is extracted from shallow wells where microbial pollutants tend to be more concentrated, leading to a significant risk for gastrointestinal diseases among the rural population (Alcocer et al., 1999).

During 2003-2008, a water quality monitoring study was performed for 106 main water wells, corresponding to one in each municipality of the Yucatan State. More than 86% of these wells reported total and faecal coliforms concentrations above the MCL for drinking water requirements. It was concluded that 24% of these were associated to human origin and 10% to livestock activity (Osorio, 2009). In addition, water supply wells in up to 21 municipalities had nitrate concentration above the MCL (45mg/l), which was associated with agriculture, livestock and domestic sources (Perez-Ceballos and Pacheco, 2004). Furthermore, Mendez et al., (2005), reported that 88% of the 106 main water supply wells had higher coliform concentration than the MCL. From these wells 23% were associated to human origin while 43% were associated to animal origin, which increases water pollution from livestock.

Due to the above, a particular concern is the livestock wastewater generated in the priority area, which in 2013 accounted for 2.98 million cubic meters, mainly from porcine activity. Special attention is needed for the solid excreta loads discharged together with these wastewaters through the aquifer. These excreta loads account to 1,417 tons per year from porcine, and 234 tons per year from poultry, together with 262,800 tons per year from solid urban waste, altogether producing lixiviation that goes straightforward to the Yucatan aquifer (Yucatan Government, 2013).

Other important contaminants detected in the Yucatan aquifer are high loads of organic matter from nixtamalisation wastewater from (residuals from corn processing by local factories) and sludge from septic tanks. Only for Merida, these loads account for 273,020 million m^3 per year. High waste loads from these activities are generated all around Yucatan State without treatment, which makes them difficult to manage and thus to control (Yucatan Government, 2013). A summary of documented nitrate concentrations in Yucatan groundwater is presented in Chapter 5, which pointed out the importance to monitor and control nitrate in the MAM case study.

3.6.3. Water-related diseases and public health

Mexican epidemiological statistics reports water-related diseases based on the Intestinal Infectious Diseases (IID) defined by the International Classification of Diseases (ICD-10). These are ten groups: A00: Cholera; A01: Typhoid and paratyphoid fever; A02: Other salmonella infections; A03: Shigellosis, A04: Other bacterial intestinal infections (i.e. *E. coli*; and *Yersinia*); A05: Other bacterial foodborne intoxication; A06: Amoebiasis; A07: Other protozoal intestinal diseases such as Balandiasis, Cryptosporidiasis, Giardiasis, and Isosporiasis; A08: Intestinal infections caused by viruses and other organisms; and A09: Other gastroenteritis and colitis of infectious and unspecified origin (Table 15).

Even though infant mortality rate in Mexico has decreased over decades (156/1000 in 1930; 39.9/1000 in 1980; 23.9/1000 in 1990; and 13.9/1000 in 2000); high values of IID case are reported, in most cases with no further identification of the specific pathogen causing the disease, these are generalised as “IID” group (Gonzalez et al., 1991; Mendez et al., 2004; Zaidi et al., 2006).

Table 15 Water-related diseases in Mexico

Disease group	Number of cases per year							
	2002	2003	2004	2005	2006	2007	2008	2009
IID	6.8E+6	6.2E+6	5.9E+6	5.91E+6	5.7E+6	5.5E+6	5.5E+6	5.5E+6
A01	7.9E+3	2.0E+4	2.6E+4	3.18E+4	3.7E+4	4.5E+4	4.4E+4	4.7E+4
A02	8.1E+4	1.0E+5	1.1E+5	1.10E+5	1.1E+5	1.2E+5	1.2E+5	1.4E+5
A03	3.1E+4	2.8E+4	2.2E+4	1.94E+4	1.6E+4	1.5E+4	1.3E+4	1.2E+4
A05	2.2E+4	3.6E+4	3.9E+4	4.06E+4	3.8E+4	3.6E+4	3.6E+4	3.9E+4
A08	5.4E+6	4.8E+6	4.8E+6	4.77E+6	4.7E+6	4.5E+6	4.6E+6	4.6E+6
Total	1.2E+7	1.1E+7	1.1E+7	1.09E+7	1.1E+7	1.0E+7	1.1E+7	1.1E+7

IID: Intestinal Infectious Diseases based on (WHO, 2010) comprises groups A00-A009 as follows A00: Cholera, A01: Typhoid and paratyphoid fevers; A02: Other salmonella infections; A03: Shigellosis; A04: Other bacterial intestinal infections i.e. E. coli; A05: Other bacterial foodborne intoxications; A06: Amoebiasis; A07: Other protozoal intestinal diseases i.e. Balantidiasis, Cryptosporidiasis, and Giardiasis; A08: Viral and other specified intestinal infections such as Rotavirus and Adenovirus; A09: Other gastroenteritis and unspecified origin. Source: CONAGUA, (2010a).

Mexican statistic data reported IID as the first or second cause of death in Mexico during 1930-1980. It dropped to the seventh cause of death in 1990, and to the fifteenth in 2000 with an incidence rate of 5.4 per 100,000 inhabitants and 5,216 total deaths. For 2008 it was reported as the 19th cause of death with incidence rate of 3.4 per 100,000 inhabitants and 3,574 deaths. Nevertheless, it is still one of the top five cause of death in Mexico for infant mortality with a reduced infant mortality rate (IMR) from 3,667.9/100,000 live births (LB) in 1922 to 37.1/100,000 LB in 2008 (SSA, 2011).

Of particular concern for Yucatan is the Acute Diarrhoea Diseases (ADD). In Mexico, ADD incidence has been increasing in the last 30 years due to improvements of methods for diagnosis (from 1716.5/100000 inhabitants in 1980, up to 7945.7/100000 inhabitants in 1998). Since 1999 it has been reduced from 7473.9/100000 inhabitants in 1999 down to 5264.2/100000 inhabitants in 2010), but it is still a major public health concern in Mexico (SINAVE, 2012). Similarly in Yucatan, ADD has been reduced (Table 16) but it is has been always higher than national rate, thus it remains one of the major public health concern to be tackle. In the MAM, main water-related diseases from 2007 to 2009 were diarrhoea and gastroenteritis based on health statistics (Table 17).

Table 16 Acute Diarrhoea Diseases (ADD) statistics in Yucatan

Year	Mexico		Yucatan	
	Cases	Rate*	Cases	Rate*
2000	6891063	7000.4	221113	13043.7
2005	5912952	5688.4	139288	7624.9
2010	5706232	5264.2	124590	6402.9

*Rate is given in number per 100, 000 inhabitants

ADD have a seasonal pattern considerable increasing during rainy season, thus it is called diarrhoea season (Dohering and Buttler, 1974; Lutz et al., 2000; Marin and Perry, 1994). For instance, in 2010 incidence rate per population age-group were as follows: 4.83 in adults, 47.64 in children less than 1 year-old, 2.31 in children between 1 to 4 years-old. Thus, children under 1 year-old and elders are at highest death risk from acute diarrhoeic disease in Yucatan.

Table 17 Water-related diseases number of cases in MAM from 2007-2009

#	Water-related diseases	Number of cases in MAM
1	Toxoplasmosis	3
2	Ascariasis	1
3	Strongyloidiasis	1
4	Typhoid and paratyphoid fever	2
5	Salmonellosis	2
6	Bacterial intestinal infections	14
7	Amoebiasis	4
8	Viral Intestinal Infections	3
9	Leptospirosis	2
10	Diarrhoea and gastroenteritis	247
11	Dengue	4
12	Hepatitis A	2
13	Cysticercosis	2
14	Intestinal Parasites	2

Source: INEGI, (2013)

In 2010 Yucatan has the third highest mortality rate of the 31 States of Mexico with 11 172 deaths due to IID (Table 18), which was almost the double of national death rate (4.83 per 100 000 Yucatan 2.9/100,000 National) for the same year (SINAVE, 2012).

Table 18 Mortality rate in Yucatan, from Intestinal Infection Diseases

Mortality rate from Intestinal Infection Diseases (IID)								
Population age/year	**1980**	**1985**	**1990**	**1995**	**2000**	**2005**	**2008**	**2010**
Pop <1	10.8	7.1	4.3	1.7	0.7	0.8	39.1	47.6
Pop 1-4	90.1	54.1	36.5	14.9	7.6	14.5	8.5	2.3
Pop 5-14	7.1	6	1.6	1.3	1	0.3	0.5	
Pop 15-64	44.7	44.7	18.9	10.7	6.9	4.7	4.2	
Pop >65	356.3	286.7	167.8	113.8	58	47.2	43.1	
Yucatan average	63.5	41.4	22.6	11.6	5.8	5.6	4.7	4.83

*Rate is given in number per 100, 000 people. Source: SINAVE, (2012).

In Merida, more than 40% of the deaths of children under-six years in 1960 was due to water-related pathogens. Between 1990-2000 Yucatan infant mortality rate (IMR) has been in average the highest among the three States of the Yucatan Peninsula (13.83), and also higher than the national average (13.8) as shown in Figure 16.

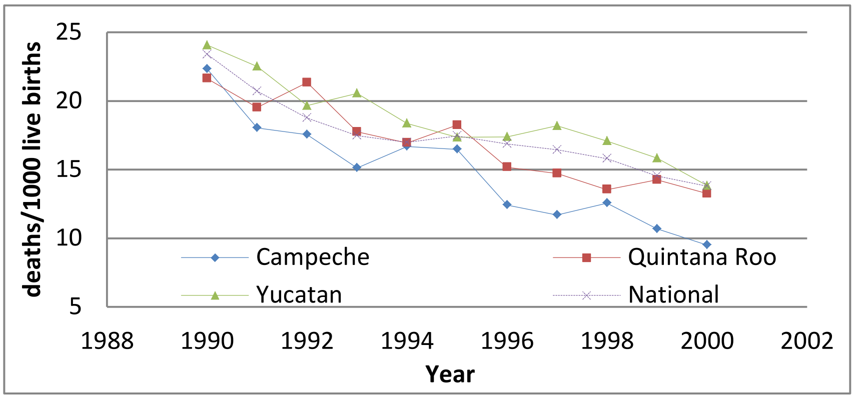


Figure 16 Infant mortality rate in the 3 states of the Yucatan Peninsula. Source: SSA, (2001); INEGI, (2001) cited by Mendez et al., (2004)

Even though water related diseases have decreased because of improvements in water supply system, faecal pollution from septic tanks, poultry and porcine farms remains one of the main issues in the aquifer of the MAM, thus of health (Alcocer et al., 1999; Mendez et al., 2004).

In 2000, acute diarrhoea diseases incidence in Yucatan was the highest in Mexico (8 698 per 100 000 inhabitants - Yucatan, 4 955 cases per 100 000 inhabitants -national) (Osorio, 2009; Mantilla et al., 2002; Alonzo and Acosta, 2003). Furthermore, Table 19 shows incidence rate for specific disease in 2011 which were significantly higher than the national rate (SINAVE, 2012).

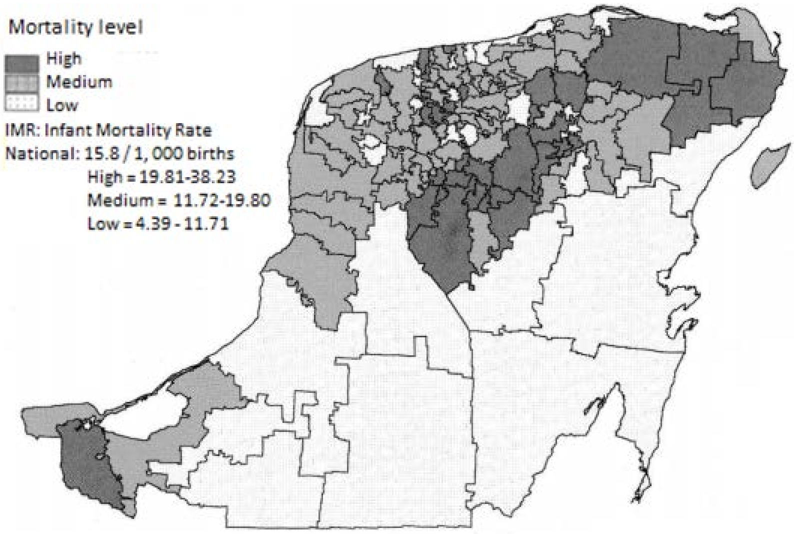
Table 19 Incidence rate of Intestinal Infectious Diseases in Yucatan, 2011

Intestinal infectious Diseases	Incidence rate*	
	Yucatan	Mexico
Intestinal amebiasis	745.54	384.15
Helminthiasis	464.48	284.96
Intestinal infectious from protozoans	179.62	71.9
Ascariasis	231.56	71.37
Paratyphoid and other salmonellosis	30.77	112.02
Typhoid fever	3.5	44
Bacterial food poisoning	21.42	40.71

*Incidence rate is given per 100, 000 inhabitants. Source: SINAVE, (2012).

A study in Yucatan aimed to identify the cause of salmonellosis in children with diarrhoea reported the same types of salmonella *(S. Typhimurium and S. Enteritidis)* as in raw meat (chicken, pork and steak meat). The study also reported antibiotic resistance from these types of salmonella; specifically *S. Typhimurium* was resistant to ten antibiotics (including ampicillin, chloramphenicol, trimethoprim-sulfamethoxazole, aminoglycosides, nalidixic acid and extended-spectrum cephalosporin). During the study (2003-2005) there were three deaths due to *S. Typhimurium* in children under-6 months. *S. Typhimurim, S. Typhi* and *S. Enteritidis*, which may cause sepsis and meningitis (Zaidi et al., 2006).

Mendez et al., (2004) have reported the highest infant mortality rate by municipalities in Yucatan State (Figure 17), which could be attributed to poor/deteriorated water quality through short-circuit contamination of the aquifer. Municipalities with the highest infant mortality rate (IMR) in 2000 were: Chichimila (71.11); Chankam and Chikindzonot (63.26), Yaxcaba (52.18), and Quintana Roo, Sudzal, Tunkas and Cenotillo (50.42). Morbidity causes in Yucatan in 2000, are shown in Table 20. Zoonotic diseases are intrinsically related to water quality due to the water vector transmission route of pathogens (Reyes-Novelo et al., 2011).



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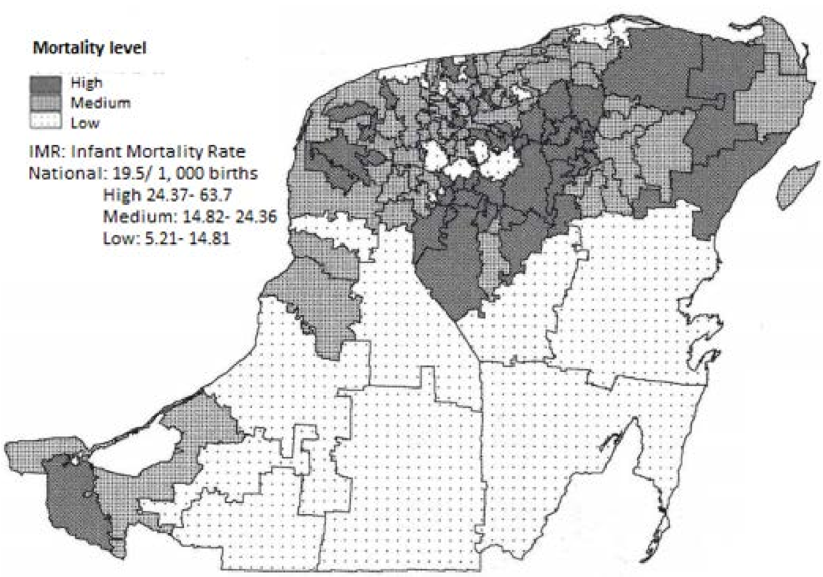


b.

Figure 17 Infant mortality rates (IMR) in the Yucatan Peninsula, a) 1990, b) 2000.
Source: Mendez et al., (2004)

For instances, Leptospirosis in Yucatan is excreted through animal urine and infects humans through direct ingestion of contaminated food or water. Diagnosis is difficult due to minimal symptomatology.

Table 20 Morbidity top ten causes in Yucatan, 2000

Cause of disease	Total	Rate
Acute respiratory infections	891796	53172.4
Intestinal infectious	150885	8996.3
Urinary infectious	78515	4681.3
Intestinal amebiasis	62037	3698.9
Gastritis, duodenitis and ulcers	41947	2501
Other helmintiasis	35015	2087.7
Ascariasis	25115	1497.4
Asma	18057	1076.6
Acute otitis media	13525	806.4
Varicela	12365	737.2

Source: Yucatan Government, (2009). Rate is given per 100,000 inhabitants

Nonetheless, Zavala-Velazquez et al., (1998) found 14% of patients first diagnosed with dengue in Yucatan were positive to *Leptospira interrogans,* with 2.2/100,000 inhabitants of incidence rate in 2000 in rural Yucatan, where it is predominant during rainy season (Vado-Solis et al., 2002b).

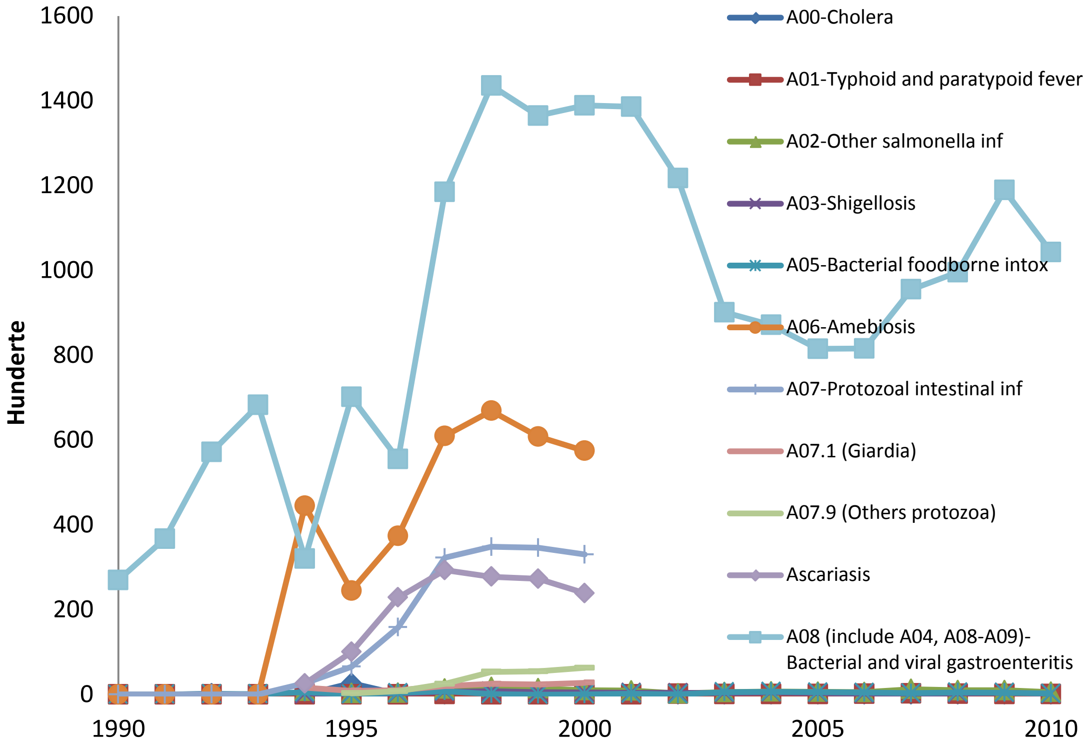


Figure 18 Water related diseases in Yucatan reported as Intestinal Infectious Diseases (IID). Source: Annual epidemiological information from Secretary of Health, 1990-2010, SSA, (2014).

Figure 18 summarises annual records (1990-2010) of water-related diseases reported for Yucatan State. It was found that the majority of water-related diseases in Yucatan were caused by the groups A04, A08 and A09.

Although Mexican government provides adequate water quality levels for drinking purposes, the majority of the urban population prefers bottled water as drinking water. This is mainly due to the unreliability of the quality control monitoring system at different water distribution network points. Mexican government rigorously monitors the water quality only at the potabilization plant, thus a subsequent water contamination may not essentially be detected. This is of important concern especially in the Yucatan Peninsula due to the close proximity of the water table, antiquity of the pipeline distribution system (>50 years old), and easy propagation of contaminants related to the lack of sewerage and wastewater treatment systems (JAPAY, 2009).

To summarise:

- Contamination of groundwater within the MAM has been evaluated through punctual field work research and periodic monitoring by water authorities. These could give a good insight to identify those potential pollutants, and pollutant's sources that might be causing most of the microbial and chemical water-related diseases.
- Pollutants such as total and faecal coliforms, nitrate, and heavy metals (chrome, cadmium, iron and lead) have been reported as extremely high in some MAM areas. In addition, high levels of secondary contaminants such as chloride, detergents, sulphate and hardness have been documented (Pacheco et al., 2004a; Osorio, 2009).
- Even though in National health statistics there are no diseases records related to chemical water pollution (i.e. heavy metals), diverse field studies have reported significant high concentration of these in the MAM aquifer. Thus these could help to the identification of potential water-related diseases (Marin et al., 2003; Gonzalez et al., 2007).
- Yucatan government first attempt to protect and avoid further groundwater quality deterioration created in 1994 a regulation for the control and adequate disposal of wastewater within the city of Merida (Yucatan Government, 1994). Ten years later, a new regulation was created to enforce the former, related to the adequate planning and construction around Merida. On this new regulation, specification of ST were established to be of 1.3L minimum capacity, with a minimum of 24hr retention time (considering 80% of water consumption); and with at least once ST empty within 2 to 5 years (Yucatan Government, 2004).

Chapter 4. Methodology for the model development

…"collaboration between academics (e.g. hydrologists geomorphologists, engineers and ecologists), practitioners (e.g. water and habitat managers) and stakeholders (e.g. landowners, anglers and recreational users) –(is needed)- *to balance the multiple and often conflicting pressures associated with the management of the system (e.g. flood alleviation and reservoir management) against the protection, and even enhancement, of ecosystem properties (e.g. conservation and restoration of habitats)" (Hannah et al., 2008).*

This chapter describes the model developed for this research, along with the underlying assumptions and selection of input parameters, it includes:

- A brief description of the model with scopes and limitations
- The model structure, based on the conceptual model (Figure 20)
- A discussion of the model assumptions and setting of relevant parameters

4.1. Model: Scopes and boundaries

The Sustainable Integrated Water Management (SIWM) model was designed and tested for the Metropolitan Area of Merida (MAM), case study of this research. Nevertheless, the model is structured in a generic sense; meaning that it could be applied to other cases studies taking into account the specific socioeconomic activities of a given area under spatial and temporal conditions. In general there were three main steps to develop the SIWM model: conceptualization, mathematical representation, and system dynamic modelling (Figure 19).

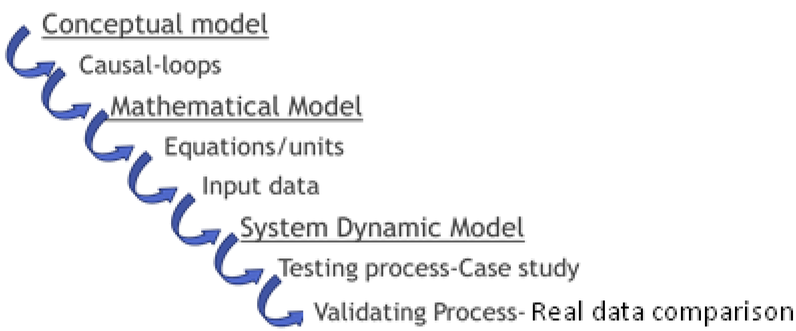


Figure 19 Steps and sub-steps to develop the Sustainable Integrated Water Management (SIWM) model of this research.

The scopes of the SIWM model are as follows:

- Identification of the most significant pollutant sources of public health concern for the study area.
- Forecast future concentrations of selected indicator pollutants in the case study.

- Model the effect of water management interventions on pollutant concentrations.

The limitations of this SIWM include:

- For the study area, due to incomplete or fragmentary (and sometimes inconsistent) data to set the required data input for model input parameters some data was estimated on the basis of “typical” pollutant concentrations reported in the literature.
- For the study area, a simplified approach was applied to the modelling of groundwater flow patterns and distribution of pollutants, which may not be robust enough to represent the karstic nature of the aquifer. This focuses on addressing regional and long-term water pollution issues rather than local or seasonal patterns.
- Modelled interventions are suggested on the basis of current technical status and cost-effectiveness thus needs to be updated for future scenarios.

4.2. Conceptual model

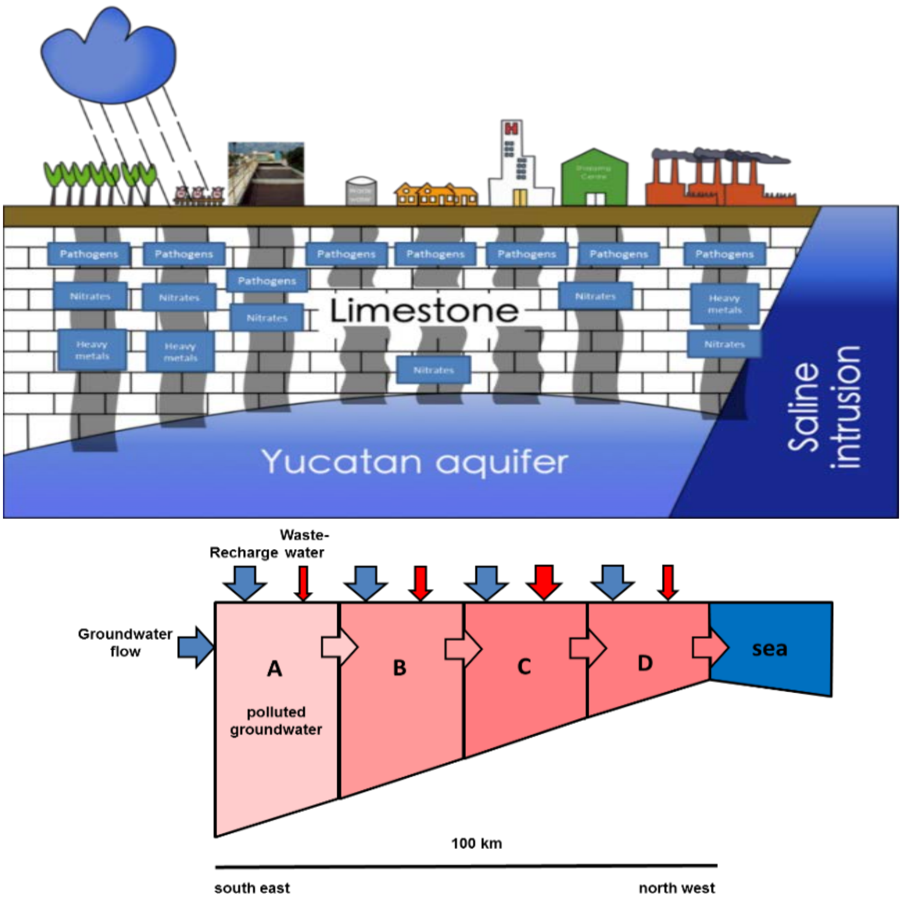


Figure 20 Top-diagram: Conceptual model of the case study area, as a physical representation of the groundwater pollution scenario. Bottom-diagram: Simplified graphical representation of the model structure. Colour within one aquifer section illustrates homogeneous distribution of pollutants within the water volume of this section. Colour grading between sections illustrates that different pollutant concentrations built in different aquifer sections, as a consequence of different wastewater inflow volumes and pollutant loads.

The starting point of the SIWM model was to build a conceptual model in order to represent a qualitative analysis of anthropogenic activities interacting with the water cycle. Figure 20 illustrates the central idea of the conceptual model for the MAM.

The left diagram shows how physically all wastewater discharged by the different socio-economic activities in the area infiltrates the aquifer (See Chapter 3 for study area description). Diagram in the right of shows a simplified graphical representation of the model structure, with a focus on aquifer sub-sections, groundwater flow, recharge, wastewater discharge, and sectional homogeneity of the aquifer with respect to pollutant distribution. This representation already implies certain assumptions which are explained in more detail in the next chapter.

These two diagrams were conceptually translated within system dynamics platform into the causal-loop shown in Figure 21. Vensim platform allows the visualization of all variables comprised by the system as well as their potential interactions, represented through arrows pointing toward and/or away from the variables, giving an idea of how the variables are interconnected and how the causal loops are embedded in the dynamic model structure. Once all variables and their interrelations are declared within the platform, Vensim run the model and shows results as tables and graphs.

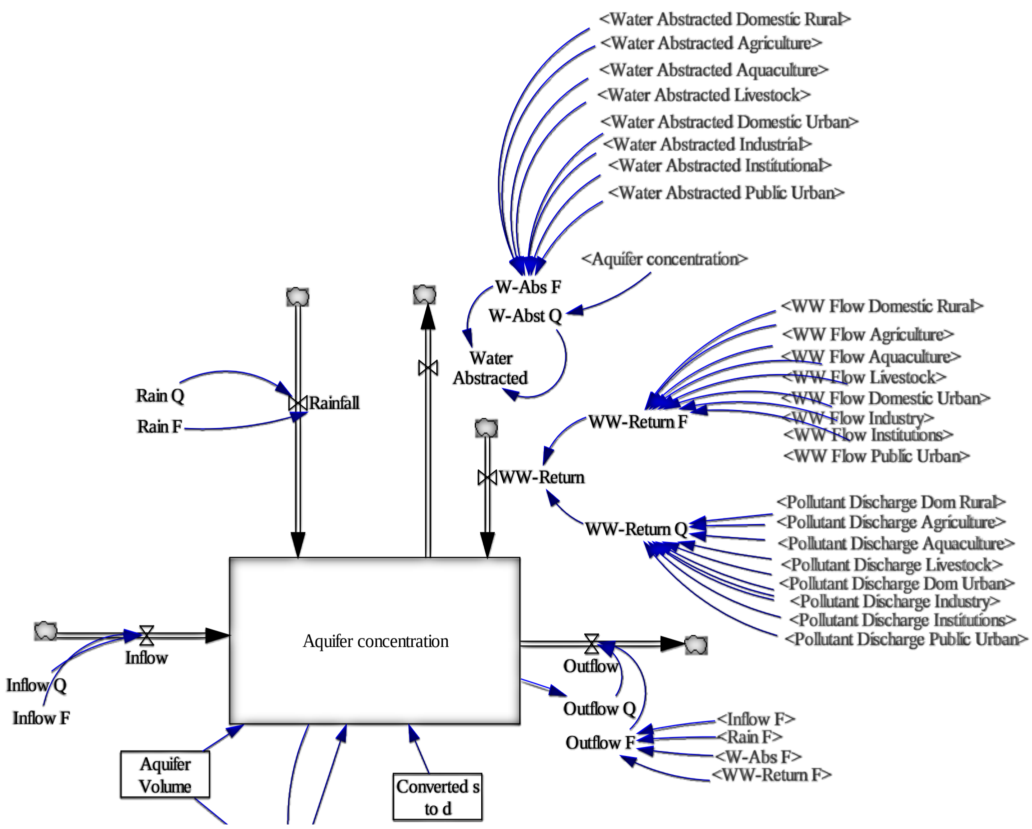


Figure 21 Causal-loop diagram of the Sustainable Integrated Water Management Model for the Metropolitan Area of Merida (MAM), Yucatan, Mexico. Arrows in and out indicates inflows and outflows for the MAM aquifer, each having its own F: Flow of water in m^3/s, andQ: Quality parameter such nitrate concentration in mg/m3 or faecal coliforms in CFU/m^3, W means “water”; WW means “wastewater”.

Based on the literature (Pokrajac, 1999; Jonch-Clausen, 2000; Liu et al., 2008; Arellanos, 2009) and the current scientific evidence of the Yucatan aquifer conditions (Marin et al., 2003; OECD, 2008; Holliday, 2007), the sustainable and integrated water management shown in Figure 22 was derived.

This approach also considers major problems of water management in Latin American and the Caribbean countries discussed by San Martin, (2002), which emphasize the imperative need to shift to a more integrated and comprehensive approach in the water sector. Figure 22 includes the anthropogenic factors acting as major water polluters, linked to the natural water system in order to identify those interconnections where water-human management could be assess to quantify public health risks through the SIWM model. The SIWM model ultimately provides pollutants concentrations which serve to quantify microbial and chemical risks, which are translated to cost-saved in the public health sector. These costs-saved are benefits quantified by cost-benefit analysis, in order to recommend the most cost-benefit public health intervention.

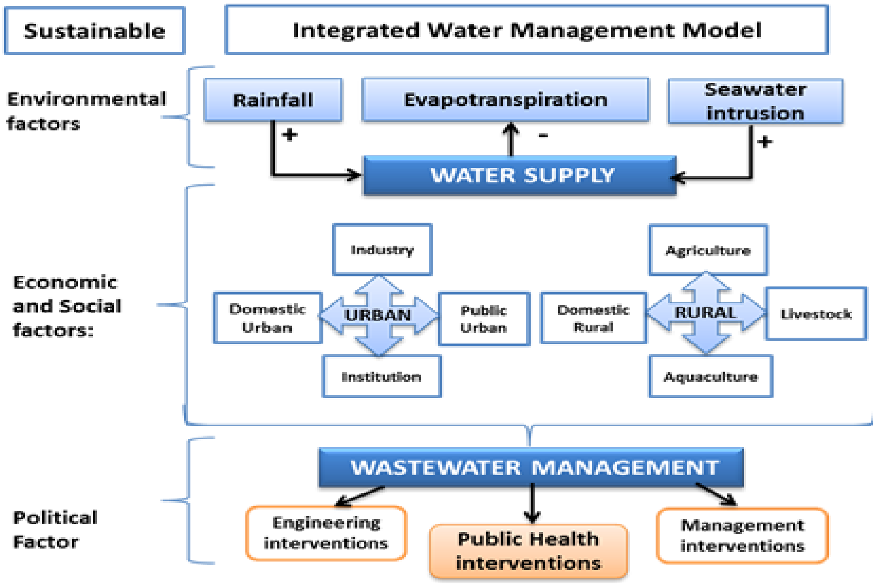


Figure 22 Concept of the SIWMM, relating natural and anthropogenic influences on water supply and quality, wastewater management and public health

4.3. Model structure

This section includes: description of the study area, sub-models comprising the SIWM model, and a simplified graphical representation of the model structure.

4.3.1. Study area

The study area encompasses a rectangular area of about 100 km x 45 km that includes the MAM defined by the Yucatan government along with peripheral municipalities of the MAM (Figure 23). The geographic orientation of this area is based on the groundwater flow direction (southeast to northwest), to consider pollutant transport along with groundwater flow.

This area is divided into 4 equally-sized sections: A, B, C and D from southeast to northwest direction, referred to as “aquifer sections”. Each section has its own characteristic for example in terms of population and wastewater load (both are highest in C that includes majority of Merida City and lowest in A, a predominantly rural area).

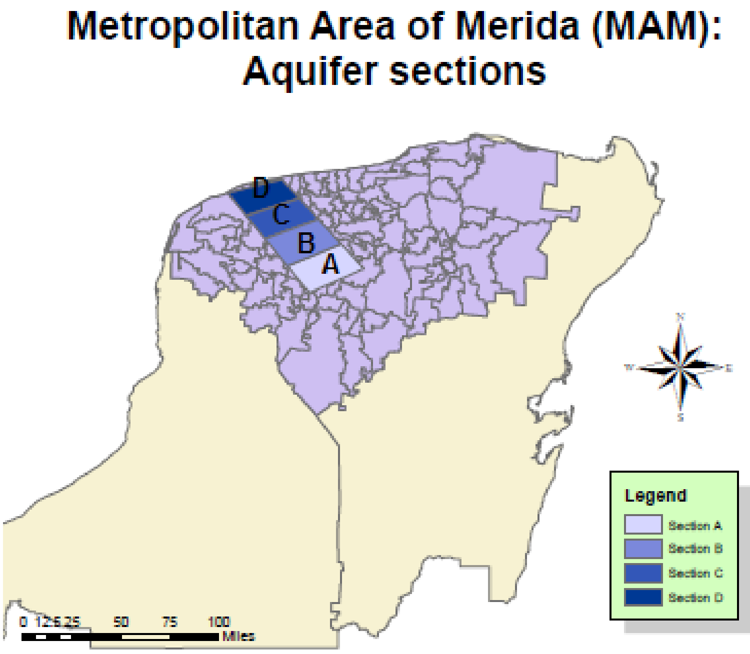


Figure 23 Rectangular study area of the model with the four aquifer sections A, B, C and D.

4.3.2. Sub-model structure

The model includes 10 sub-models, representing the water cycle (aquifer sub-model) and human influence (9 sub-models). Aquifer sub-model was conceived as the environmental factor for the conceptual model of this research (Figure 20). It comprises water quality data as input and output from the MAM case study.

Table 21 Sub-model structure of the present research

Factor	Group	Sub-model	Coding name in the model
Water cycle	Aquifer	Aquifer	AQUIFER
Human	Population	Population	POP
	Urban	Domestic Urban	DU
		Industry	IND
		Institution	INS
		Public Urban	PU
	Rural	Domestic Rural	DR
		Agriculture	AGR
		Aquaculture	AQU
		Livestock	LIV

The model was developed with data from the 4 aquifer sections of the MAM case study (Figure 23), which was delimited considering the site-specific groundwater pollution documented and data availability from local research. The human influence comprises 9 sub-models. The population sub-model, which affects all the other 8 sub-models that represent the anthropogenic activities and are classified into two groups: “urban” and “rural”, as shown in Table 21 together with their coding name used within the model environment.

Figure 24 shows the 3 stages of data flow from input data, data processes and calculations, and final outputs. Once the most significant health risks associated with drinking water are identified, by quantifying pollutants concentration in the aquifer, potential public health interventions are modelled. Quantitative Microbial Risk Assessment (QMRA) was performed in order to estimate the burden of diseases for specific pollutants. Finally, cost-benefit analyses for the public health interventions are evaluated to identify interventions that could most effectively tackle the highest public health risks related to specific pollutants.

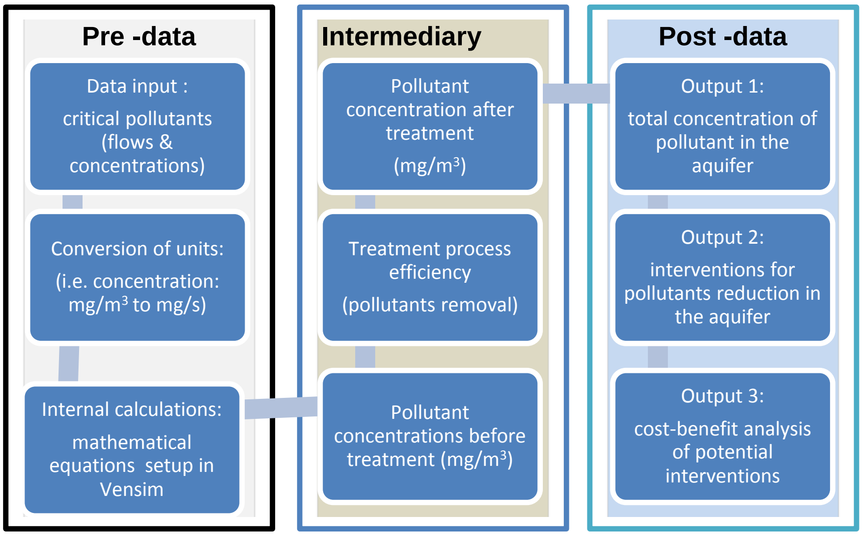


Figure 24 Data flow of the Sustainable Integrated Water Management (SIWM) model

In order to explain the relevance of each of ten activities (sub-models in the SIWM model), a brief description is presented below. These impacts qualitative and quantitative to the water cycle through the pollutants loads estimated as wastewater discharged. Pollutants load values were estimated based on the total production or number of people (i.e. workers, students) per specific activity, which were obtained from statistical databases of the MAM case study.

1. *AQUIFER*. This is segmented in 4 sections (A, B, C, and D), taking into consideration the very different water consumption and wastewater generation within each section. It serves to integrate the 8 human activities-based sub-models in order to represent a holistic scenario of the water cycle for the MAM case study, and the scenario analysis for all interventions.
2. *POP*. This sub-model was built as a dynamic element linked to the growth rate for the 8 activities, which are depending on population growth.
3. *AGR*. It comprises three main agricultural groups: vegetables, fruits and maize. Vegetable group includes tomatoes, cabbage, onion, and pepper. Fruits group

includes watermelon, pineapple, banana, and citrus. Maize was treated as single group due to its significantly high production and water requirements.

4. *AQU*. It includes two main aquaculture groups: shrimps and fish. Due to the significant production, shrimps are considered a separate group, whereas the rest of the aquaculture production is grouped as “fish”, including mainly tilapia and ornamental fishes.
5. *LIV*. It includes four main livestock activities: porcine, ovine, poultry and bovine. Poultry comprises: chicken, turkey and duck production.
6. *DR*. This includes rural households classified according to the type of wastewater treatment currently operating: HH1. Households connected to septic tanks (HH-ST), HH2. Households connected to improved septic tanks (HH-iST), as an intervention; HH3. Households connected to wastewater treatment plant (HH-WWTP), and HH4. Households connected to improved wastewater treatment plants (HH-iWWTP) – as an intervention.
7. *DU*. This includes the same groups as in DR.
8. *PU*. This includes two groups: services and trade. Trade comprises shopping centers, city market, and small shops. Service includes all public spaces such as parks, museums, stadiums, etc. in the urban area.
9. *INS*. This comprises hospitals, hotels, and offices and schools. Offices and schools are grouped together because of the similarity on water consumption and type of pollutants discharged.
10. *IND.* This includes two main groups: manufacture and construction. Manufacture comprises food, plastics, wood, metals, textile, mechanic, electronics, and other manufactured products. Construction includes all registered contractors and activities related to the construction sector, including sites of materials extraction.

4.4. Model input: Assumptions and parameter settings

The study area (Figure 23) comprises altogether 31 municipalities, some of which are entirely located within an aquifer section but the majority are fractionalized and grouped into the corresponding area. Since many relevant data are registered at municipality level, these were implemented as weighted contributions by area percentage of each municipality. Table 22 gives an overview of important literature sources from which relevant data have been extracted.

4.4.1. Population

This sub-model represents the internal engine to establish the dynamics of the model by building the simulation on the quantitative relation of human activities to the population growth. This is the case in particular for public urban, institutions, domestic urban and domestic rural activities, and accounts in good approximation for the remaining activities (the growth of livestock activity, for instance, has been slightly higher in recent years due export of porcine meat to the USA).

As an example, Figure 25 shows the schematic representation in Vensim® of the interconnection between population sub-model and domestic water consumption in aquifer section A.

Table 22 Major sources of input data for modelling the case study

Data	Source	Comprehensiveness and time period
Catchment characterization: Biological, chemical, and heavy metals; GPS data	Dr. Julia Pacheco project: "Protection of peripheral zone and influence wells areas to supply the current MAM".	Most recent data generated of groundwater quality in Yucatan (2010-2012)
Annual production of agriculture and livestock in Yucatan State	SIAP: Yucatan State Office of Information for Sustainable Rural Development	Statistic database of the institution (1990-2010)
Urban and rural statistic of the MAM	YUCATAN: Yucatan Government Portal	Statistic database of institutions (1990-2010)
Statistic data of aquaculture in the Yucatan State	SFAYP: Ministry of Agriculture and Fisheries of Yucatan State	Institution database (1990-2010)
Water quality and quantity of the Metropolitan Area of Yucatan	JAPAY: Board of Water Supply and Sewerage of Yucatan State-personal communication.	Statistic data of the institution (1990-2010)
Statistic data of rural municipalities of MAM and statistics of Yucatan	OEDRIS: Yucatan State Office for the Sustainable Rural Development	Rural activities database (2008-2012)
Environmental Plan for Yucatan State and environmental data for the MAM	SEDUMA: Ministry of Urban Development and Environment of Yucatan Government	Yucatan State water resource management plans (2008-2012)
Rural and Urban areas of Yucatan State	SEMARNAT: Ministry of Environment and Natural Resources	Statistic data from the database of the institution (2000-2010)
Aquaculture areas of Yucatan State. Data per activity: total number and production.	CONAPESCA. National Commission of Aquaculture and fishing. (SAGARPA)	Punctual data from a sample of 80 places from Yucatan State. (2012-2013)
Quality and quantity of Yucatan water basin. Domestic and industrial wastewater characterization.	CONAGUA. (National Water Commission)	Data from internal monitoring quality control (2009-2011).
Water nexus public health statistics of monitoring control for Yucatan State	COFEPRIS: Federal Commission for the Protection against Sanitary Risk	Statistic data from the database of the institution (2008-2010)
Yucatan health statistics for last 10 years: mortality, nativity, causes of death.		SSY (Health Secretariat of Yucatan State)	Statistic data from the database of the institution (2000-2010)
Population statistics: households, economic activities, GPS location.	INEGI (National Population and household census)	Statistic database of the institution (2000-2010)
Wastewater treatment plants characterization: Removal efficiency of chemical and biological indicators	Determination of pathogen microorganisms present on domestic wastewater treatment plants (Basulto-Solis, 2010).	Punctual study of 3 of 17 wastewater treatment plants in Yucatan State (2008-2010).

Population growth is estimated by the following integral, which use historic population records from 1990 to 2010, and then extrapolate these data for the full simulation period (2010-2060).

$$Population\ growth = \int_{to}^{t}[Births(s) - Deaths(s)]ds + Population\ (i) \qquad \text{Equation 1}$$

This equation is equivalent to the defined equation used in Vensim® as follows:

$$Population\ growth = INTEGRAL(Births - Deaths,\ Population(to)) \qquad \text{Equation 2}$$

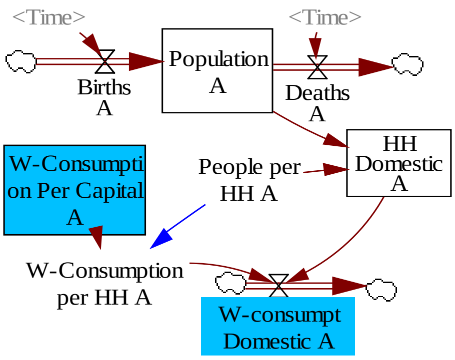


Figure 25 Interconnection between population and domestic sub-models, exemplified for aquifer section A of the MAM case study

From historic records of the study area, an annual population growth of 1.74% has been derived for all 4 sections of the MAM.

4.4.2. Aquifer

a. Aquifer sections area and volume

The water volume of each aquifer section (Table 23) was calculated as the product of section area, average aquifer "thickness", and aquifer porosity. Average values for aquifer section thickness were derived from depth data of the municipalities located in the section. Aquifer porosity is the fraction of water volume over total space volume, the remaining volume being taken by soil minerals.

Table 23 Water volume of the four aquifer sections

Aquifer section	Area (km^2)	Aquifer thickness (m)	Aquifer porosity (Dimensionless)	Aquifer water volume (m^3)
A	1148	72	0.35	2.89E+10
B	1148	55	0.35	2.14E+10
C	1141	44	0.35	1.99E+10
D	1144	21	0.35	5.42E+09

Source: Authors estimates based on: aquifer area from INEGI,(2010); aquifer thickness from Gonzalez-Herrera, (1992); aquifer porosity from González-Herrera et al., (2002).

b. Aquifer recharge

Average annual rainfall data for the Yucatan municipalities is reported by the national government (INEGI, 2010), from historic records (1960-2009), which were used to calculate the rainfall per aquifer section (Table 24).

Table 24 Rainfall per aquifer sector in (mm/year)

Aquifer section	Rainfall- as annual precipitation (mm/y)
A	1305
B	1268
C	1264
D	1013

Source: INEGI, (2010), average values from historic records (1960-2009)

The water balance of the Yucatan karst aquifer in the study area is maintained by groundwater flow (with coastal outflow) and recharge by precipitation. Recharge is a crucial parameter for the modelling of contaminant concentrations since it determines the “dilution” of contaminants by both rainwater and flowing groundwater. Recharge of the aquifer within the study area results from average annual precipitation, after abstraction of actual evaporation.

Estimates of the water recharge (as % of annual precipitation) for larger areas of the Yucatan karst aquifer have been given by several authors, and the values vary somewhat, most of them within a range of 10 - 20 % (Table 25). Based on these data, a uniform value of 15% of the average annual precipitation per section has been assumed as aquifer recharge per section.

Table 25 Data reference to estimate effective rainfall for aquifer recharge

Reference	Area	Recharge	Method
Bauer-Gottwein et al., (2011) based on Lesser, (1976)	Yucatan aquifer	14	Water balance
Aranda, (2011) based on Hanshaw and Black, (1980)	Yucatan aquifer	5-15	Rain - Eo
Graniel, (2010) based on BGS et al., (1995)	Merida	9	Rain - Eo
Graniel (1999) based on Rodriguez, (1984); SARH, (1988), and Black, (1988)	Merida	15-25	Rain - Eo
Gondwe et al., (2010)	Yucatan Peninsula	23	Rain - Eo
CONAGUA, (2010a)	Yucatan Peninsula	15	Runoff fraction of rain
Gonzalez-Herrera et al., (2014)	Yucatan aquifer	20	Assumption

Eo: Evapotranspiration; Recharge is given as % of precipitation

c. *Groundwater Flow*

As already mentioned in Chapter 2, karstic aquifers in general are complex and heterogeneous systems with dissolution conduits, fractures and underground channels, which makes very challenging the modelling of groundwater flow. It is generally accepted that the karst aquifer in the north-western part of Yucatan (including the study area) is characterized by high hydraulic conductivity but very low hydraulic gradient of only 7-10 mm/km. Groundwater velocity measurements in the Yucatan aquifer have recently been summarized and are highly variable, ranging from only 7 millimetres per day to 10 meters per day for conduit flow (Casares-Salazar et al., 2013). While small-scale laboratory experiments with soil or rock samples may reveal low hydraulic conductivity, the effective hydraulic conductivity of the Yucatan karst aquifer at the 100 km scale (as for the study area) may reach very high values of 1 m/s (Worthington and Ford, 2009).

The north western Yucatan aquifer has been modelled by implementing such high effective hydraulic conductivity values and assuming a homogeneous porous medium (Gonzalez, 2002). The model reveals a close match between the simulated and measured water table in the area. The water table contours indicate a groundwater flow direction from southeast to northwest, i.e. parallel to the long edge of the rectangular study area (Figure 23) of the present research.

The present research also assumes an equivalent porous medium for the aquifer sections, as well as an equivalent groundwater flow from southeast to northwest. Groundwater flow velocity, however, increases from section to section (A to D) due to additional contributions of recharge and decreasing aquifer section volumes.

d. Groundwater inflow into section A

Groundwater inflow from outside the study area is mainly from the Ring of Sinkholes, located at the south-eastern end of section A, this is the starting point of the study area. The fraction of groundwater that flows through the Ring of Sinkholes (RS) in north western direction (i.e. towards Progreso the geometrical centre of the ring) is effectively feeding aquifer section A of the study area.

In order to estimate this flow, the following assumptions and data were considered:

- The RS is the only source of groundwater inflow for the aquifer section A
- Water flow in the RS divides into: 40% to east coast, 40% to west coast, 20% north west
- According to INEGI (2002) total recharge (rainfall - evapotranspiration) in the RS is 1317 Mm^3/y, so:

If total recharge in the RS = $1317 \times 10^6\ m^3/y$

Then total groundwater flow from RS in all three directions = $41.76 m^3/s$

Then assuming that 20% of the total flow from RS is entering section A yields the value $8.32 m^3/s$ for groundwater inflow into aquifer section A.

4.4.3. Water abstraction and wastewater discharge flows

In this section, a detailed description of water abstraction and wastewater discharge estimates for each of the 8 socioeconomic activities is presented.

4.4.3.1. Domestic Urban (DU)

Water usage for each type of household (HH) is a function of the total population in urban area and the water usage per capita. Table 26 shows the total number of households per aquifer section, together with the connection to different wastewater treatment facilities in 1990. Table 27 shows total water usage and wastewater discharge per aquifer section in 1990. For per capita water consumption, statistical data for the Mexican population in 2006 was 4.23×10^{-6} m^3/s (740 m^3/y) (OECD, 2009).

Total population of aquifer sections was obtained from Mexican census (INEGI, 1990). The wastewater discharge shown in Table 27 was obtained by assuming that 80% of total water consumption is transformed into wastewater.

Table 26 Total urban households by treatment option (HH1-HH4), 1990

Aquifer section	Domestic Urban HH (number)				
	HH1	HH2	HH3	HH4	Total
A	4958	0	261	0	5219
B	23745	0	1250	0	24995
C	65106	0	3427	0	68533
D	33578	0	1767	0	35345
Total	127387	0	6705	0	134092

HH1: households connected to septic tanks; HH2: households connected to improved septic tank; HH3: households connected to wastewater treatment plants; HH4: households connected to improved wastewater treatment plants.

Table 27 Water use and wastewater from domestic urban (m^3/s), 1990

Aquifer section	Total HH* (number)	Water usage	Wastewater discharge
A	5219	8.83E-02	7.06E-02
B	24995	4.23E-01	3.38E-01
C	68533	1.16E+00	9.28E-01
D	35345	5.98E-01	4.78E-01
Total	134092	2.27E+00	1.82E+00

* HH: Household; DU: Domestic Urban

4.4.3.2. Industry (IND)

Mexican government data on industrial water usage in the study area are rather limited since industries usually have their own water supply infrastructure and are not dependent on the public water supply system. To estimate industrial water consumption and wastewater production in the study area, the number of employees by industry sector in the MAM was extrapolated from Mexican government statistical data, and typical sector-specific water consumption per employee was derived from international literature.

Table 28 shows the average water usages for different industries based on number of employees (Smith et al., 2012). An average of these values was used for water usage for manufacture industries of the study area.

Water usage in construction comprises temporary accommodations, tool washing, wet trades (i.e. brickwork, screening, concreting and plastering); ground works (i.e. grouting and drilling), dust suppression (i.e. road and wheel washing); hydro-demolition, among others (Waylen, 2011). Meanwhile for manufacture industries, water use includes cooling machines, cleaning and sanitary (Hoi and Mui, 2002).

Table 28 Typical water use and wastewater discharge for different industries

Industry	Water-use (GED)[1]	Water use (m^3/s*employee)	WW-discharge (m^3/s*employee)[2]
Construction	31	1.36E-06	1.16E-06
1. Plastic	120	5.28E-06	4.49E-06
2. Chemical	833	3.66E-05	3.11E-05
3. Wood	2144	9.43E-05	8.02E-05
4. Metal	738	3.25E-05	2.76E-05
5. Electric	284	1.25E-05	1.06E-05
6. Textile	1660	7.30E-05	6.21E-05
7. Food & Beverage	1967	8.65E-05	7.35E-05
8. Materials	86	3.78E-06	3.22E-06

[1]GED: Gallons per Employee per Day. [2] Wastewater discharge was estimated based on the assumption that 85% of the water usage is converted to wastewater (Metcalf and Eddy, 1991; Metcalf and Eddy, 2003).

One of the major water-using manufacturing industries is food processing. It spends most water on washing and carrying products through the plants. A characteristic of their wastewater is a high concentration of organic waste. Of particular concern is the meat processing, resulting in grease and fats contaminants that are difficult to remove with conventional treatments (Tchobanoglous & Schroeder, 1987).

Both types of industries are predominantly within the MAM. The latest census of industries in 2013 reported a total of 516 construction companies in Yucatan from which 477 (92%) are located within the study area. For the same year, a total of 534 manufacturing companies have been reported in Yucatan, from which 480 (90 %) are located within the study area, being food processing the main manufacture industry with 113 companies (INEGI, 2014; SIEM, 2014).

Total number of employees per industries in 1990 is show in Table 29. The 1990 value was obtained by extrapolating back (based on population growth) the available data of industries per aquifer section.

Table 29 Total number of employees by industry, 1990

Aquifer section	Construction industry	Manufacture Industry	Total
A	0	0	0
B	740	1180	1920
C	2705	4666	7371
D	1273	1918	3191
Total	4718	7764	12482

As it is mentioned in the literature, data of water usage for industries is rather limited and is not reported as part of the public water supply. Usually industries have their own supply and are not dependent on the public water supply system (Metcalf and Eddy, 1991; Smith et al., 2012). This is the case in the study area, thus industrial water usage reported by government may be underestimated.

Water usage is a function of the number of employees per type of industry and typical water usage per employee (Table 30). Total number of employees per industry in 1990 was estimated by extrapolating back from the national economic census (INEGI, 1998, 2004, 2008).

Typical wastewater discharges from different industries (Table 30) were estimated by assuming an average of 85% of the water usage in industry is converted in wastewater (Metcalf and Eddy, 1991; Metcalf and Eddy, 2003).

Table 30 Water use and wastewater from industry (m^3/s), 1990

Aquifer section	Construction industry		Manufacture industry		Total	
	Water use	Wastewater	Water use	Wastewater	Water use	Wastewater
A	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
B	1.01E-03	8.58E-04	5.34E-02	4.54E-02	5.44E-02	4.63E-02
C	3.69E-03	3.13E-03	2.07E-01	1.76E-01	2.11E-01	1.79E-01
D	1.74E-03	1.48E-03	8.89E-02	7.55E-02	9.06E-02	7.70E-02
Total	6.44E-03	5.47E-03	3.49E-01	2.97E-01	3.56E-01	3.02E-01

This assumption was applied to Yucatan industries based on literature review and due to the fact that there is no wastewater reuse reported by the industrial sector (SEMARNAT, 2008). From the literature review in Mexico, beverage and food industry reported a total water consumption of 15,938m^3/month and a wastewater discharge of 14,095m^3/month which corresponds to the 88% of the water usage converted in wastewater. Water consumption for metal industry from the same study was 25,900 m^3/month and wastewater discharge was 19,130 m^3/month, which correspond to the 74% of the water usage converted in wastewater (Cortez et al., 2009).

4.4.3.3. Institutions (INS)

The institution sub-model developed in this research is covering four types of institutions: hospitals, hotels, offices, and schools. Offices and schools were grouped together based on the similarity of both water requirements and wastewater characteristics. Measurement of water usage and wastewater discharge from institutions should be based on particular characteristics such as size of the facility, water use per student (for schools), per bed (for hospitals), or per any representative units used to define water usage (Tchobanoglous and Schroeder, 1987; USGS, 2000). The latter is because those measurements vary according to the geographical location, climate and type of facility (Metcalf and Eddy, 1991). Nevertheless, due to lack of data for water usage in specific activities for the case study, average water usage and average wastewater discharge were taken from the literature (NCDENR, 2009; Qasim, 1999; Tchobanoglous and Schroeder, 1987; Metcalf and Eddy, 2003; Metcalf and Eddy, 1991).

Average water and wastewater values per sub-activities from institutional facilities are reported in Table 31. For Yucatan case study, Table 32 to Table 34 present these total values per type of institution. A summary of water usage and wastewater discharge from all institutions for the baseline year of simulation (1990) is reported in Table 37. Total number of facility (or measure unit used) corresponds to the activities reported from national census (INEGI, 1990; SEFOE, 2014).

Table 31 Average data of water use and wastewater from institutions

Sub-activity	Unit	Water use (L/unit*day)	WW flow (L/unit*day)	% of Water use converted in WW
Hospitals	Bed	741	693.5	93.6
Hotels	Guest	190	182.4	96
Offices & Schools	Employee	65.2	62.1	91.0

*Average values. Sources: water use from Tchobanoglous and Schroeder (1987),Metcalf and Eddy (1991); wastewater discharge from Metcalf and Eddy, (1991); Metcalf and Eddy, (2003). WW: Wastewater.

Water usage in hospitals is a function of the number of beds, average water use per bed and wastewater discharge rate (Table 32). As the data available in Yucatan is given in total number of hospitals, an average number of 15 beds per hospital were assumed. Then the total number of beds per aquifer section was obtained by multiplying the average of 15 beds per total number of hospitals in each municipality, times the fraction of the municipality that correspond to the geographic location within the aquifer section. Water usage in hotels is a function of the number of guests, average water usage per guest, and wastewater discharge rate (Table 33).

Table 32 Water use and wastewater from hospitals (L/day), 1990

Aquifer sections	Total beds	Water usage	Wastewater
A	257.7	1.9E+05	1.8E+05
B	554.8	4.1E+05	3.8E+05
C	969.5	7.2E+05	6.7E+05
D	678.4	5.0E+05	4.7E+05

Table 33 Water use and wastewater from hotels (L/day), 1990

Aquifer sections	Total guest	Water usage	Wastewater
A	1.7E+01	3.3E+03	3.1E+03
B	2.0E+04	3.9E+06	3.7E+06
C	7.6E+04	1.4E+07	1.4E+07
D	4.4E+04	8.4E+06	8.0E+06

The data available is in total number of hotels from this, an average of 25 guests per hotel was assumed. The total numbers of guests per aquifer section was obtained by multiplying the average of 25 guests per total hotels in each municipality times the municipality fraction that corresponds to the geographic location within the aquifer section. Wastewater discharge in schools and offices is a function of the number of students, average water use per student and wastewater discharge rate (Table 34).

According to the literature the total flow of water usage in schools is converted to wastewater discharge (Metcalf and Eddy, 1991). Therefore, a hundred per cent of the water usage in schools is used as wastewater discharge value for schools.

Table 34 Water use and WW from schools and offices (L/day), 2010

Aquifer sections	Students type 1	Students type 2	W-use schools	WW schools	Employees in offices	W-use offices	WW offices
A	4.3E+02	1.1E+04	6.4E+05	6.4E+05	1.1E+03	5.8E+04	4.7E+04
B	7.2E+03	3.8E+04	2.8E+06	2.8E+06	4.1E+04	2.2E+06	1.8E+06
C	2.4E+04	1.1E+05	8.6E+06	8.6E+06	7.5E+04	4.0E+06	3.3E+06
D	1.3E+04	5.3E+04	4.1E+06	4.1E+06	2.8E+04	1.5E+06	1.2E+06

Type 1: High school & University; Type 2: Kinder, elementary & secondary school. W-use: water usage; WW: wastewater

The total number of students per aquifer section was obtained by multiplying total number of students per municipality times the fraction of the municipality that correspond to the geographic location within the aquifer section.

The total number of employees from offices was obtained by multiplying the total number of employees per municipality by the fraction of the municipality that correspond to the geographic location within the aquifer section.

Overall, offices and schools are the biggest water users from the institutions evaluated (Table 35). It may be because of the large numbers of young people that represent the majority of the Yucatan State population. An exception is hotels water usage in aquifer sections C and D, where majority of touristic area is located.

Table 35 Water use from institutions (L/day), 2010

Sub-activities	Unit	Aquifer sections				Total Water usage (m^3/s)
		A	B	C	D	
Hospitals	Bed	1.9E+05	4.1E+05	7.2E+05	5.0E+05	2.1E-02
Hotels	Guest	3.3E+03	3.9E+06	1.4E+07	8.4E+06	3.1E-01
Offices & Schools	Emp/Student	7.0E+05	5.0E+06	1.3E+07	5.6E+06	2.8E-01
Total		8.9E+05	9.3E+06	2.8E+07	1.4E+07	6.1E-01

In terms of geographical location, the highest consumption of water and hence highest discharge of wastewater is located in section C (Table 36 and Table 37); it is because most of Merida the capital city is located in section C, which has the majority of institutions infrastructure.

Table 36 Wastewater discharge from institutions (L/day), 2010

Sub-activities	Unit	Aquifer sections				Total WW (m^3/s)
		A	B	C	D	
Hospitals	Bed	1.8E+05	3.8E+05	6.7E+05	4.7E+05	2.0E-02
Hotels	Guest	3.1E+03	3.7E+06	1.4E+07	8.0E+06	3.0E-01
Offices & Schools	Emp/Student	6.9E+05	4.6E+06	1.2E+07	5.3E+06	2.6E-01
Total		8.7E+05	8.7E+06	2.6E+07	1.4E+07	5.0E+07

These available data for 2010 were extrapolated back to the year 1990, based on population growth rate of 1.74%/y. Table 37 summarises institutional water usage and wastewater discharge for 1990, as baseline input data for the model.

Table 37 Water use (WU) and wastewater (WW) flows in m^3/s from institutions, 1990

Aquifer sections	Office & Schools		Hospital		Hotel		Total (m^3/s)	
	WU	WW	WU	WW	WU	WW	WU	WW
A	5.7E-3	5.6E-3	1.6E-3	1.5E-3	2.7E-5	2.6E-5	7.3E-3	7.1E-3
B	4.0E-2	3.7E-2	3.3E-3	3.1E-3	3.2E-2	3.0E-2	7.5E-2	7.1E-2
C	1.0E-1	9.7E-2	5.9E-3	5.5E-3	1.2E-1	1.1E-1	2.3E-1	2.1E-1
D	4.6E-2	4.3E-2	4.1E-3	3.8E-3	6.8E-2	6.5E-2	1.2E-1	1.1E-1
Total	1.9E-1	1.8E-1	1.5E-2	1.4E-2	2.2E-1	2.1E-1	4.3E-1	4.1E-1

4.4.3.4. Public Urban (PU)

For the estimation of both, water usage and wastewater discharge flows in trade activities, an average value from activities grouped in trade activity from the literature (Table 38) was used. Water demand for service activities is a function of a typical municipal water use for public services of $5.5x10^{-7}$ m^3/s per capita (Metcalf and Eddy, 1991), and the total population in urban areas.

Total population in urban areas was obtained per municipality from data census multiplied by the fraction of each municipality corresponding to each section of the aquifer. Total water usage and wastewater discharged from services in each aquifer section is shown in Table 39 for 1990. In terms of wastewater discharge, 80% of water usage was assumed to be converted in wastewater, similarly as in domestic.

Table 38 Typical flow of water and wastewater per customer from trade, 1990

Trade	W-use Range (gal/unit/d)			Average m^3/s	*WW (gal/d*unit)			Average m^3/s
	Min	Max	Average		Min	Max	Average	
Restaurant	8	10	9	3.96E-7	2	4	3.0	1.32E-7
Food take-away	3	8	5.5	2.42E-7	1	5	3.0	1.32E-7
Bar & cocktail	2	4	3	1.32E-7	1	5	3.0	1.32E-7
Cinema	2	4	3	1.32E-7	2	4	3.0	1.32E-7
Department store	8	13	10.5	4.62E-7	8	12	10.0	4.40E-7
Cafeteria	4	10	7	3.08E-7	1	3	2.0	8.80E-8
Coffee shop	15	30	22.5	9.90E-7	4	8	6.0	2.64E-7
Store, retail	5	20	12.5	5.50E-7	1	4	2.7	1.17E-7
Average	5.9	12.4	9.13	4.01E-7	2.5	5.6	4.1	1.80E-7

Source: Metcalf and Eddy, (1991); Tchobanoglous and Schroeder, (1987). *WW: wastewater discharged

Table 39 Water usage and wastewater flows (m^3/s) from service, 1990

Aquifer section	Urban population	Water usage	Wastewater discharge
A	2.09E+04	1.15E-02	9.18E-03
B	1.00E+05	5.50E-02	4.40E-02
C	2.74E+05	1.51E-01	1.21E-01
D	1.41E+05	7.77E-02	6.22E-02
Total	5.36E+05	2.95E-01	2.36E-01

Total water usage for trade activities is a function of the average water usage in trade activities, an average wastewater discharge, and the total number of people in trade activities. Total number of people in trade sector was obtained from the total number of employees per municipality from census data multiplied by the fraction of each municipality corresponded to each section of the aquifer. Total water usage and wastewater from trade are shown in Table 40.

Table 40 Water usage and wastewater flow from (m^3/s) trade, 1990

Aquifer section	Trade employee	Water usage	Wastewater
A	2.51E+02	1.01E-04	4.51E-05
B	5.07E+02	2.04E-04	9.13E-05
C	4.27E+02	1.72E-04	7.69E-05
D	6.40E+02	2.57E-04	1.15E-04
Total	1.83E+03	7.33E-04	3.29E-04

Table 41 Water use from public urban activities (m^3/s), 1990

Activity	A	B	C	D	Total
Service	1.15E-02	5.50E-02	1.51E-01	7.77E-02	2.95E-01
Trade	1.01E-04	2.04E-04	1.72E-04	2.57E-04	7.33E-04
Total	1.16E-02	5.52E-02	1.51E-01	7.80E-02	2.96E-01

Table 42 Wastewater discharge from public urban activities (m^3/s), 1990

Activity	A	B	C	D	Total
Service	9.18E-03	4.40E-02	1.21E-01	6.22E-02	2.36E-01
Trade	4.51E-05	9.13E-05	7.69E-05	1.15E-04	3.29E-04
Total	9.23E-03	4.41E-02	1.21E-01	6.23E-02	2.36E-01

In terms of water usage, the service sector reports a usage for services significantly higher than trade sector in section C and D. This may be due to predominant service activity around Merida (Table 41). Overall, the main water usage from public urban activities in the study area is the service sector, which accounts for a total of 2.9×10^{-1} m^3/s in 1990. It is not the same for wastewater discharge, where sections B, C and D have a significantly higher wastewater discharge in service (Table 42). Even though the wastewater discharges from service activity is 80% and 45% for trade, these does not change the pattern of wastewater discharge for each section. Therefore for Yucatan case study, the main wastewater from public urban activities is discharged from service activities which accounts for a total of 2.36×10^{-1} m^3/s.

4.4.3.5. Domestic Rural (DR)

Table 43 shows the total number of households (HH) per aquifer section together with connection to different treatment facilities (HH1-HH4) per section of Yucatan in 1990, showing that only simple septic tanks are relevant to the rural section.

Table 43 Total rural households by treatment (HH1-HH4), 1990

Aquifer section	Total Rural HH (number)				
	HH1	**HH2**	**HH3**	**HH4**	**Total**
A	1739	0	0	0	1733
B	8332	0	0	0	8332
C	22844	0	0	0	22844
D	11782	0	0	0	11782
Total	44801	0	0	0	44801

HH1 & HH2: households with septic tank and improved septic tank respectively; HH3 & HH4: households connected to wastewater treatment plants and improved wastewater treatment plants respectively.

Table 44 shows total water usage and wastewater discharge of domestic rural in Yucatan in 1990 (details explained in section 4.4.3.1.). Water usage is a function of the per capita water abstraction and total population or total number of households. Data of water abstraction per capita for Mexico, and total population were obtained from Mexican census (OECD, 2009; INEGI, 1990).

Table 44 Water use and wastewater from domestic rural (m^3/s), 1990

Aquifer section	Total HH (number)	W-use (m3/s)	WW (m3/s)
A	1733	2.93E-02	2.35E-02
B	8332	1.41E-01	1.13E-01
C	22844	3.89E-01	3.11E-01
D	11782	2.49E-01	1.99E-01
Total	44801	8.08E-01	6.46E-01

HH: Household; W-use: Water usage; WW: Wastewater discharge

4.4.3.6. Agriculture (AGR)

The agriculture sub-model comprises two groups: vegetables and fruits, which are representative of the main agriculture in the study area (INEGI, 2010). Vegetables include: tomato, maize, cucumber, habanero and green chilli, zucchini, avocado, and henequen. Fruits group includes: watermelon, orange, lemon, mamey, coco fruit, papaya, and other fruits. In order to estimate water usage and wastewater discharge from agriculture activity, it is important to consider fundamental principles based on FAO, (1986). Crop water utilization is defined as “the amount of water needed to meet the water loss through evapotranspiration (Eo), and to grow in optimal conditions”.

In order to derive crop water utilization, there are three environmental parameters that need to be considered: climate, crop type, and growth stage of the crop. The influence of the last two parameters is reported as crop factor (Kc).

- Climate influence on crop water needs (Eo) - it is given by the reference crop evapotranspiration (Eo) used by FAO (grass), expressed in millimetres per unit of time (i.e. mm/day; mm/month; mm/season). The rate of Eo is determinate from a large area, by green grass 8 to 15cm tall, which grows completely in shades and with no shortage of water; either by experimental evaporation (such as the pan method) or by measured climate data (such as the Blaney-Criddle method).
- Crop type influence on the crop water needs (Kc) – it depends on three factors: the type of crop, the growth stage of the crop and the climate. In order to determinate Kc, there are three steps: 1. Determine total growing period of each crop is needed; 2. Determine the various growth stages of each crop; and 3. Determine the Kc value for each crop at each stage of growth or ETo (estimated above). Typical values of total growing period and growth stages for various crops are documented by FAO, (1986). All these values need to be considered.

Table 45 Water utilization efficiency per crop (kg/m^3)

Crop	Type of crop	Crop water utilization efficiency (kg/m^3)		
		min	max	average
Tomato*	vegetables	10	12	11
Cabbage*	vegetables	12	20	16
Onion*	vegetables	8	10	9
Pepper*	vegetables	1.5	3	2.25
Maize	vegetables	0.8	1.6	1.2
Pasture	vegetables	1.5	2	1.75
Watermelon*	fruit	5	8	6.5
Pineapple*	fruit	8	12	10
Banana*	fruit	2.5	4	3.25
Citrus*	fruit	2	5	3.5
Vegetable	vegetables	7.9	11.3	9.6
Fruits	fruit	4.4	7.3	5.8

* Vegetables and fruits used to estimate average for each type of crop. Maize was estimated with its original value.

From the above explained, FAO, (2013) have reported the water utilization efficiency for harvested yield (Ey) for specific crops (Table 45). Water utilization efficiency (Y/ET) is defined as yield of plant product (tonnes of crop, Y) per unit of crop water use (litres of water lost by evapotranspiration, ET) (Atwell et al., 1999). Table 46 shows agriculture production by crop type (SAGARPA, 2010).

Table 46 Total crop production (m^3/s) in Yucatan in 2005

Crops	Type	Total production (ton)			
		A	B	C	D
Tomato	vegetables	0.0E+00	4.5E+02	1.5E+03	1.1E+03
Maize	vegetables	7.1E+03	1.0E+03	0.0E+00	0.0E+00
Cucumber	vegetables	8.4E+01	3.0E+02	3.4E+02	3.4E+02
Habanero chilli	vegetables	0.0E+00	2.6E+02	3.9E+02	2.4E+02
Green chilli	vegetables	6.3E+01	0.0E+00	4.0E+01	0.0E+00
Zucchini	vegetables	0.0E+00	4.1E+02	6.4E+02	4.4E+02
Vegetables	vegetables	0.0E+00	3.7E+02	4.3E+02	2.9E+02
Avocado	vegetables	0.0E+00	2.7E+02	4.8E+02	3.2E+02
Henequen	vegetables	1.2E+02	8.3E+02	0.0E+00	0.0E+00
Watermelon	fruit	6.3E+02	6.8E+02	3.0E+02	3.0E+02
Mamey	fruit	0.0E+00	2.7E+03	2.9E+03	0.0E+00
Coco fruit	fruit	0.0E+00	2.4E+02	4.0E+02	0.0E+00
Papaya	fruit	0.0E+00	2.3E+02	2.3E+02	2.3E+02
Orange	fruit	0.0E+00	1.8E+02	3.7E+02	3.2E+03
Lemon	fruit	3.4E+02	2.0E+03	2.0E+03	1.6E+03
Other citrus	fruit	0.0E+00	1.1E+03	1.6E+03	1.3E+03
Other fruits	fruit	0.0E+00	5.5E+02	7.6E+02	5.5E+02
Total fruits	fruit	9.7E+02	7.7E+03	8.5E+03	7.1E+03
Total Veg	vegetables	7.4E+03	3.9E+03	3.9E+03	2.8E+03

In order to derive the initial data input for the model, which starts simulating in 1990, agriculture production growth was assumed to follow population growth (average population growth rate of 1.74%/y). Then agriculture production was extrapolated back to 1990, and the results are given in Table 47.Table 48 is grouping crop production from Table 46 into the three groups: vegetables, fruits and maize. Maize was classified separately since it is a major crop in the area but its value for water utilization efficiency is much lower than the vegetable value.

Table 47 Agriculture production (Tons), 1990

Crops	Type	Total production			
		A	B	C	D
Tomato	vegetables	0.0E+00	3.5E+02	1.2E+03	8.7E+02
Maize	vegetables	5.5E+03	7.8E+02	0.0E+00	0.0E+00
Cucumber	vegetables	6.5E+01	2.3E+02	2.6E+02	2.6E+02
Chile habanero	vegetables	0.0E+00	2.0E+02	3.0E+02	1.8E+02
Chile verde	vegetables	4.8E+01	0.0E+00	3.1E+01	0.0E+00
Zucchini	vegetables	0.0E+00	3.1E+02	4.9E+02	3.4E+02
Vegetables	vegetables	0.0E+00	2.8E+02	3.3E+02	2.2E+02
Avocado	vegetables	0.0E+00	2.1E+02	3.7E+02	2.5E+02
Henequen	vegetables	9.6E+01	6.4E+02	0.0E+00	0.0E+00
Watermelon	fruit	4.9E+02	5.2E+02	2.3E+02	2.3E+02
Orange	fruit	0.0E+00	2.1E+03	2.2E+03	0.0E+00
Lemon	fruit	0.0E+00	1.9E+02	3.0E+02	0.0E+00
Mamey	fruit	0.0E+00	1.7E+02	1.7E+02	1.7E+02
Coco fruit	fruit	0.0E+00	1.4E+02	2.9E+02	2.5E+03
Citrus	fruit	2.6E+02	1.6E+03	1.5E+03	1.2E+03
Other fruits	fruit	0.0E+00	8.5E+02	1.2E+03	9.7E+02
Papaya	fruit	0.0E+00	4.2E+02	5.8E+02	4.2E+02
Total fruits	fruit	7.5E+02	5.9E+03	6.5E+03	5.5E+03
Total Veg	vegetables	5.7E+03	3.0E+03	3.0E+03	2.1E+03
Total Production	Veg + Fruit	6.4E+03	8.9E+03	9.5E+03	7.6E+03

* Total veg is estimates without maize. Total Production includes: vegetables, fruits, and maize.

Table 48 Total agriculture production (Tons), 1990

Crops	A	B	C	D	Total production
Vegetables	2.1E+02	2.2E+03	3.0E+03	2.1E+03	7.5E+03
Fruit	7.5E+02	5.9E+03	6.5E+03	5.5E+03	1.9E+04
Maize	5.5E+03	7.8E+02	0.0E+00	0.0E+00	6.3E+03
Total	6.4E+03	8.9E+03	9.5E+03	7.6E+03	3.2E+04

In 2013, Yucatan government reported a total water usage in the Metropolitan Area of Merida (MAM) for agriculture of 3.71×10^7 m^3/y (1.2 m^3/s), which corresponds to a total area of 480 km^2 (Yucatan Government, 2013). The total agriculture area of MAM covered by the four sections of this study is 595 km^2. Therefore the total water usage for agriculture corresponding to the four sections was extrapolated to 1.5 m^3/s in 2013. The wastewater estimation was obtained considering the following assumptions:

- Water needed for the crop (water utilization by crop, Table 45) is less than the water irrigation volume reported, therefore, the remaining of the irrigation volume in Yucatan is considered as wastewater discharged to the aquifer (irrigation – crop water utilisation = wastewater).

- As there is no wastewater treatment infrastructure placed in agriculture fields, total wastewater reported will be considered untreated, which means the total load of pollutants (chemical and microbial) will reach the aquifer.

By combining the data of agriculture production per crop type in 1990 (Table 47) with the data of water utilization per crop type (Table 45), the water utilization per crop type and aquifer section were estimated and reported in Table 49, with a total crop water utilization of 0.29 m^3/s in 1990 for the study area.

Table 49 Water utilization per crops (m^3/s), 1990

Crops	A	B	C	D	Total
Vegetables	6.9E-04	7.3E-03	9.8E-03	7.0E-03	2.5E-02
Fruit	4.1E-03	3.2E-02	3.6E-02	3.0E-02	1.0E-01
Maize	1.4E-01	2.1E-02	0.0E+00	0.0E+00	1.7E-01
Total	1.5E-01	6.0E-02	4.5E-02	3.7E-02	2.9E-01

By extrapolating back based on population growth the total irrigation water use for agriculture from 2013 to the year 1990, a value 0.79 m^3/s is obtained for the study area (Table 50).

The corresponding wastewater discharge, as given in Table 51, is derived by “subtracting” Irrigation – crop water utilization = Wastewater. The total wastewater discharge in the study corresponds to 0.5 m^3/s, i.e. 63% of the irrigation water usage.

Table 50 Water use in agricultural Irrigation (m^3/s), 1990

Crops	A	B	C	D	Total
Vegetables	1.9E-03	2.0E-02	2.6E-02	1.9E-02	6.7E-02
Fruit	1.1E-02	8.7E-02	9.6E-02	8.1E-02	2.8E-01
Maize	3.9E-01	5.6E-02	0.0E+00	0.0E+00	4.5E-01
Total	4.0E-01	1.6E-01	1.2E-01	1.0E-01	7.9E-01

Table 51 Wastewater discharge from agriculture (m^3/s), 1990

Crops	A	B	C	D	Total
Vegetables	1.17E-03	1.25E-02	1.67E-02	1.19E-02	4.22E-02
Fruit	6.94E-03	5.51E-02	6.07E-02	5.10E-02	1.74E-01
Maize	2.47E-01	3.50E-02	0.00E+00	0.00E+00	2.82E-01
Total	2.5E-01	1.0E-01	7.7E-02	6.3E-02	5.0E-01

4.4.3.7. Aquaculture (AQU)

This sub-model comprises two predominant aquaculture activities in Yucatan State which are classified in two groups: “shrimp”, and “fish and others”; the latter is mainly represented by tilapia (*Oreochromis niloticus*) and ornamental fish; based on their volume of production.

An important reason for considering aquaculture activity as sub-model for this research was its local and extended impact on the water quality along the 1500 km of the coastal area in the Yucatan Peninsula. A monitoring program was conducted in 1999 in four coastal cities at the north of Yucatan (Progreso, Dzilam, Sisal and Celestun). The aim of this study was to identify anthropogenic impact in terms of water quality. This study reported the municipalities of Sisal and Progreso had the worst water quality along the coastal area of Yucatan in terms of ammonium, nitrate, nitrite, phosphate, silicate, and chlorophyll-a. Impacting factors identified were the shrimp farm effluents from aquaculture activity, harbor effluents, and inadequate wastewater disposal. Thus, these issues should be integrated in the water management for Yucatan (Herrera-Silveira et al., 2004).

Potential sources of nitrate were identified from shrimp ponds, which are fertilized with nitrate and silica to favour the growth of diatoms. The highest concentrations of salinity, nutrients and chlorophyll-a were related to shrimp farms and harbor effluents. Ammonium, nitrite and phosphate were related to anthropogenic impact from tourism due to its seasonal variation (higher during the summer).

Overall aquaculture activity within the study area of this research is small. In Yucatan, there were in total 51 aquaculture production units (aquaculture facilities) in 2013, which are located in 23 municipalities. Seven of these municipalities are within the study area of this research, which in total account for 21 aquaculture production units. These are show in Table 52, were they are classified in the two aquaculture groups mentioned above.

Table 52 Sections of the case study with aquaculture activity

Municipality	Aquifer section	Group
Hunucma	C and D	Shrimp, fish and others*
Kanasin	B and C	Shrimp, fish and others*
Merida	B, C and D	Shrimp, fish and others*
Mococha	C	Shrimp, fish and others*
Progreso	D	Fish and others
Seye	B	Fish and others
Uman	C	Fish and others

*Fish and others: includes tilapia and ornamental fish

Due to the current small presence of aquaculture in Yucatan, it contributes a relatively small water usage compared to other activities such as agriculture or livestock. It is important to quantify this activity due to the fact that surface water is not available and that the waters around the peninsula are well endowed with marine life. Therefore, the karstic aquifer underneath the study area could be endangered if there is not adequate management of these effluents.

Some studies in high densities of fish farms in Chile, Scotland, Mediterranean, and the Kingdom of Norway reported little risk of regional eutrophication in coastal water with good water exchange (Soto and Norambuena, 2004; Gowen and Ezzi, 1994; Pitta et al., 2006; Husa et al, 2010). Moreover, other studies in the salmon farming at the north of Norway have found, based on monitoring of water transport and typical nitrogen and phosphorous along coastal areas, that the release of nutrients from aquaculture has an insignificant effect in coastal waters (Aure et al., 2010).

In 2006 Yucatan authorities reported a total aquaculture water use of $1.73x10^5$ m^3, which was considered as "non-consumed" (SEMARNAT, 2006). It was therefore assumed that hundred per cent of this water usage turns into wastewater. Water usage in aquaculture is a function of the total production per year and average water usage per product. Average water usage for fish (tilapia) reported in the literature as "water use efficiency for tilapia" is 2.9 m^3 per kg of fish produced (range from 2.7m^3/kg to 3.1m^3/kg), and shrimp 0.75 m^3/ton (Boyd, 2007; Van der Heijden, 2012).

Table 53 Aquaculture production (Kg), 1990

Activities	A	B	C	D	Total
Fish and others	0.00E+00	2.65E+03	1.08E+04	5.28E+03	1.87E+04
Shrimp	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total	0.00E+00	2.65E+03	1.08E+04	5.28E+03	1.87E+04

Total aquaculture production per section in 1990 is reported in Table 53. Water usage for the same year is reported in Table 54, and wastewater discharge is the same, assuming that total water usage is returned as wastewater.

Table 54 Water use in aquaculture (m^3/s), 1990

Activities	A	B	C	D	Total
Fish and others	0.00E+00	2.44E-04	9.91E-04	4.86E-04	1.72E-03
Shrimp	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total	0.00E+00	2.44E-04	9.91E-04	4.86E-04	1.72E-03

4.4.3.8. Livestock (LIV)

This sub-model includes porcine, bovine, poultry (chicken and turkey), and ovine activities, which were selected based on the representative livestock production for Yucatan reported over the last two decades (SAGARPA, 2005). It is important to consider this activity as part of the model due to its water quality impact. Table 55 shows water demand for specific livestock reported in the literature. For the present research this corresponds to the blue water portion (surface and groundwater) of the water footprint (WFP) of these products.

Table 55 Specific water demand per livestock product for Mexico

Product	Average water demand (m^3/Ton meat)[1]	Type of production
Bovine	157	Grazing
Porcine	602	Industrial
Poultry	305	Industrial
Ovine	276	Grazing

[1]blue water portion of the water footprint of the product produced in Mexico. Source: Melkonnen & Hoekstra, 2010; Doreau, Corson, and Wiedemann, 2012.

WFP is considered the main method to assess water use in terms of green, blue and grey water. Blue water is the most direct measurement that represents the drinking water and the services required for the animal growth (Melkonnen & Hoekstra, 2010;

Doreau, Corson, and Wiedemann, 2012). In this study, livestock blue water reported for Mexico was used as average water usage to estimate livestock water use.

Yucatan State is classified in three regions in terms of agriculture and livestock activities. The northeast is the largest livestock area; all south is corn production area and the northwest was the henequen region, nevertheless henequen production has declined and now porcine and poultry are the predominant activities. In Yucatan, the largest area of bovine activity is located at the northeast portion, as well as poultry and swine industry in the MAM which have flourished in the last two decades. Therefore, the study area of this research has an important risk of groundwater contamination by livestock wastewater (INIFAP, 2010).

In terms of economic growth, swine farming has a recent growth, from which about 25 thousand people depend directly or indirectly since 2006. In addition, local consumption of pork meat in Yucatan is among the highest in Mexico, therefore the production is expected to increase mainly with the demand of increasing population (SEMARNAT, 2006). Livestock production in 1990 corresponding to the four sections of the study area is reported in Table 56 (INEGI, 1990).

Table 56 Livestock production (Tons), 1990

Aquifer section	Bovine	Ovine	Porcine	Poultry	Total
A	4.23E+02	1.58E+01	4.84E+02	2.01E+03	2.94E+03
B	3.09E+03	1.68E+01	2.87E+04	1.10E+04	4.27E+04
C	3.19E+03	1.76E+01	3.10E+04	2.47E+04	5.89E+04
D	3.02E+03	5.41E+00	3.00E+04	1.20E+04	4.50E+04
Total	9.73E+03	5.57E+01	9.01E+04	4.97E+04	1.50E+05

Total water consumption of 10 hm^3 per year (0.32 m^3/s) for the 300 swine farming in Yucatan and total wastewater discharge of 7 hm^3 were reported in 2006 (SEMARNAT, 2006). It was assumed that the 70% of water usage is converted to wastewater. In this way, using production values from Table 56, data were derived for water usage and wastewater production by livestock activity as shown in Table 57 and Table 58.

Table 57 Water use in m^3/s from livestock, 1990

Aquifer section	Bovine	Ovine	Porcine	Poultry	Total
A	2.11E-03	1.39E-04	9.23E-03	1.95E-02	3.10E-02
B	1.54E-02	1.47E-04	5.47E-01	1.06E-01	6.69E-01
C	1.59E-02	1.54E-04	5.91E-01	2.39E-01	8.46E-01
D	1.50E-02	4.74E-05	5.72E-01	1.16E-01	7.04E-01
Total	4.84E-02	4.87E-04	1.72E+00	4.81E-01	2.25E+00

Table 58 Wastewater discharge in m^3/s from livestock, 1990

Aquifer section	Bovine	Ovine	Porcine	Poultry	Total
A	1.48E-03	9.71E-05	6.46E-03	1.36E-02	2.17E-02
B	1.08E-02	1.03E-04	3.83E-01	7.42E-02	4.68E-01
C	1.11E-02	1.08E-04	4.14E-01	1.67E-01	5.92E-01
D	1.05E-02	3.32E-05	4.00E-01	8.15E-02	4.93E-01
Total	3.39E-02	3.41E-04	1.20E+00	3.36E-01	1.57E+00

4.5. Summary of model settings and data input

Table 59 summarises the data for water usage and wastewater release, mostly derived from literature information on typical water usage for the specific activity or sub-module, respectively estimated in this chapter. These data were extrapolated in Vensim to the starting year 1990 and used for the simulations.

Table 59 Data input for the Sustainable Integrated Water Management Model (SIWMM) in 1990 for full study area

Sub-module	Variable (units)	Typical Water usage		W-usage	WW
		(Reported unit)	(Converted unit)	(flow unit)	(flow/unit)
Agriculture	**unit**	m^3/Ton	m^3/s *Ton	m^3/s	m^3/s
	Vegetable	104.2	3.3E-06	6.7E-02	4.3E-02
	Fruits	172.4	5.5E-06	2.8E-01	1.8E-01
	Maize	833.3	2.6E-05	4.5E-01	2.8E-01
	Total			8.0E-01	5.0E-01
Aquaculture	**unit**	m^3/kg	m^3/s * kg	m^3/s	m^3/s
	Shrimp	750	2.4E-05	0.E+00	0.0E+00
	Fish & others	2.9	9.2E-08	1.7E-03	1.7E-03
	Total			1.7E-03	1.7E-03
Livestock	**unit**	m^3/Ton	m^3/s *Ton	m^3/s	m^3/s
	Porcine	602	1.9E-05	1.7E+00	1.2E+00
	Bovine	157	4.9E-06	4.8E-02	3.4E-02
	Poultry	305	9.6E-06	4.8E-01	3.4E-01
	Ovine	276	8.7E-06	4.9E-04	3.4E-04
	Total			2.2E+00	1.6E+00
Industry	**unit**	GED	m^3/s *empl	m^3/s	m^3/s
	Construction	31	1.3E-06	6.4E-03	5.5E-03
	Manufacture	817	3.5E-05	2.8E-01	2.4E-01
	Total			2.9E-01	2.4E-01
Institutions	**unit**	L/day*people	m^3/s *people	m^3/s	m^3/s
	Hospitals	741	8.5E-06	1.5E-02	1.4E-02
	Hotels	190	2.1E-06	2.2E-01	2.1E-01
	Office & School	53.2	6.1E-07	6.2E-02	5.1E-02
	Total			4.6E-01	4.1E-01
Public Urban	**unit**	G/day*people	m^3/s*people	m^3/s	m^3/s
	Service	12.5	5.4E-07	2.5E-02	2.0E-02
	Trade	9.13	4.0E-07	7.3E-04	4.4E-04
	Total			2.6E-02	2.0E-02
Domestic Urban	**unit**	G/day*people	m^3/s*people	m^3/s	m^3/s
	HH1	366	4.2E-06	8.8E-02	7.1E-02
	HH2	366	4.2E-06	4.2E-01	3.4E-01
	HH3	366	4.2E-06	1.2E+00	9.3E-01
	HH4	366	4.2E-06	6.0E-01	4.8E-01
	Total			2.3E+00	1.8E+00
Domestic Rural	**unit**	L/d*people	m^3/s*people	m^3/s	m^3/s
	HH1	366	4.2E-06	2.9E-02	2.4E-02
	HH2	366	4.2E-06	1.4E-01	1.1E-01
	HH3	366	4.2E-06	3.9E-01	3.1E-01
	HH4	366	4.2E-06	2.0E-01	1.6E-01
	Total			7.6E-01	6.1E-01

HH1 & HH2: households with septic tank and improved septic tank respectively; HH3 & HH4: households connected to wastewater treatment plants and improved wastewater treatment plants respectively.

All data for the 8 socio-economic activities (8 sub-models) including agriculture, livestock, aquaculture and industry sub-models were estimated based on statistical data and production records at aquifer section scale. Based on 20 years period of data input (1990-2010) together with the mathematical equations defined throughout this chapter, Vensim estimates current and future water demand and wastewater production in term of water quality (Q), using for instance nitrate (in mg/m^3) or faecal coliforms (in CFU/m^3) and in terms of water quantity (F) using flows (in m^3/s), as illustrated in the causal-loop of the SD model developed for the MAM case study (see Figure 21).

In general, values of flows are given in m^3/s, and pollutant concentrations (see Chapter 5) are given in CFU/m^3 or mg/m^3 for FC or NO_3 respectively, so that pollutant loads in CFU/s or mg/s for FC or NO_3 respectively van be derived. Then a series of equations are declared in Vensim to interconnect the data input of water flow (including groundwater flow, rainfall, water abstraction and wastewater release) and pollutant concentrations through mathematical equations. These equations allow Vensim to simulate pollutants concentration over 50 years of simulation period.

Population growth is used as the linking dynamic parameter to forecast the future development of water abstraction and wastewater release (and consequently pollutant loads), in particular since the majority of the social and economic production is related to local consumption. In the near future, however, livestock and in particular porcine activity could become an external factor along with increasing export sales.

Concerning the estimation of pollutant loads, there are other methods than the used in this thesis (compare Chapter 5) as discussed by Benham et al., (2006). For instance, for bacteria loads Bacteria Source Load Calculator (BSLC) by Zeckoski et al., (2005) and Bacteria Indicator Tool (BIT) by USEPA (2000) methods might be used. Other indirect methods to estimate pollutants loads are through technical guidelines (e.g., USEPA, 2012c) as described by Niu and Phanikumar, (2015), which could lead to significant uncertainty associated to specific management practices and pollutant stage of change for each socioeconomic activities. For instance, between storage and application of manure on land (DeGuise and Mostaghimi, 2000) reported a decrease in FC of two orders of magnitude but also a re-growth of bacteria during field application has been observed (Crane et al., 1980; Wang et al., 2004). For the present thesis, a significant amount of local data was available, mainly from historic data records, together with data from local field research collected. Thus pollutant loads in this research were largely estimated on the basis of existing data, with some years of data inconsistency that were extrapolated.

Chapter 5. Methodology for modelling pollutants

“Most modellers would argue that the primary benefits of modelling are to bring clarity of thought and to integrate all the information about one question. Models can also be used to quantify processes and effects, make predictions and give some understanding of uncertainty, but these benefits are less easy to realise…modelling tools are to help analyse many aspects of the groundwater….but not all” (UKEA, 2009).

This chapter includes the pollutant’s selection, pollutant’s sources, and specific implications for model inputs with assumptions and parameter settings for each of the two pollutants modelled: Faecal coliforms (FC), and nitrate (NO_3). It also present a summary of the interventions proposed in this study together with the description of the cost-benefit analysis developed for the selection of the most suitable intervention proposed in this research.

5.1. Pollutants selection: Nitrate and faecal coliform

Groundwater pollutants in Yucatan aquifer and associated diseases have already been discussed in chapter 3. Nitrate (NO_3) and faecal coliforms (FC) were selected for the SIWM model because of their impact for public health.

Nitrate is a chemical indicator of water quality and an example of a conservative pollutant, i.e. nitrate may remain mainly unchanged in groundwater for decades and is not essentially degraded during wastewater treatment processes but requires specific removal techniques (such as ion exchange or biological denitrification). The latter accounts in particular for the oxygen rich groundwater of the Yucatan aquifer which is expected to prevent anoxic denitrification processes. Tracing the fate of a conservative pollutant could help to understand the long-term effects of pollution of the natural aquifer resource by anthropogenic activities in the study area.

In terms of public health, excessive concentration of nitrate in water can cause cyanosis (also called blue babies’ syndrome or methemoglobinemia) in children under 5 years due to reduction of nitrate to nitrite in the upper digestive tract of infants (Zaporozec, 1983). Some studies have associated nitrate with hypertension, and as a consequence of chemical reaction to nitrosamines also with cancer and premature diabetes in adults (Pacheco and Cabrera, 2013; Parslow et al., 1997; Comly, 1987; Hill et al., 1973). In addition, it has been reported as a cause of recurrent stomatitis (Gupta et al., 1999). Nitrate analysis of selected deep water supply wells of all 106 Yucatan municipalities revealed that in 2007-2008, 25 municipalities had higher nitrate concentrations than the allowed maximum of 45 mg/L (Osorio, 2009).

Faecal coliform (FC) are a microbial indicator of water quality and represent the non-conservative pollutants, which are characterized by a relatively fast decay and often effective removal by a variety of wastewater treatment options. Faecal coliform bacteria are a subgroup of total coliforms that traditionally have been associated with faecal contamination. Faecal coliform in water are not essentially harmful to human, but indicate a higher risk of pathogens being present in the water and causing various diseases, in particular acute diarrhea.

The MAM is located in the north west of Yucatan, where Dohering and Buttler, (1974) reported that by that time, more than 40% of deaths in children under 6 years were attributed to gastrointestinal diseases from waterborne pathogens.

In 1995 a collaborative study carried out by BGS et al., found contamination with faecal coliforms (FC) exceeding 1000MPN/100ml in shallow wells (at a depths close to the water table) beneath Merida city. High surface concentration of FC entails a significant contamination risk also for deeper parts of the aquifer. Osorio, (2009) reported an average FC concentration of 43.2 CFU/100ml ranging from 0 to 470 CFU/100ml in 106 deep water supply wells of the Yucatan municipalities.

MPN (Most Probable Number) and CFU (Colony Forming Units) are considered as equivalent counts for FC (see section 5.3.3.). Coliform concentrations from cited references up to latest 90's, were reported in MPN whereas CFU has been used more recently. MPN procedure using multiple tube fermentation is more variable than CFU using membrane filtration as a result of the probabilistic basis for calculating the MPN as documented by Gronewold and Wolpert, (2008). Therefore, international standard methods of water and wastewater for microbiological examination have shifted from MPN to CFU.

5.2. Nitrate

5.2.1. Sources of nitrate

In general, agriculture (fertilizers) and livestock (animal waste) have been considered as the largest anthropogenic sources of nitrate in groundwater (Zaporozec, 1983). Domestic sewerage systems such as septic tanks are documented as the major non-agricultural source of nitrate in groundwater (Rao et al., 2013). A significant portion of groundwater nitrate has not been released as such, but is formed from ammonia-nitrogen or organic nitrogen by bacterial nitrification (Figure 26). Other examples of pointed nitrogen sources are industrial discharges, waste-disposal sites and manure pits. Examples of non-point or diffuse nitrogen sources also include: natural background in groundwater and atmospheric deposition (Deshmukh, 2012; Zaporozec, 1983; Ballester et al., 2001; Velazco et al., 2009; Panno et al., 2001).

In Yucatan, nitrate in groundwater was considered an issue of rural areas because of its well-known association to agricultural runoff of nitrogen-containing fertilizers (Pacheco et al., 1997; Pacheco and Cabrera, 2003). Nevertheless, high concentrations of nitrate in urban areas have been found in the last two decades, which are above Maximum Contaminant Level (MCL). These have been attributed to sewerage failures and wastewater infiltration (CONAGUA, 2010a). Specifically, Merida city has a massive discharge of septic tanks to the aquifer (CONAGUA, 2014; BGS et al., 1995), resulting in high punctual nitrate concentrations in and near the city, often exceeding at least in the upper part of the aquifer the MCL, which is 45mg/l established by Mexican regulation in line with WHO guidelines (Pacheco et al., 1997).

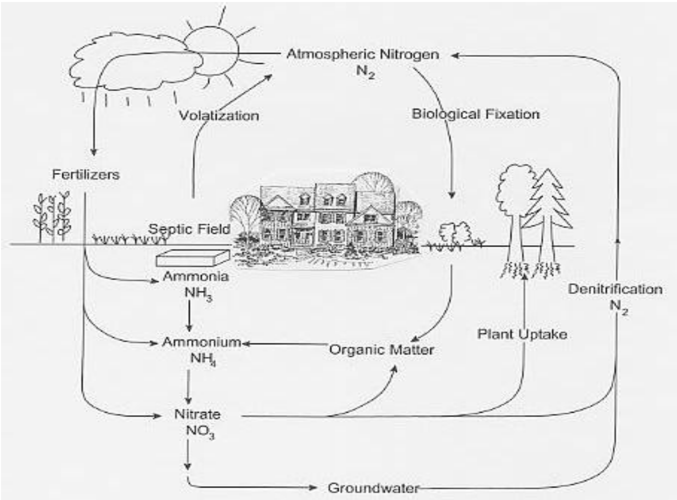


Figure 26 Sources of nitrate in groundwater and nitrogen cycle (Taylor, 2003)

5.2.2. Spatial variability of nitrate in groundwater

In karstic aquifers, depth dependent nitrate concentration variability has been documented (Marin et al., 2000; Bakhsh et al., 2005). Nitrate varies from shallow to deep groundwater wells. Generally, nitrate concentrations tend to be higher in shallow groundwater than in deep groundwater (Ohou, 2008; Torres, 2010). It is because there is less transport distance from the origin sources (i.e. septic tanks, fertilizers from agriculture) and consequently less dilution (Nolan et al., 1998). For example, in a study of the Chalk aquifer in Hampshire, UK, nitrate concentration at shallow depths (around 12m) was found around 60mg/l, while in about 26m depth, concentrations are decreased by half (Stuart et al., 2009). For the Yucatan karst aquifer, similar observations were reported, with 1.8 fold nitrate concentration in shallow wells (near surface) compared to deep wells (40 m) (Osorio, 2009).

It can be important to identify and locate potential nitrate sources (e.g. septic tanks) and the distance between them, in order to improve natural attenuation of pollutant by the aquifer system itself. For example, in the context of septic tanks regulation, the Pennsylvania Department of Environment Protection (PADEP) suggest a minimum area of 1.4 acres by septic tanks to locally ensure sufficient nitrate dilution (Taylor, 2003; Almasri, 2007).

5.2.3. Temporal variability of nitrate in groundwater

An example for the seasonal fluctuation of nitrate in groundwater is documented by Chilton and Foster, (1991) for the Chalk aquifer in the UK (Figure 27). Various factors may contribute to the fluctuations such as time of fertilizer application, plant uptake of fertilizer in summer (growing season), and enhanced leaching due to higher rainfall in autumn and winter months (Taylor, 2003; Deshmukh, 2012; IGME, 2002). Seasonal variation of nitrate in areas with distinct wet and dry seasons has been generally reported as higher in wet periods and lower in dry periods (Wall et al., 1998; Chiroma et al., 2007; Cidu and Biddau, 2012; Hayden, 2012).

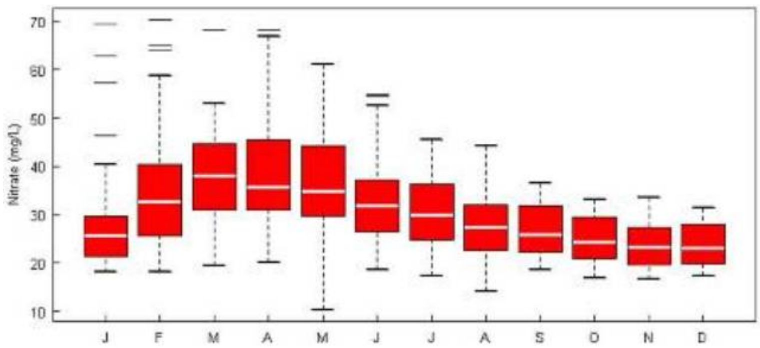


Figure 27 Seasonal fluctuation of nitrate in Chalk groundwater aquifer of the UK. Source: Chilton and Foster, (1991)

A time-series data study held by BGS in 2006 in the UK (Stuart and Chilton, 2007), to define past trends in order to estimate future nitrate concentration a linear increase of nitrate was derived from three different linear regression methods (ordinary least squares-OLS; robust linear regression; and KT-Sen slope – a non-parametric test. In spite of the significant temporal fluctuation of concentration, a linear trend of increasing nitrate is found, in average at 0.34mg NO_3/l annually, which is in line with the general trend in Europe of about 0.4 mg nitrate increase annually, reported by European Environmental Agency in 1999 and other studies (Beeson and Cook, 2004).

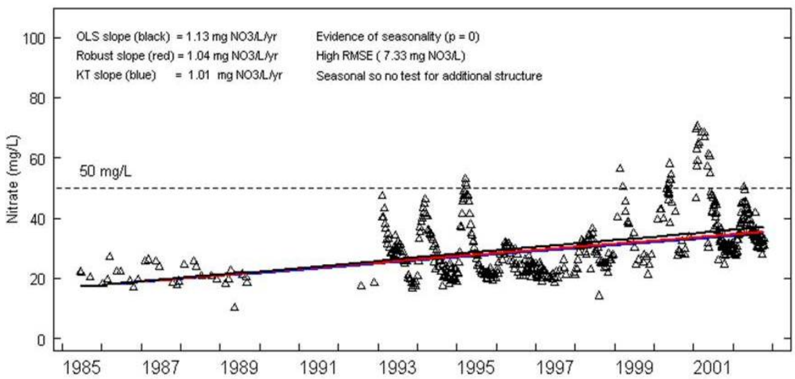


Figure 28 Long-term fluctuations of nitrate concentrations in boreholes Chalk. Source: Stuart and Chilton, (2007)

The irregular fluctuations observed in Figure 28 depend on three main factors: land use changes, i.e. fertilizer application; hydrogeological conditions i.e. groundwater flow direction, and local groundwater pumping patterns (Stuart and Chilton, 2007). Another example in Valencia Spain by Ballester et al., (2001), documented historic records of nitrate concentration with significant increases in groundwater over time at the north of Valencia: 19 mg/l in 1968; 60mg/l in 1984; 84mg/l in 1995; and 104 mg/l in 2000. The latest concentrations are two or three times above the Maximum Contaminant Level for drinking water supply of the population (45mg/l).

A variety of field studies for the MAM confirm seasonal fluctuations of nitrate concentration in the karstic groundwater of the MAM (Pacheco et al., 2001; BGS et al., 1995). However, such temporal fluctuations appear to be less pronounced in deep water supply wells; for example, continuous monitoring of nitrate 2000-2008 in selected deep water supply wells of aquifer sections B, C, and D indicates that the nitrate concentration keeps within a relative narrow range (CONAGUAc, 2008).

In spite of significant spatial and temporal variations of nitrate concentration in groundwater, simplified approaches such as a linear regression models for averaged, long term concentration modelling and estimating future trends are often applied, see for example Stuart and Chilton, (2007).

5.2.4. Model input: assumptions and parameters settings

Only the major nitrogen containing wastewater sources are considered for nitrate concentration modelling, these include: Wastewater from urban and rural domestics, wastewater from livestock, and fertilizer-N from agriculture (Table 60). As discussed in Chapter section 5.2.1., a significant portion of nitrate is not present in the wastewater as such, but in the form of nitrogen containing precursor compounds such as ammonia and organic nitrogen.

Table 60 Typical nitrate concentration in wastewater from different activities

Sub-model	Type	Reported values	Reference	NO_3 (mg/l)	NO_3 (mg/m^3)
AQUIFER	Background	10 .5 mg NO_3/l	Torres, (2010)	10.5	1.05E+04
DU	septic tanks	79 mg TKN/l	Castillo et al., (2011)	347.6	3.47E+05
DR	septic tanks	79 mg TKN/l	Castillo et al., (2011)	347.6	3.47E+05
AGR	- Vegetable	136 mg NO_3-N/l	Moratalla et al., (2009)	598	5.98E+05
	- Fruits	136 mg NO_3-N/l	Moratalla et al., (2009)	598	5.98E+05
LIV	- Ovine	52 mg TN/l	Hegg, (1983)	229	2.29E+05
	- Poultry	40.5 mg TN/l	Hegg, (1983)	178	1.78E+05
	- Porcine	131 mg TN/l	Hegg, (1983)	576	5.76E+05
	- Bovine	40 mg TN/l	Hegg, (1983)	176	1.76E+05

TKN: Total Kjeldahl Nitrogen; NO_3-N: nitrate-nitrogen; TN: Total Nitrogen including TKN and NO_3-N.

The concentration of nitrogen in pollutant sources is commonly reported in the literature as TN (Total nitrogen) or TKN (Total Kjeldahl Nitrogen that does not include nitrate and nitrite N, the latter species generally have only insignificant contribution to the total nitrogen content of domestic and livestock wastewater). In agriculture in contrast, nitrate can be a major component of fertilizers.

From the literature, typical concentrations in wastewater (Table 60) were used to model the load of nitrate discharged into each aquifer section of the MAM, considering that the karstic aquifer is intrinsically vulnerable and any discharged nitrogen leaches into the groundwater. The literature values given in mass of N in Table 60 were multiplied by a factor of 4.4 for N transformation to NO_3 (see below), taking into account the mass difference between N (atomic mass 14) and NO_3 (molecular mass 62).

a. *Conversion of nitrogen to nitrate in the aquifer*

Any nitrogen present in the wastewater is assumed to be converted; either in soil or in the groundwater, by microbial oxidation (nitrification) to nitrate that persists in the MAM aquifer. Conversion to nitrate depends on the efficiency of bacterial nitrification and is in general not easy to predict. Total conversion of N to nitrate is a specific assumption for the highly vulnerable, oxygen rich aquifer of the study area, displaying similar oxygen levels in shallow wells and deep boreholes (Torres, 2010). It is also in line with an assumption of (BGS et al., 1995; USEPA, 2005) for the Merida area. The assumption is further confirmed by nitrogen load modelling for the Barton Spring Zone, a karst aquifer in central Texas (Mahler et al., 2011). Over an observation period of 2.5 years, a balance between total nitrogen input by stream recharge and nitrogen load at discharge sites was found. However, the portion of organic + ammonia N is high in recharge water and becomes rather low in deep water supply wells or at discharge sites, indicating extensive conversion to nitrate. Specifically for fertilizer-N application in the MAM area have estimated 44% leaching of total fertilizer N as nitrate into the groundwater (Gonzalez-Herrera et al., 2014).

b. *No attenuation by soil*

Considering the highly permeable soil matrix of the Yucatan karstic aquifer and rapid solute transport on the one hand, and the general high mobility of nitrate on the other hand, sorption or retardation of nitrate by soil was neglected, as suggested for instance by Shamrukh and Abdel-Wahab, (2011). In a south-western Georgia (USA) karst aquifer, no correlation between nitrate concentration and sampling well depth at 0-80 m was observed (Katz et al., 2014).

c. *Homogeneous distribution in the groundwater*

There is a spontaneous and complete mixing between wastewater, recharge water from rainfall, and the aquifer groundwater, which results in a uniform nitrate distribution in the groundwater within an aquifer section. That means, the model averages higher (as detected in the proximity of point sources) and lower local concentrations. This is an approximation with respect to the depth-dependent nitrate gradient observed in deep and shallow water of karstic aquifers, as described in section 5.2.2. It is also a simplification with respect to the lateral concentration gradient of nitrate originating from point sources. On the other hand, the very high hydraulic conductivity of aquifer in the study area, the groundwater flow, and the time scale applied in the simulation (70 years) favour a long-term homogeneous mixing of nitrate by flow and diffusion.

d. Persistence, no denitrification

It is assumed in this research that nitrate is not degraded by bacterial denitrification or other processes, i.e. it has an “infinite” lifetime in the karst groundwater of the study area. Anoxic denitrification is generally believed to be blocked by dissolved oxygen in the groundwater (Tesoriero et al., 2007). In a report covering both shallow and deep wells of the aquifer in the study area, significant dissolved oxygen concentrations between 1.54 and 7.58 mg O_2/L have been reported (Torres, 2010), with higher concentrations in the deep wells.

e. Aquifer nitrate concentration in 1990

Assigning an aquifer nitrate concentration at the beginning of simulation in the year 1990 was not straightforward since suitable data is relatively scarce. Considering groundwater nitrate concentration in both the water already present in all aquifer sections, as well as in the groundwater flowing into section A, a value of 10.5mg/l nitrate was chosen, as given as a depth-independent average of 7 sinkholes within the ring of sinkholes. The latter constitutes both a part of aquifer section A and the contributes much of the groundwater inflow into that section (Perez-Ceballos et al., 2011). For simplicity, this value was selected for initial nitrate concentration in the 4 aquifer sections. It is possible that this underestimates the actual concentration in 1990, since nitrate inflow is expected to have already been substantial at this time. A range of 9-22 mg/L nitrate was reported from 1991 measurements in deeper groundwater in Merida (BGS et al., 1995).

5.3. Faecal coliform

5.3.1. Sources and spatial variability of faecal coliform (FC)

Generally, numerous point (a discharge that comes out of any identifiable conveyance) and non-point sources (discharge that does not come out of an identifiable conveyance) of microbial pollutants have been identified (Figure 29). These include from non-pointed (or diffuse pollution): urban runoff and livestock over extensive areas, agricultural irrigation with primary treated water; and from point sources (or punctual pollution): septic tanks, wastewater discharges, leakage and spills of sewage (Mahler et al., 2000; Sullivan et al., 2005).

Once pollutants are released, many factors could affect microbial survival in groundwater such as light intensity, nutrients availability, pH, dissolved oxygen and temperature (Carter and Knox, 1986). Bacteria survive longer in groundwater than in surface water: at limited light intensity, such as in many of the sinkholes around the MAM, faecal coliforms have been reported to survive up to 170 days and *E. coli* up to 120 days (Kudryavtseva, 1972).

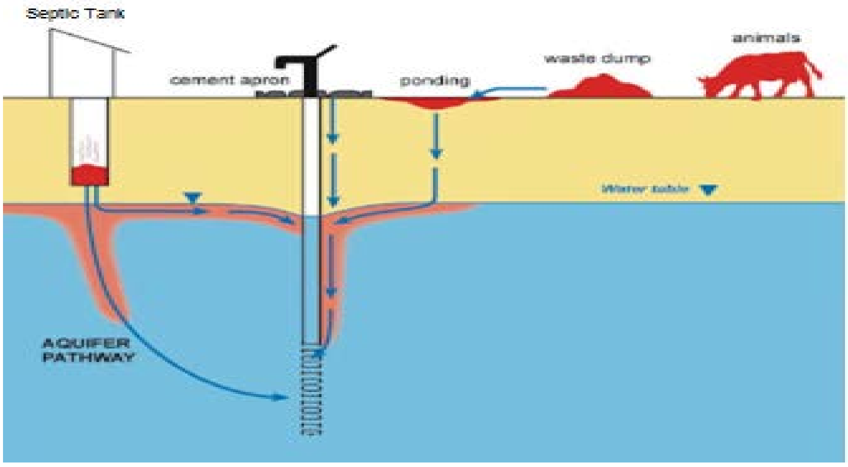


Figure 29 Sources of faecal pollution from rural in groundwater. Source: ARGOSS, (2001)

Groundwater contamination by FC is widespread in Yucatan due to the high water table and porosity of the karst matrix with vertical cracks that facilitate the direct microbial entrance to the aquifer. The study carried out by BGS et al., in 1995 found bacterial contamination, with faecal coliforms (FC) exceeding 1000 CFU/100ml in shallow wells (at a depth close to the water table) beneath Merida city. High surface concentration of FC entails a significant contamination risk also for deeper parts of the aquifer. Osorio, (2009) reported an average FC concentration of 43.2 CFU/100ml ranging from 0 to 470 CFU/100ml in deep water supply wells of the 106 municipalities of Yucatan. Based on the literature data, the main FC sources in the MAM are:

- Septic tanks (ST)

Due to their large number in Merida, the capital city, accounting in total for about 200,000, septic tanks have been identified as one of the main sources of faecal contamination within the MAM (Morris et al., 2003). These septic tanks are neither well-constructed nor well maintained, and therefore facilitate the large discharge of faecal pollution into the aquifer (Lutz et al., 2000).

- Livestock waste

A study conducted by Pacheco et al., (2000a) in a porcine production unit from 1998 to 1999 reported FC from not detected to $1.71\text{x}10^5$ CFU/100ml in shallow wells (5-6.5m deep) within the farm and outside the farm, thus identifying the impact of porcine livestock lixiviation to the groundwater under this area. Within this relatively small area (4 ha) with 100m distance between wells, an increased FC concentration during rainy season was documented. Within the 12 wells monitored, three different concentration levels were identified: two wells with FC concentrations up to $7x10^3$ CFU/100ml; three wells with FC concentrations up to $1.2x10^4$ CFU/100ml; and the other seven wells with FC concentrations up to $2.2x10^4$ CFU/100ml.

FC concentrations in this rural area are several orders of magnitude higher than in the Merida urban environment (BGS et al., 1995). An increased downstream concentration of FC in Merida urban environment due to downstream transport of FC from rural areas in the south of the city was assumed by Marin, (1990). However, unlike in the case of nitrate, a lateral transport of FC over long distances (10th of kilometres) is considered unlikely in view of the rapid die-off and – in spite of very high hydraulic conductivity - the limitation of groundwater flow rate by the very low hydraulic gradient of the area.

5.3.2. Temporal variability of FC in groundwater

Microbial pollution in the Yucatan aquifer may increase significantly during rainy season (June to September), and consequently gastrointestinal diseases increases, thus the rain season is often named diarrhoea season (Lutz et al., 2000; Marin and Perry, 1994). Such increases during heavy rains are attributed to the increased hydraulic loading that spreads pollutants in the aquifer. Pacheco et al., (2000b), confirmed high seasonal variation in a study carried out in 4 small towns in the north of Merida city. Agriculture and particularly porcine activity are predominant in this area. Thus, pollution has been associated to infiltration of these discharges (Pacheco et al*.,* 2004b).

5.3.3. Units of FC concentration

In the literature, FC are reported either as CFU/100ml, meaning colony forming unit per 100ml (most recent studies), or as MPN/100ml, meaning most probable number per 100ml (most historic data). While some studies suggest sophisticated produces for CFU/MPN conversion (Gronewold and Wolpert, 2008; Gronewold et al., 2011), in this research these two units are used as equivalent, as it is widely reported (Foster et al., 2000; WHO, 2001; Jimenez, 2008; USEPA, 2010; Cho et al., 2010).

5.3.4. Decay of FC

In absence of adequate nutrients, FC and other bacteria are generally assumed to decrease or “die-off” at a first-order rate, described by the Chick Law (Benham et al., 2006; Wilkinson et al., 1995; Rosen, 2000):

$$C = C_0 \times 10^{-kt}$$

Where: C= bacterial concentration at time t

C_0= bacterial concentration at time 0

t= time (in days, for example)

k = rate constant for the decay

The crucial parameter in this equation is the k value (decay or die-off rate constant), which determines how fast bacteria will extinguish from groundwater. The half-life is related to k by t_{50} = ln2/k. FC is widely use as indicator microorganism for pathogens of faecal origin in the groundwater. Nevertheless, there are bacteria with shorter and longer lifetime, and significantly longer half-life has been documented for pathogenic viruses in groundwater (Mitchell and Chamberlain, 1978; John and Rose, 2005). Bacterial half-life in groundwater is highly dependent on environmental factors such as temperature and soil characteristics, as explained below.

5.3.5. Model input: assumptions and parameters settings

a. Concentration of faecal coliform in wastewater

While the wastewater flows for the 8 socioeconomic activities have already been determined as described in Chapter 4, in this section the concentration of FC in the wastewater for selected activities will be described. Three main faecal coliform sources were selected in order to estimate the faecal coliform concentration in the aquifer of the MAM, based on the documented FC presence in the MAM. These are: livestock, septic tanks (ST) from domestic urban, and ST from domestic rural.

Table 61 Typical FC concentration by type of wastewater source

Sub-model	Source	Average value reported (CFU/100ml)	Reference	FC (CFU/m^3)
DU	Wastewater	$3x10^5$	Castillo et al., (2011)	$3x10^9$
DR	Wastewater	$3x10^5$	Castillo et al., (2011)	$3x10^9$
LIV	Ovine	$1x10^4$	Estimated[(a)]	$1x10^8$
	Poultry	$1.2x10^3$	USEPA (2012a)	$1.2x10^7$
	Porcine	$3.6x10^5$	USEPA (2012a)	$3.6x10^9$
	Bovine	$6.2x10^5$	USEPA (2012a)	$6.2x10^9$

ST: Septic Tanks; DU: Domestic Urban; DR: Domestic Rural; LIV: Livestock. [(a)] An intermediate value of poultry and porcine for ovine values.

Data per activity were obtained from the literature and are summarized in Table 61. Whenever available, data from local studies were used, otherwise the values refer to national and international data. These values serve as data input for the model, to forecast faecal coliform loads to simulate for the Metropolitan Area of Merida (MAM).

b. Decay rate in groundwater

John and Rose, (2005) have reviewed die-off rates for microorganisms in groundwater and reported a high variability of literature values for coliform bacteria (values given for base 10, range for logarithm of k= 0.007-1.5 log/d, corresponding to k= 0.016-3.4/d, base e). Die-off is not only depending on microorganisms' species but on environmental factors such as temperature, pH, and soil matrix. In addition, even for comparable species and conditions, a significant variability among investigations was observed. John and Rose, (2005) reported from all the studies reviewed a median value of k=0.27/d for coliforms in groundwater at a temperature range 21-37° C.

Data for the in-situ die-off of FC in the karstic matrix under conditions of the Yucatan aquifer were not found in the literature. The die-off constant was therefore derived from literature data for closely related soil-groundwater matrices. For the aquifer under the MAM, water temperatures in shallow and deep wells are between 25 and 30° C, and the pH is around 7 (Torres, 2010). Leal-Bautista et al., (2013) reported a karstic aquifer water temperature of 25° C and pH 6.8, both constant over a depth of 0-38m below water level for the Tulum region within the Yucatan aquifer (but distant from the MAM). Howell et al., (1996), reported for groundwater in the presence of a highly porous soil matrix (sand) at 25° C a die-off rate constant of 0.12/d. The conditions under which this value was determined are close to the MAM conditions. Therefore, k=0.12/d ($1.4x10^{-6}$/s), corresponding to a half-life of 5.8 days, was assumed as the die-off rate constant of faecal coliforms in the MAM aquifer.

c. No attenuation by soil

As already outlined in Chapter 2, microbes can be transported at a faster rate through fractures and conduit flow in karstic limestone than small molecules (such as nitrate) due to size exclusion from smaller pores (Harvey, 1997). A tracer study with 1 µM fluorescent microspheres (simulating bacteria) in a karst conduit system of the Alps revealed 40% recovery over a transport distance of 2.5 km, with peak between 18-83h depending on groundwater flow velocity (Göppert and Goldscheider, 2008). It is therefore assumed those faecal coliforms are not attenuated by soil in the Yucatan aquifer with its very high hydraulic conductivity.

d. Homogeneous distribution in the groundwater

With similar arguments as described above, a homogeneous distribution of faecal coliform in the water of an aquifer section is assumed. FC may well be transported tens of meters and contribute to deeper levels of the aquifer (20-40m) where most of drinking water of the MAM is abstracted. The model does not consider seasonal variations of FC groundwater concentration with rainfall but averages concentration over the year. This scenario may be closer to the real situation in the rain season, with often daily heavy showers, than in the dry season. In the latter, bacteria may die-off close to the point of release before they distribute in the aquifer.

5.4. Summary of engineering interventions

Table 62 summarises the seven potential interventions proposed in this thesis, by specific removal efficiencies for each intervention. Examination of the response to this set of seven interventions aims to improve the overall quality of wastewater discharged to the groundwater. Improvement is defined as the removal of two pollutants: Faecal Coliforms (FC) and Nitrate (NO_3), which have been identified in the literature as two of major concern in the area (see Chapter 3 of case study description). While removal of faecal coliforms is greatly controlled by adequate wastewater treatment from domestic activities, nitrate removal requires several other actions due to the variety and complexity of the sources for this pollutant (i.e. diffuse source from agriculture and livestock activities). For example, reducing nitrogen compounds in wastewater reduces nitrate load since all nitrogen is converted to nitrate by nitrification in the aquifer.

Table 62 Summary of engineering interventions for the MAM

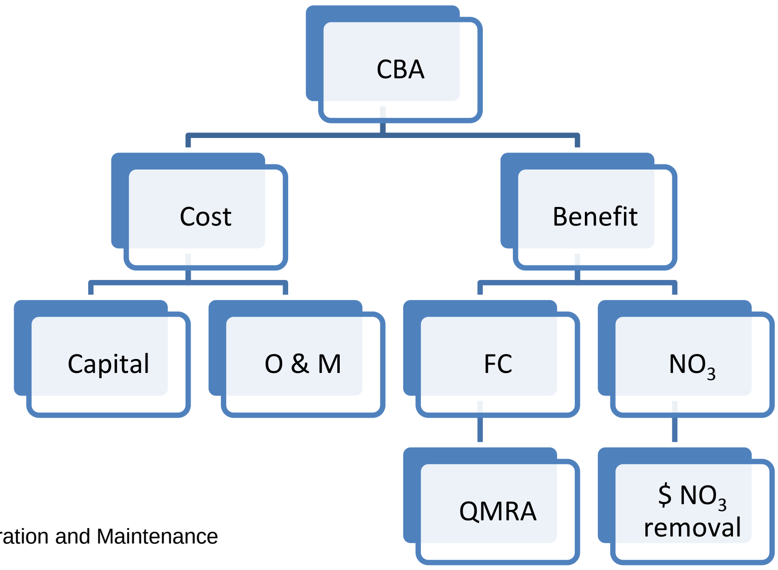
Intervention	Wastewater treatment	% [3]Dom WW treated	% [4]Liv WW treated	% nitrate removal	Log [5]FC removal
Baseline	[1]ST	94	0	0	1
	[2]WWTP	6	0	80	3
1	ST	94	0	30	3
	WWTP	6	0	80	3
2	ST	0	0	0	1
	WWTP	100	0	80	3
3	ST	0	0	0	1
	WWTP	100	0	80	3
4	ST	94	0	0	1
	WWTP	6	0	100	6
5	ST	0	0	0	1
	WWTP	100	0	100	6
6	ST	0	0	0	1
	WWTP	0	100	100	6
7	[6]BMPs	0	0	65	0

[1]ST= Septic Tanks; [2]WWTP=Wastewater Treatment Plants; [3]Dom= domestic urban and domestic rural; [4]Liv= Livestock; [5]FC= Faecal Coliforms; [6]BMPs: Best Management Practices in agriculture. NOTE: the first 5 interventions are designed for the domestic urban and rural sectors; intervention 6 is designed for the livestock sector, and intervention 7 is designed for the agriculture sector, as discussed on section 6.6.

The interventions proposed differ in terms of pollutant removal efficiency, treatment capacity and the costs and benefits obtained. Even though interventions 2 and 3, could achieve similar benefits in terms of removal efficiency, it is expected a significant cost difference, which is evaluated through cost-benefit analysis described below. For instance, considering the karstic conditions of the aquifer in the MAM, which exacerbate drainage construction due to the porous carbonate rocks and limestone beneath soil, intervention 2 (connecting ST to WWTP) would be significantly more expensive than intervention 3 (collecting ST to WWTP), which requires instead vacuum trucks to transport the wastewater.

5.5. Cost-benefit analysis

Figure 30 shows a schematic representation of the cost-benefit analysis (CBA) performed in this thesis. The cost component was derived from Capital (investment), and operation and maintenance (O&M), based on data available from existing wastewater treatment infrastructure. For the benefit analysis, two components were considered: a) economic value gained due to disease averted estimated through QMRA specifically for diarrhoea caused by pathogenic *E. coli* in drinking water, and b) economic value gained from averted nitrate removal treatment prior to drinking water use.



O&M= Operation and Maintenance

Figure 30 Schematic process to develop cost-benefit analysis (CBA) for the MAM case study

a) Cost-saved due to disease averted

Drinking water standards are generally based on the assumption that natural water is used as water supply (Hutton and Haller, 2004). Therefore, for the cost-benefit analysis of this research, direct use of groundwater consumed as drinking water was assumed for MAM case study. The former is considered as the worst case scenario in order to estimate Quantitative Microbial Risk Assessment (QMRA).

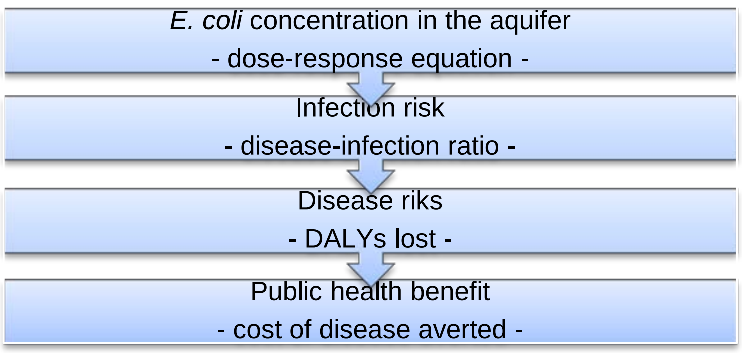


Figure 31 Steps taken to develop QMRA for *E. coli* in the case study

In real scenario, direct groundwater consumption represents around 20% (mainly in rural areas) of the MAM case study, which still rely on backyard shallow water wells (5-6.5m depth), which have no data of water quality or any monitoring control (Alonzo and Acosta, 2003). Figure 31 shows a schematic representation to describe the development of the health benefits for this thesis, measured through QMRA.

b) cost-saved from nitrate removal treatment averted

The economics of different alternatives for nitrate removal during wastewater treatments were evaluated based on data from current wastewater infrastructure designed for this purpose. In general, costs and benefits were estimated based on the data source reported in Table 63. These are existing and potential wastewater infrastructure for the different interventions under evaluation.

Table 63 Data source to estimate interventions costs

Costs	**Data**	**References**
	Materials and labour	Varela, (2008); Gonzalez, (2013)
	Wastewater treatment cost	SEDUMA, (2009)
	Operation and maintenance costs	JAPAY, (2013)
	Current WWTP: costs, treatments, capacity and specifications	Perez, (2006a); Perez, (2006b); BIY, (2007)
	Minimum wage	CONASAMI, (2014)
	Interest rate	SEDUMA, (2009)
Benefits	**Data**	**References**
	Local study data	SEDUMA, (2009)
	B-Poisson coeff. for *E.coli*	Haas et al., (1999)
	QMRA for drinking water	Howard et al., (2006)
	Cost-saved for nitrate removal treatment	Arquiespacios del sureste, (2007); Dunas, (2007)
	Cost-saves for averted diarrhoea disease	Aviv, (2007); CONAGUA, (2008); SEDUMA, (2009)

Chapter 6. Results of modelling interventions

This chapter contains modelling results, which are presented by intervention from the baseline scenario to the intervention 7.

As the basis of all scenario simulations and the 7 interventions, population growth is defined assuming a growth rate of 1.74% per year for the full simulation period (1990-2060) for the 4 aquifer sections (Figure 32).

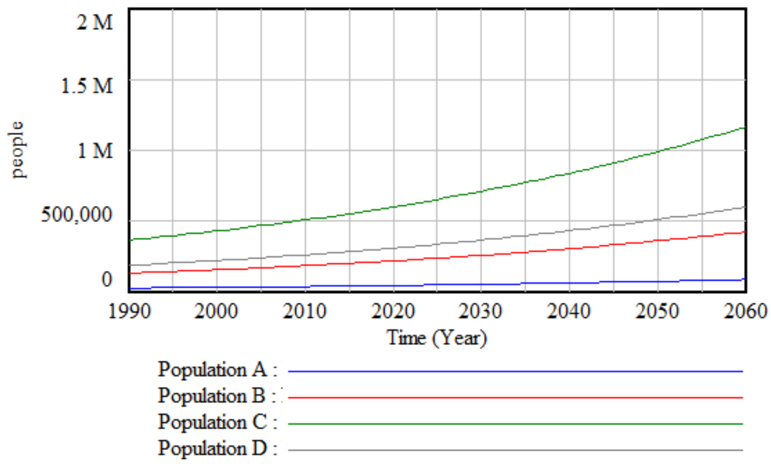


Figure 32 Simulated population development in the four aquifer sections of the study area

As a reminder, Table 64 summarises selected model input data with respect to aquifer dimensions and flow rates.

Table 64 Model input for pollutants simulations per aquifer section

Data		Section A	Section B	Section C	Section D
*Recharge (m^3/s)		7.3	7	7	5.5
GW inflow (m^3/s)		8.32	15	21	28
WW return (m^3/s)		0.52	1.65	3.32	1.95
Aquifer volume (m^3)		2.88E10	2.14E10	1.99E10	5.42E9
Aquifer area (km^2)		1,148	1,148	1,141	1,144
Population	1990	27832	133308	365510	188507
	2010	38790	185794	509419	262726

*Recharge is considered 15% of total rainfall. GW= groundwater

Table 65 Percentage of wastewater treated by current infrastructure in the MAM

Wastewater Treatment	Aquifer Section A	Aquifer Section B	Aquifer Section C	Aquifer Section D
[1]ST	100	97	83	95
[2]WWTP	0	3	17	5

Author estimation based on CONAGUA (2011). [1]ST= Septic Tank; [2]WWTP= wastewater Treatment Plant.

Another general input for each modelled scenario in the absence of interventions relates to the treatment infrastructure in place which is distributed as shows in Table 65. There is a total of 25 wastewater treatment plants distributed within the MAM area, which in total corresponds to less than 15% wastewater treatment coverage.

6.1. Modelling nitrate concentration

6.1.1. Baseline and “stopped inflow”

Scenario 1: Figure 33 shows the “baseline” simulation, which represents nitrate concentration by aquifer section, based on the assumptions and parameter input as described in Chapters 4 and 5, with the treatment infrastructure currently in place.

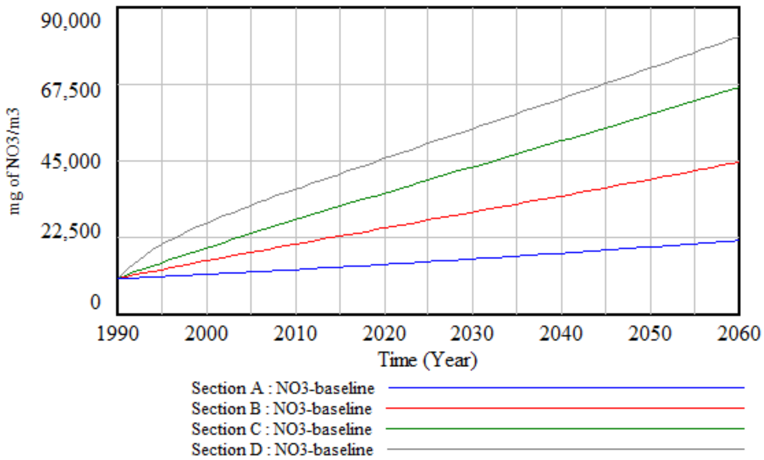


Figure 33 Nitrate concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; source of nitrate are: Aquifer background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock, with 80% removal by WWTP in place for DU and DR; no treatment for LIV and AGR.

Scenario 2: As scenario 1 but from the year 2010, any nitrate inflow into the aquifer is stopped, with exception of background concentration of groundwater in section A (Figure 34).

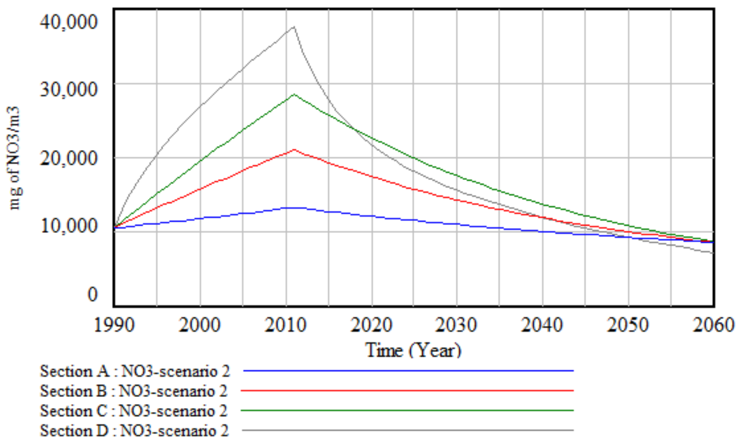


Figure 34 Nitrate concentration in the 4 aquifer sections of the MAM. Conditions: as in scenario 1 but after 2010 any nitrate inflow from anthropogenic sources is stopped.

Scenario 2 is a hypothetical scenario that displays the delayed concentration decrease of nitrate, even when 100% removal of nitrate from wastewater is considered. In section C, for example, the decrease in the first year after improvement is negligible, and it takes about 25 years to achieve a 50% nitrate reduction. This can be explained by the very large aquifer volume that delays the wash-out effect through groundwater inflow and rainwater. It is important to consider this delay for predicting the effectiveness of treatment interventions.

6.1.2. Validation of NO_3 results

In order to compare the values obtained from the model with field data per aquifer sections A-D (2009 and 2010), available field data are summarized in Table 66.

Table 66 Field data of nitrate concentration within the MAM, from water supply wells

Location	Average in mg/m^3 (range)	Year tested	Reference	Conditions
Section A	1.75E+04 (1.5E+04-2E+04)	2009/10	Torres, (2010)	Deep water supply-wells, about 20 located in aquifer sections A-D. Concentration range derived from nitrate concentration map.
Section B	2.25E+04 (1.5E+04-3E+04)	2009/10		
Section C	3 E+04 (1.5E+04-4.5E+04)	2009/10		
Section D	2.25E+04 (2E+04-2.5E+04)	2009/10		
Section A	1.3E+04	2003/4	Osorio, (2009)	Deep water supply-wells from 106 municipalities of Yucatan 2003-2004.
Section B	1.8 E+04	2003/4		
Section C	2.1 E+04	2003/4		
Section D	1.6 E+04	2003/4		
Section A	1.9 E+04	2007/8		Deep water supply-wells from 106 municipalities of Yucatan 2007-2008.
Section B	2.8 E+04	2007/8		
Section C	1.9 E+04	2007/8		
Section D	2.5 E+04	2007/8		

All data in Table 66 refer to nitrate concentration in deep water supply wells (45m depth). The model assumes homogeneous distribution of nitrate within an aquifer section, and therefore reflects better the nitrate concentration in the deep aquifer than elevated concentrations in near-surface water. Near-surface water might not be representative of the entire aquifer due to its proximity to pollution point sources. The most recent study in this respect was documented by Torres, (2010), covering north-western Yucatan, which covers most of the study area. Torres, (2010) reported average values for 15 shallow water supply wells (≤15m depth), and 20 deep wells (45m depth), which were measured from September 2009 to February 2010. In average nitrate concentration in shallow wells is only 1.8 fold higher than in deep wells. This confirms that the homogeneous dilution approach taken in this research is a reasonable approximation for nitrate, even though there is a certain depth gradient and punctual peak concentrations, in particular within Merida area (Aquifer section B, C and D). Another relevant study was documented by Osorio, (2009) which covers full Yucatan State. Samples were taken from the 106 water supply wells at a depth of 45m, which decreases toward the coast. Osorio, (2009) also addressed time-dependent changes in nitrate concentration by comparing field data reported for 2003-2004 and the same field data for 2007-2008.

Figure 35 illustrates the reasonable agreement of modelled nitrate levels and field data (2003, 2007 and 2009) for section A, B, and C. The difference between simulated data and field data 2003-2004 and 2007-2008 is within 50%. Moreover, simulated data lie within the range of the nitrate concentrations derived in the study of Torres, 2009 (section D is not considered in this study). For section D, the model strongly overestimates nitrate concentration relative to field data. This may be related to the difficulty of estimating the aquifer thickness in section D where several coastal municipalities quantify the thickness with only 1m. Due to the low thickness of the freshwater lens in section D, a significant portion of the pollutants may diffuse into the underlying seawater as documented by Wissmeier et al., (2009). Another factor that could be responsible for the discrepancy for aquifer section D is the saline intrusion from the coast that facilitates the formation of tidal freshwater wetlands (TFWs), favouring denitrification by a mechanisms called dissimilatory reduction of nitrate to ammonia (DNRA) with up to 30% of nitrate reduction in coastal sites (Osborne et al., 2012; Giblin et al., 2013).

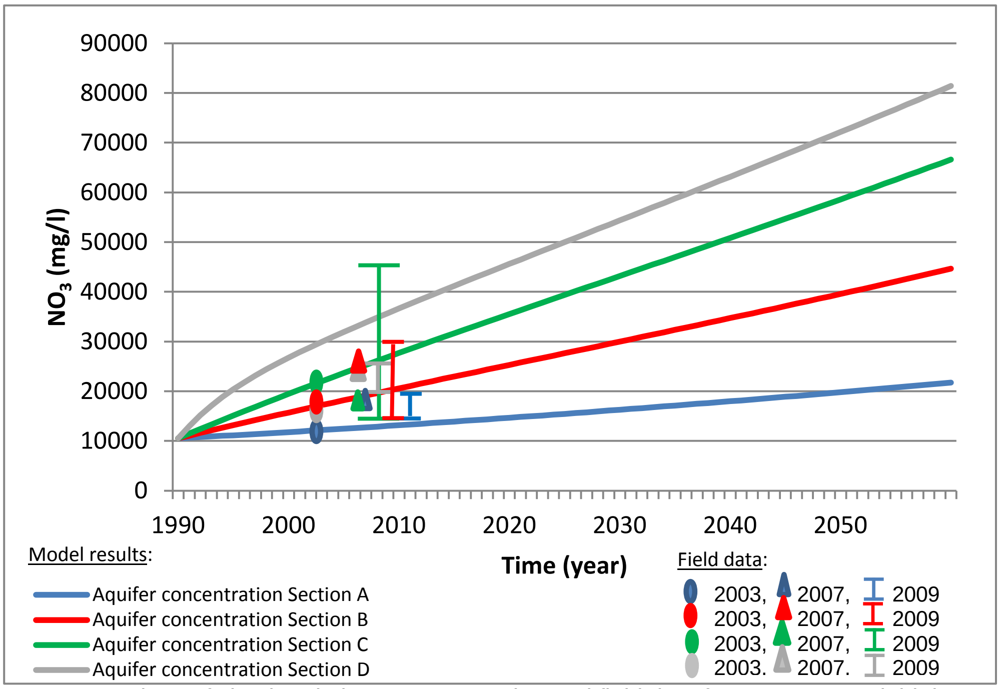


Figure 35 Comparison of simulated nitrate concentration and field data for 2003-2009. Field data 2003, 2007 from Osorio, (2009) and intervals from Torres, (2010)

Beyond these studies, there are several investigations described in the literature regarding nitrate concentration for specific, small areas of the MAM, in particular, close to Merida city, but these point source data were considered neither representative nor appropriate for comparison with the modelling results. Furthermore, due to diffuse and point sources of pollution, wastewater discharges contribute significantly to the overall composition of the groundwater as reported in different studies documented by USEPA, (2010).

In summary, simulated nitrate concentrations are in reasonable agreement with field data for deeper sections of the aquifer, where a more homogeneous distribution of nitrate is expected due to diffusion and/or groundwater flow over extended time periods. Therefore, no additional adjustments to the model were carried out for nitrate simulation. A limitation of the model is that it overestimates nitrate levels in section D at low thickness of the freshwater lens. Besides, the model was not designed for and does not reflect elevated nitrate levels in the proximity of near-surface point sources.

6.2. Modelling faecal coliform concentration: initial attempt

6.2.1. Baseline and "stopped inflow"

Scenario 1. This is the "baseline" simulation which represents faecal coliform concentration by aquifer section (Figure 36), based on the assumptions and parameter input as described in chapters 4 and 5, assuming homogeneous distribution of the pollutant over the whole aquifer section volume, and with the wastewater management infrastructure currently in place.

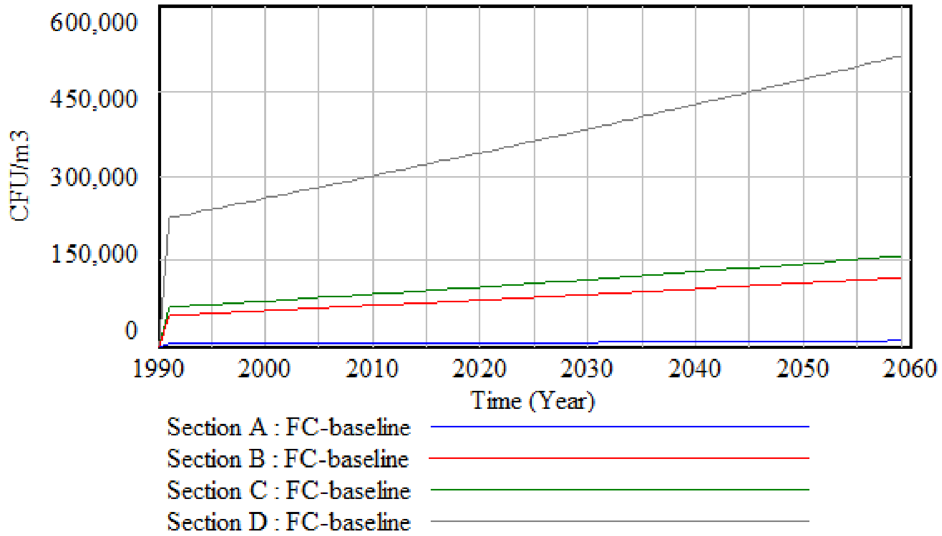


Figure 36 FC baseline concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of FC are: DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. For DU and DR, ST removes 1 log FC and WWTP 3 log FC, no treatment for LIV and AGR.

Scenario 2. As scenario 1, but from the year 2010, any FC inflow into the aquifer is stopped. Scenario 2 (Figure 37) is a hypothetical scenario that displays the rapid elimination of faecal coliform from the groundwater when 100% removal from wastewater is considered. This illustrates the very different response times of faecal coliform and nitrate to treatment.

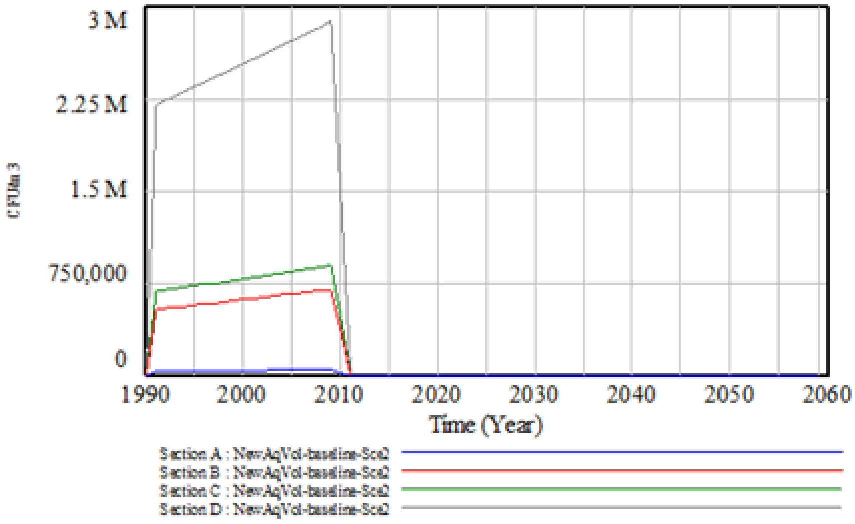


Figure 37 FC concentration in 4 aquifer sections of the MAM. Condition: as in Scenario 1, but after 2010 any FC inflow is stopped

6.2.2. Validation of FC results

In order to compare model results with published data, field data from Yucatan, Merida and specifically from sections A-D from deep water supply wells have been summarized in Table 67.

Table 67 Faecal Coliforms (FC) concentrations reported within the study area

Location	Average reported CFU/100ml	Value converted CFU/m^3	Year tested	Reference	Conditions
Section A	1.70E+01	1.70E+05	2007/8	Osorio, (2009)	Deep water supply-wells from 106 municipalities of Yucatan over 1 year
Section B	7.72E+01	7.72E+05	2007/8		
Section C	8.06E+01	8.06E+05	2007/8		
Section D	2.22E+01	2.22E+05	2007/8		
Merida	4.00E+04*	4.00E+08	1983	Cabrera et al., (1983)	Boreholes and shallow wells in rural area
Merida	1.00E+03	1.00E+07	1991-1993	Foster and Chilton, (2004) BGS et al., (1995)	Based on BGS et al., (1995)
Merida	2.50E+03	2.50E+07	2003	Morris et al., (2003)	Based on BGS et al., (1995)
Merida	4.39E+01*	4.39E+05	2010	Mendez et al., (2010)	Merida
Merida	2.66E+03	2.66E+07	2012	Heise, (2013)	29 wells Merida-Progreso
Yucatan	5.00E+03	5.00E+07	2002	Pacheco et al., (2002)	55 shallow wells of 92 municipalities
Yucatan	6.90E+01	6.90E+05	2003	Osorio, (2009)	106 water supply wells of the 106 municipalities
Yucatan	4.70E+03	4.70E+07	2003	Pacheco and Cabrera, (2013)	106 water supply wells of the 106 municipalities
Yucatan	4.32E+01	4.32E+05	2007	Osorio, (2009)	106 water supply wells of the 106 municipalities
Yucatan	1.00E+03*	1.00E+07	2009	Cabrera et al., (2010)	32 sinkholes in Yucatan

*reported as MPN/100ml

The model assumes homogeneous distribution of FC within an aquifer section, and might therefore reflect better the FC concentration in the deep aquifer than elevated concentrations in near-surface water in the proximity of point sources (as the other data in Table 67 for Merida and Yucatan).

Figure 36 illustrates that simulated FC concentrations are much lower, by a factor of about 10, than field data for the deep aquifer in sections A, B and C. Only for section D, field data match the simulated data. The latter might however be an accidental match, considering the factors that may reduce pollutant concentration in the low-thickness freshwater lens of section D, as discussed in section 6.1.2. with respect to the nitrate levels in this aquifer section.

Poor correlation for FC simulation could be attributed to two main facts that may be governing the transport of FC in the MAM karstic aquifer: 1. The aquifer is modelled in this thesis as a medium with homogeneous porosity but karstic aquifers are intrinsically heterogeneous, involving for instance a conduit-matrix structure, and 2. FC and in general microbial transport is complex to reproduce due to their relatively short half-life compared with conservative chemical pollutants, and the half-life also depends on environmental factors. In addition, strong seasonal fluctuation of FC concentration in karstic groundwater has been documented (Desai, 2010). A more detailed explanation of the features that complicate the modelling of FC transport and concentration in karstic aquifers are described below.

1. Karst aquifers nature. Due to the heterogeneity of the porous medium, even within the same aquifer section, site-specific water fluxes may govern pollutant transport. In certain deep wells in aquifer section C, where most of Merida municipality is located, 8 CFU/100 mL have been measured, while in others, up to 230 CFU/100mL have found. This indicates that local concentration observes high variations, possibly due to the fact that FC are transported through a conduit system from local point sources to deeper sections of the aquifer. Based on the literature, some of the most common difficulties for modelling fracture networks such as the MAM karst aquifer are as follows (Savarovsky et al., 2012; Drew & Holtz, 1999):
    - Limited amount of site-specific information about fracture and conduit position, conductivity, and interconnection
    - Complex flow paths
    - Diffuse flow (non-linear)
    - Shallow water table (predominant conduits in the phreatic zone)
    - Heterogeneous permeability (complex groundwater flow)
    - Non-linearity between water flow velocity and hydraulic gradient

2. FC transport. Modelling the behaviour of FC as a non-conservative pollutant in a karstic aquifer is complicated by the following:
   - Spatial variability is highly correlated to land use (Petersen, 2006, and Desai, 2010**)**.
   - Storm events have been associated with increases in bacterial concentration in aquifers due to erosion and re-suspension of sediments (Boehm et al, 2002; Reeves et al., 2004).
   - Solar radiation and in-situ regrowth are recognized as important factor that control bacteria levels in aquatic systems. Microbial enhancement leading to biofilm formation can obstruct the porous medium, known as pore clogging or bio-clogging, resulting in small changes in the physical structure of the porous medium i.e. reduce hydraulic conductivity of the matrix at local scale (Taylor and Jaffe, 1990; Baveye et al., 1998).

These features could considerably affect the transport and survival of pathogens. In particular, with respect to their particle size that may limit their diffusion into small pores of the aquifer matrix and prevent a homogeneous distribution in the aquifer within the limited microbial life time.

To summarize, the specific features governing FC transport in the karstic aquifer may contribute to a general underestimation of FC concentration by the model that considers homogeneous distribution in the total aquifer volume, resulting in “over-dilution”. The following sections therefore describes a refined approach, specifically applied to FC modelling, to better reproduce FC transport by redefinition of the aquifer volume in which FC pollutants effectively distribute.

6.2.3. Refined approach for FC simulation in the accessible aquifer volume

In this work the aquifer has so far been modelled as a single continuum and equivalent porous medium. This approach appears suitable for modelling of nitrate as a persistent contaminant, since the latter is expected to reach a relatively homogeneous distribution due to diffusion, in view of the simulated time period of several decades.

Other popular modelling concepts for karst aquifers include conduit flow modelling only, suitable in particular when the matrix has low permeability, and biphasic conduit-matrix models that include exchange between the phases by diffusion. The volume of conduits and fissures in karst aquifers is generally considered only a small fraction of the total aquifer volume.

This raises the question whether an alternative model is more suitable for simulating concentrations on FC as a non-conservative pollutant in the MAM karstic aquifer. The rationale is the decay of FC with a half-life of a few days only what limits the probability of extensive permeation of the matrix, or homogeneous distribution in the total aquifer volume, respectively, simply because FC would die-off during the diffusion process.

Since transport of FC from the surface to and within the conduit system is expected to be fast, a major fraction of the FC load should be present in the small volume of the conduit system plus a limited volume (fissures and large pores in close proximity to the conduits) of the matrix that is readily accessible to FC by diffusion. In this alternative model, the FC load resides in a small fraction of the total aquifer volume rather than spontaneously distributing over the total volume. Since data for the conduit volume are not available for the study area, the issue was addressed by a calibration that varies the aquifer volume between 1% and 100% of the total aquifer volume, followed by a comparison with field data (Figure 39). The following literature observations further support the idea that FC may reside in the limited volume of the conduits and readily accessible matrix volume:

- In a karst aquifer in Florida, the flow of rain in the conduits has been directly visualized by electrical conductivity tomography (Meyerhoff et al., 2012). This technique allows monitoring of rainfall infiltration into conduits on the basis of varying salt concentrations that affect the conductivity. First of all, after a rainfall event, the conductivity of major parts of the explored aquifer volume remained unchanged. A drastic change was, however, observed at conduit sites in about 5-25 m depth, with lag times of 1-3 days, indicating the selective infiltration of the conduits by rainwater with low salt concentration. With an additional lag time of about 3 days, the rainwater seems infiltrates the matrix in close proximity to the conduits. A similar scenario may be assumed for FC that would be flushed in the conduits after rainfall. However, in contrast to the small size (<1nm or 10^{-9} m) ionic components of salts, the particle size of FC in the range of 1 µm or 10^{-6} m should disfavour diffusion from the conduit into the small pores of the matrix. Diffusion would eventually be limited further by formation of biofilms that closes small pores of the matrix. To overcome this issue, a calibration of the aquifer volume was carried out for FC modelling in order to define a reduced volume of the aquifer in which FC is effectively distributed.

6.3. Calibration of the model for FC simulation

The following section describes the calibration of the aquifer volume specifically for FC modelling. This refined approach, which is based on the literature review, assumes predominant groundwater flow through conduits and fractures. FC is assumed to be flushed from the surface along with the wastewater and rainfall into a small conduit volume, from where it diffuses into readily accessible parts (including large pores and fissures) of the limestone matrix.

Since data for the conduit and readily accessible matrix volume are not available for the study area, the issue was addressed by a calibration process that varies the aquifer volume by a “reduction” between 10% and 99% (or -0.1 to -0.99 if expressed in fractions as in Table 68) relative to the original volume, along with a comparison with field data (Table 68 and Figure 38). This should not be considered as an actual reduction of the total aquifer volume but interpreted as the effective volume (or fraction of the total aquifer volume) where FC are concentrated.

Table 68 Aquifer volume variation for calibration of the FC modelling

Aquifer vol. variation		**FC concentration (CFU/m^3) per aquifer sections**			
fraction	**per cent**	**Model A**	**Model B**	**Model C**	**Model D**
Publish data (2007)		**1.7E+05**	**7.72E+05**	**8.06E+05**	**2.22E+05**
Baseline (100%)		**4.27E+03**	**7.12E+04**	**9.10E+04**	**3.01E+05**
-0.1	-10%	4.75E+03	7.90E+04	1.01E+05	3.34E+05
-0.2	-20%	5.35E+03	8.91E+04	1.14E+05	3.75E+05
-0.3	-30%	6.09E+03	1.02E+05	1.30E+05	4.29E+05
-0.4	-40%	7.11E+03	1.19E+05	1.52E+05	5.00E+05
-0.5	-50%	8.54E+03	1.42E+05	1.82E+05	5.99E+05
-0.6	-60%	1.07E+04	1.78E+05	2.27E+05	7.47E+05
-0.7	-70%	1.42E+04	2.37E+05	3.03E+05	9.91E+05
-0.8	-80%	2.13E+04	3.55E+05	4.54E+05	1.49E+06
-0.9	**-90%**	4.25E+04	**7.08E+05**	**9.06E+05**	2.91E+06
-0.95	-95%	8.48E+04	1.40E+06	1.88E+06	5.64E+06
-0.97	-97%	**1.41E+05**	2.32E+06	2.98E+06	9.00E+06
-0.99	-99%	4.11E+05	6.62E+06	8.63E+06	2.27E+07

Calibration was carried out with published data (Osorio, 2009). If the volume of the aquifer is reduced by 90%, a very good match with field data is obtained for section B and C, while FC concentration is still underestimated (but now only by a factor of about 3) for section A and overestimated (by a factor of about 10) for section D (Figure 38).

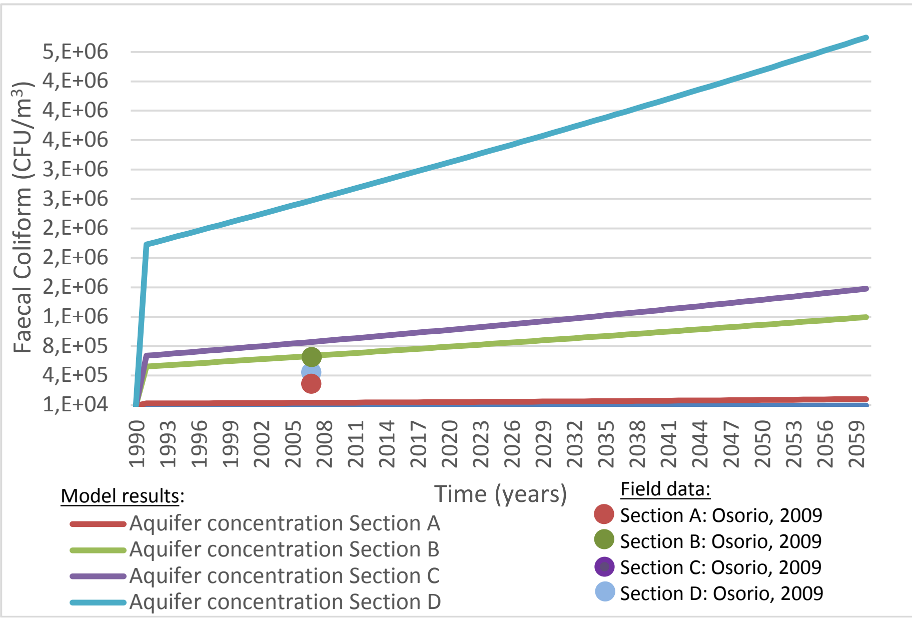


Figure 38 Comparison of simulated FC concentration and field data for deep aquifer wells in 2007

In Figure 39, the volume reduction is expressed in % reduction of the original aquifer volume. For instance, a variation by -90% means the reduced volume is 10% of the original volume.

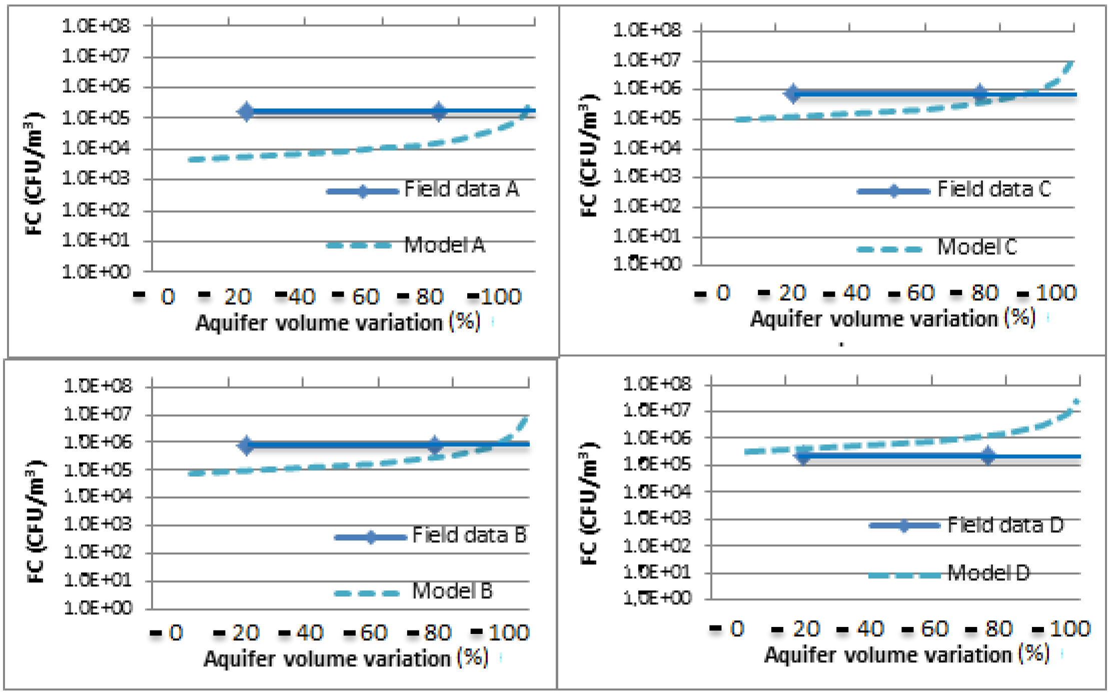


Figure 39 Calibration of FC concentration: aquifer volume variation expressed in % reduction of the original volume. Field data from Osorio, (2009)

Overall, reduction of the aquifer volume by 90%, i.e. an aquifer volume of 10% of the original volume was used for further calculations in Chapters 6 and 7, since it provides not only a reasonable (although not optimal) overall match with field data, but also the best match for section C, which is the focus area of this study as detailed in the cost-benefit analysis (Chapter 7). In this section water quality issues are of particular concern due to the highest population density and thus the highest water consumption and wastewater discharge.

6.4. Sensitivity analysis of FC concentration

Sensitivity analysis (SA) serves to test how the uncertainty of the output of a model (such as pollutant concentration in the present research) is related to the uncertainty of specific input parameters.

In order to identify those parameters of greater importance for modelling FC concentration, three input parameters are tested in the sensitivity analysis. These are: aquifer volume, rainfall and the die-off rate of FC. Based on calibration results (section 6.3), a reduced aquifer volume of 10% of the original volume was applied to all further calculations of FC concentration in the MAM aquifer.

Table 69 Input parameters tested for sensitivity analysis of FC (CFU/m^3) concentration

Parameter to test (±50%)	Aquifer section	Baseline value	Range
Aquifer volume (m^3)	**A**	2.88E+09	1.44E+09 to 4.32E+09
	B	2.14E+09	1.07E+09 to 3.21E+09
	C	1.99E+09	9.95E+08 to 2.99E+09
	D	5.42E+08	2.71E+08 to 8.13E+08
Rainfall (m^3/s)	**A**	7.3	3.65 to 10.95
	B	7	3.5 to 10
	C	7	3.5 to 10
	D	5.5	2.75 to 8.25
Die-off constant rate "k" (s^{-1})	**A-D**	1.4E-06	7E-07 to 2.83E-06

Sensitivity analysis was performed with the Vensim platform using the Monte Carlo (MC) function. For the MC simulation three parameter settings are important: 1. The minimum and maximum values to change the parameter under evaluation, which is shown in Table 69; 2. The distribution or variation of the selected parameters, which was set as a random uniform distribution, means that all possible values of the parameters would have the same probability to occur; 3. The number of simulations to run for the sensitivity test, which was set 200 (default number of simulation). Aquifer volume, rainfall and die-off rate k were varied as shown in Table 69, in order to identify the effect of these variations to the FC concentration in the 4 aquifer sections. Results of the SA are shown in Table 70. Additionally, SA results are presented from Figure 40 to Figure 42, corresponding to variations in aquifer volume, rainfall and die-off constant rate "k" respectively. The colours indicate the confidence boundaries of FC concentration from the 200 simulation results, where the beige colour represents the 50% confidence, the green is the 75% confidence, the blue is the 95% confidence and the grey is the 100% confidence. These graphs also include field data from Osorio, 2009 for comparisons of SA results for the year 2007.

Table 70 Sensitivity analysis results of FC concentration in the 4 aquifer sections

Parameter to test	Aquifer section	Published Data FC (CFU/m^3)	Model result FC concentration (CFU/m^3) for 2007		
			Average	**Min**	**Max**
Aquifer volume (m^3) (±50%)	**A**	1.70E+05	4.55E+04	2.71E+04	8.07E+04
	B	7.72E+05	7.29E+05	4.54E+05	1.34E+06
	C	8.06E+05	9.49E+05	5.81E+05	1.69E+06
	D	2.22E+05	3.12E+06	1.89E+06	5.30E+06
Rainfall (m^3/s) (±50%)	**A**	1.70E+05	4.06E+04	4.05E+04	4.06E+04
	B	7.72E+05	6.80E+05	6.78E+05	6.81E+05
	C	8.06E+05	8.69E+05	8.68E+05	8.71E+05
	D	2.22E+05	2.80E+06	2.77E+06	2.83E+06
Die-off constant rate "k" (in s^{-1}) (±50%)	**A**	1.70E+05	4.47E+04	2.71E+04	8.01E+04
	B	7.72E+05	7.48E+05	4.54E+05	1.34E+06
	C	8.06E+05	9.58E+05	5.81E+05	1.71E+06
	D	2.22E+05	3.06E+06	1.89E+06	5.37E+06

NOTE: Published data from Osorio, (2009) for the year 2007

In general, it could be observe d a good correlation between model results with field data for aquifer sections B and C, while over- and underestimations are observed for aquifer sections A and D, respectively. In summary, the model is able to reproduce adequately the FC field data for the aquifer sections B and C, which are the main interest of this research and serve to develop the following steps of this research, in particular the cost benefit analysis focusing in section C (Chapter 7).

From variations of aquifer volume (Figure 40), FC concentrations results are within 50% of confidence bound for the four aquifer sections, including the FC concentration obtained with the baseline value of aquifer volume (blue line). From variation of rainfall (Figure 41), FC concentrations results are almost the same (overlapped in the graphs), indicating the negligible effect of rainfall. From variation of "k" die-off constant rate (Figure 42), FC concentrations results are within 50% confidence bound for the four aquifer sections, including the FC concentration obtained with the baseline value (blue line) of "k".

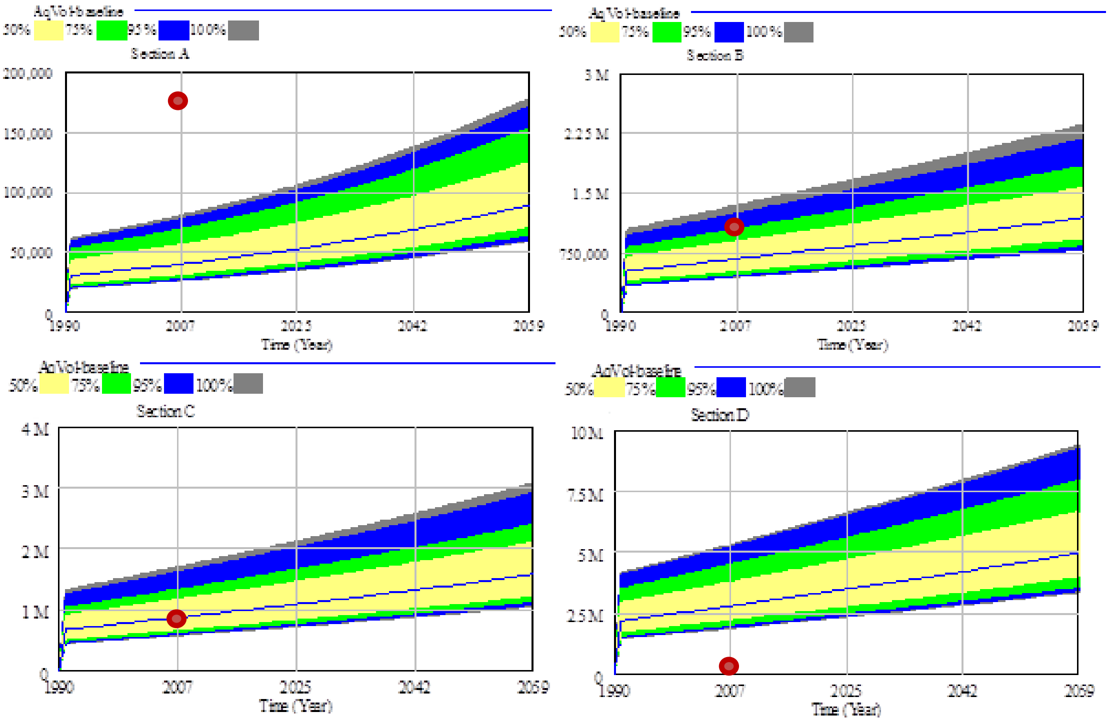


Figure 40 Comparison of SA results from aquifer volume variations (±50%) with FC concentration field data (●) from Osorio, (2009) for the year 2007

Overall, the SA results confirm the importance of the aquifer volume parameter, and the sensitivity of FC levels to variation of the aquifer volume, as well as the sensitivity variations of the die-off constant rate "k". In contrast, rainfall variation within comparable range has almost no effect on FC concentration. The general negligible effect of rainfall variation on FC concentration means rainfall is not a determinant parameter for FC in yearly bases simulation. This is further explored in the following section by short-term simulations to identify seasonal variation of FC concentration, thus to determine if rainfall could have a bigger impact in the FC concentration.

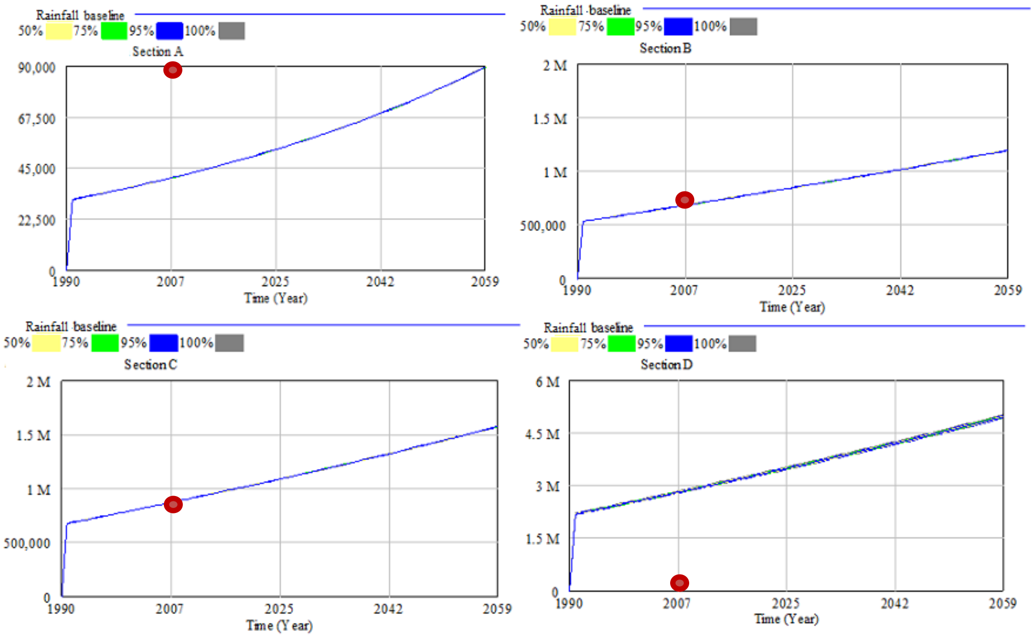


Figure 41 Comparison of SA results from rainfall variations (±50%), with field data (●) from Osorio, (2009) for the year 2007

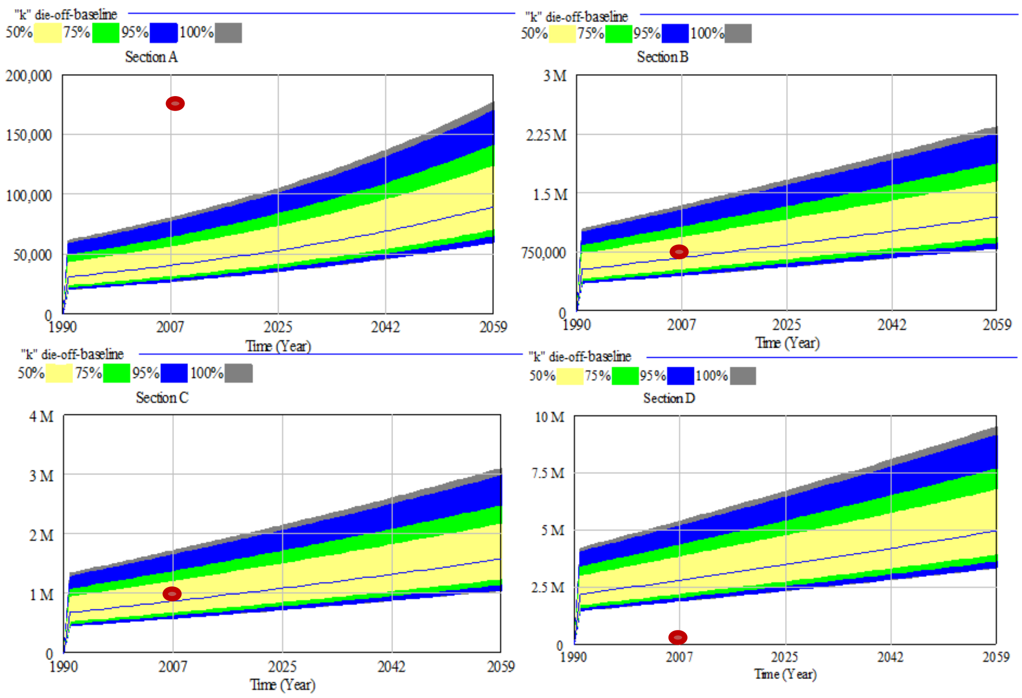


Figure 42 Comparison of SA results from "k" die-off constant rate variations (±50%), with field data (●) from Osorio, (2009) for the year 2007

6.5. Simulation of seasonal variation of FC concentration

So far the model has kept rainfall constant with time at about 7 m^3/s, without considering seasonal variations. However, in the study area there is a dry season (November-April) and a rainy season (May-October), with very different monthly precipitation. Field data about seasonal variation of FC concentration in the study area are limited, but a study of Pacheco et al., 2000 (sampling year 1983) covering 17 shallow wells in the north of Merida, strongly support the idea of a seasonal variation of FC concentration. In the dry season, FC concentration dropped to a minimum of 1 CFU/100ml, while in the rainy season peak concentrations in the range of 1x10E+05 CFU/100ml are reached. The study has shown a correlation between temporal variation of FC concentration and the precipitation pattern (Figure 43) considering that the rainfall washes FC from its points of origin at the surface into the aquifer. Physically, one may imagine this as a flushing-out effect that is particularly efficient at high rainfall, while in the absence of rain the microbial contaminants reside in the unsaturated zone (upper layer of the aquifer) and die-off.

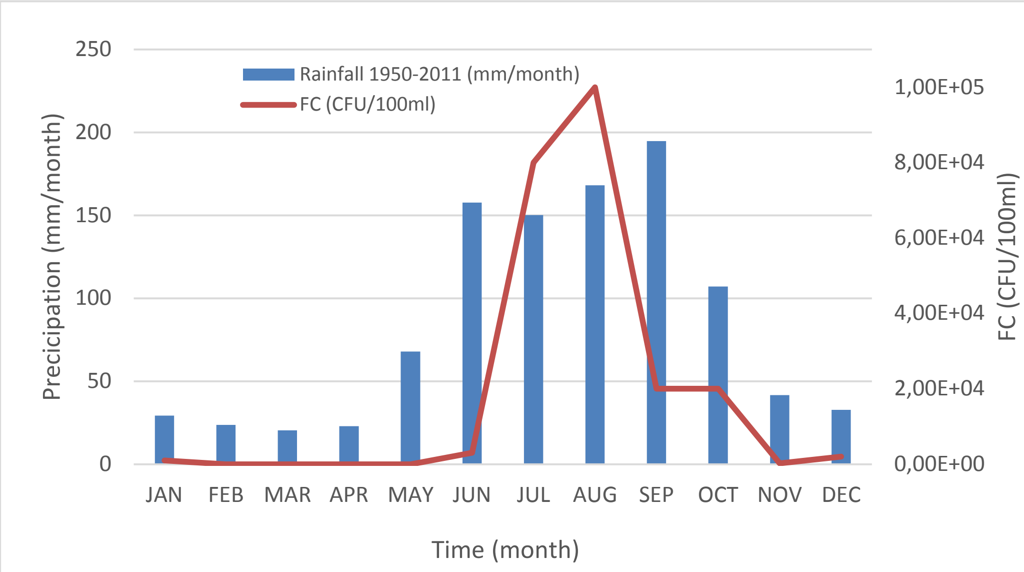


Figure 43 Comparison of seasonal variations of FC concentration (shallow wells in the north of Merida, in 1983 from Pacheco et al., (2000), and average monthly precipitation for Merida from INEGI, (2011)

As an example, the seasonal variations for section C were implemented in the model by considering monthly (rather than annual) rainfall and associating the FC concentration resulting from more or less effective infiltration in the aquifer with the amount of monthly rain. In detail, the FC level was multiplied with a factor that is the ratio between rainfall in a specific month (INEGI, 2011) and, the average monthly rainfall. Consequently, the concentration of FC in the aquifer is positively correlated with the amount of rainfall, as documented by Pacheco et al., (2000).

With these assumptions and framework conditions, the concentration of FC was simulated on a monthly timescale. Figure 44 exemplifies the results for the year 2010, where the pattern of FC concentration is closely related to the precipitation pattern. Due to the short half-life of FC, no significant accumulation is observed and FC concentration is expected to respond to changes in rainfall intensity.

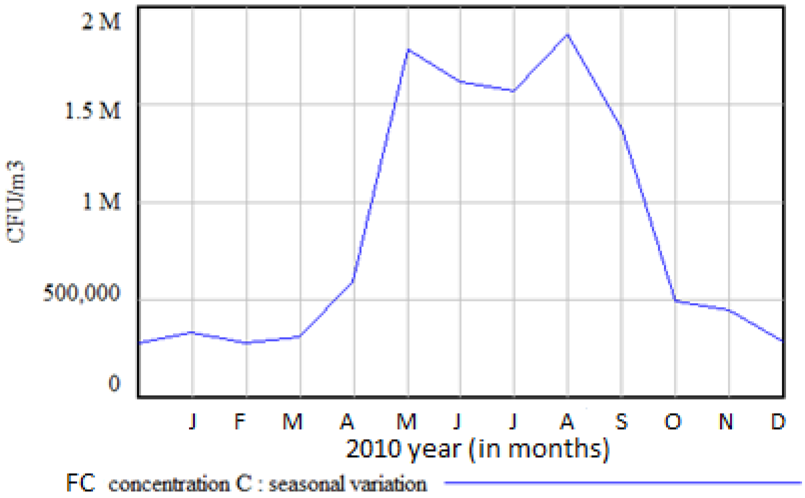


Figure 44 Simulation of seasonal variation of FC concentration in aquifer section C in the year 2010, using simulated FC loads for 2010 and the average monthly precipitation pattern of figure 43, and assuming that FC transfer in the aquifer is positively correlated with precipitation

The results of this simulation show that principally, seasonal variation of FC concentration can be modelled by the SIWMM associating infiltration of FC into the aquifer with the amount of rainfall on a monthly base. Modelling results qualitatively confirm the seasonal trends of FC concentration in shallow wells observed in a field study in the north of Merida.

This modelled FC concentration with two peaks (Figure 44) does not perfectly match the documented field data for the sampling year 1983 with a single maximum peak (Figure 43). This could be due to the limited number of experimental samples, taken from selected water wells and only within a single year rather than over extended time period.

On the longer time scale (2010-2020), implementation of seasonal variations in principle does not alter the trend of FC concentration increase (Figure 45). Average FC concentrations of the seasonal model are comparable to the FC concentration in season-independent model, as given in section 6.3.

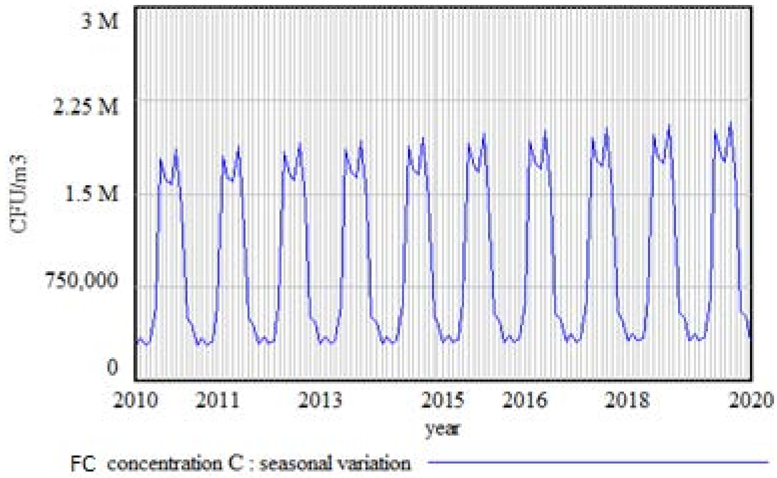


Figure 45 Simulation of seasonal variation of FC concentration in aquifer section C in the time period 2010-2020, using simulated FC loads and the average monthly precipitation pattern of figure 43, and assuming that FC transfer in the aquifer is positively correlated with precipitation.

Climate change scenarios for the Yucatan Peninsula as a consequence of global warming have been developed for the horizon 2020 and predict a regional transitional climate (semi-arid in the northern portion, sub-humid in the south) that is very sensitive to changes (Orellana et al., 2012). The results reveal important temperature increases as well as a significant decrease in precipitation for some territories and an increase for others. Such a change in precipitation would be of very limited impact for the results of the season-independent SIWM model, considering the results of sensitivity analysis of the model (section 6.4), where a variation of annual rainfall by ±50% has practically no effect on FC concentration. As soon as the infiltration of FC into the aquifer is associated to the amount of rainfall, however, as in the seasonal model described in this section, changes of precipitation are predicted to affect FC concentration and water quality in a monthly-based scale.

Furthermore, special considerations must be taken to current climate change conditions globally, which consequently affect seasonal patterns, which could produce uncertainty in the scientific modelling. For instance, Ries et al., (2015) have documented intrinsic changes in a Mediterranean karstic aquifer recharge due to seasonal changes. It was found that approximately 66% of rainfall percolates through the unsaturated zone during rainy season, compared to 0% during dry season. Nevertheless, these seasonal changes are varying over the years. Another issue for seasonal variation analysis is documented by Schmitt, (2010) realising the high risks and uncertainties for investments in resilience interventions due to alterations of seasonal variations with extreme events (i.e. droughts, hurricanes and flooding).

6.6. Modelling interventions

A total of seven interventions were modelled, effectively starting in 2010. All interventions aim to improve public health status and reduce water-related diseases through engineering strategies within the water cycle of the MAM.

The first five interventions are aimed at improving wastewater quality from both domestic urban (DU) and domestic rural (DR) origin. The sixth intervention is designed to improve wastewater quality from livestock activities. The seventh aims to implement best management practices specifically for nitrogen-based fertilizer (N-fertilizer) use in agriculture. Each intervention improves wastewater quality prior to discharge into the MAM aquifer. Therefore, the two major outcomes of these interventions are:

- Protection and preservation of the overall groundwater quality of the MAM and through this
- Reduction of water-related diseases incidence from these two pollutants to achieve the goal of improving public health in the MAM.

The 7 interventions are:

1. Improve all existing septic tanks (ST) from domestic urban (DU) and domestic rural (DR)
2. Connect all ST to existing domestic wastewater treatment plants (WWTP)
3. Collect all ST to existing domestic WWTP (cost difference to intervention 2)
4. Improve existing domestic WWTP removal efficiency:
    a. Activated Sludge/Extended Aeration (AS/EA)
    b. Anaerobic Digester/Trickling Filter (AD/TF)
5. Increase number of domestic WWTP and connect all households to WWTP
6. Create new WWTP for livestock (LIV)
7. Improve nitrogen-based fertilizers management for agriculture (AGR), referred to as Best Management Practices (BMPs)

All interventions are simulated along with the population growth of 1.74% per year. Therefore, it is also important to identify the time-dependent need for an increased wastewater treatment infrastructure, in order to cope with the increasing wastewater release by a growing population. The simulations assume same growth rate of 1.74%/y for all activities since domestic consumption is the major driving force.

In order to establish the effect of potential engineering interventions, these are compared with the current scenario named “baseline” scenario with limited treatment. It is important to notice that current treatment processes in WWTP are mainly of two types: activated sludge (AS) and anaerobic digester (AD), both with some variations but assumed to have the same removal efficiency for the two pollutants. An average of the removal efficiency summarised on Table 71, was used for the simulation.

Table 71 Main treatments used in the Wastewater Treatment Plants of the study area

Treatment Process	Number of [3]WWTP	% of treatment process	Percent of nitrate removal	Logarithm of [4]FC removal	Reference
[1]AS/EA	16	55	80	3 log	Perez, (2006b); Perez, (2006a)
[2]AD/TF	13	45	75	4 log	Alcica Construction, (2009)

[1]AS/EA= Activated Sludge/ Extended Aeration; [2]AD/TF= Anaerobic Digester/ Trickling Filter; [3]WWTP= Waste Water Treatment Plants [4]Faecal Coliforms

These two treatments are widely recommended for tropical countries, because the removal process is mainly depending on the microbial activity intrinsically present in the wastewater. Main differences are related to the costs of investment, operation and maintenance. AS requires a constant energy source for the aeration process, while AD does not. In terms of the removal efficiency for the two pollutants of interest, AS generally provides better removal of NO_3 than AD, while AD provides better FC removal than AS. Nevertheless, the overall performance of these treatments depends on various parameters such as the expert design, qualified operators, and optimal weather and operation conditions.

Intervention 1. Improve all ST from DU and DR. The aim of this intervention is to improve the removal efficiency of existing septic tanks (ST) from domestic urban (DU) and domestic rural (DR). This is achieved through considering additional processes and/or improved design of the existing ST. Hydraulic retention time (HRT) is a determinant for the microbial removal efficiency, usually ST are designed for 48h HRT, but depends on the number and water consumption rates of people per household.

Actions: Including a soil-adsorption system (SAS) after ST with a minimum 900mm of unsaturated soil from the release point to the groundwater table (most of microbial removal has been reported within the first 375mm) could increase faecal coliform (specifically *Escherichia coli* or *E. coli*) removal efficiency from 1 to 3 log reduction of FC (Samimian, 2009). By design specifications there is no NO_3 removal. Nevertheless a 30% NO_3 reduction could be achieved by including SAS immediately after the ST (Costa et al., 2002; USEPA, 1999), and this is also assumed in the simulation. Outcomes: Final outcomes of this intervention are:

FC removal. FC removal increased from 1 log removal in existing ST to 3 log removal in improved ST, with no changes in WWTP. Figure 46 shows results.

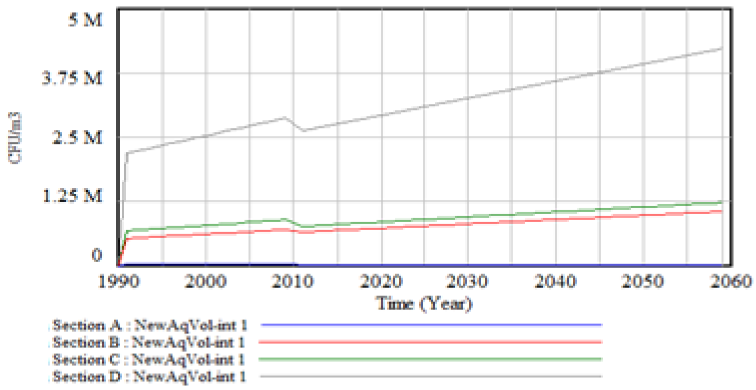


Figure 46 Intervention 1: FC concentrations in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of FC are: DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 3 log FC removal by improved ST. Other treatment practices as in baseline scenario: 3 log FC removal by WWTP for DU and DR; no treatment for LIV and AGR.

a. **NO3 removal.** NO3 removal increased from 0% to 30% removal in existing ST with no changes in WWTP. Results are shown in Figure 47.

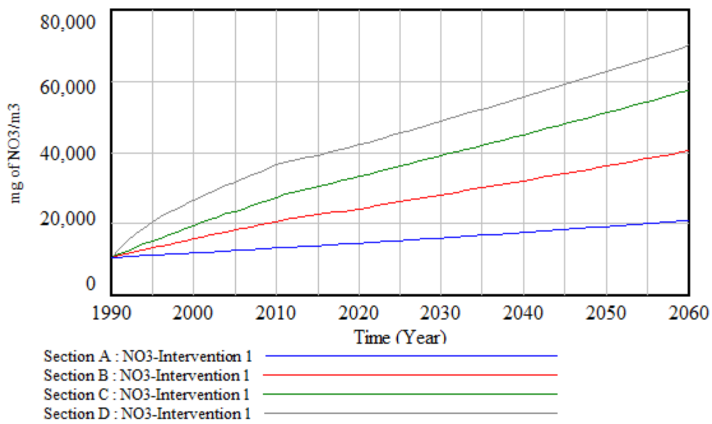


Figure 47 Intervention 1: NO_3 concentrations in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of NO_3 are: Aquifer background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 30% NO_3 removal by improved ST. Other treatment practices as in baseline scenario: 80% NO_3 removal by WWTP for DU and DR; no treatment for LIV and AGR.

By comparing FC concentration before and after intervention 1 since 2010 (Figure 48), it can be concluded that this intervention has a strong reducing effect only in aquifer section A, where 100% of wastewater is treated by ST, and contribution of livestock to FC is minor. By comparing NO_3 concentrations obtained before and after intervention 1 (Figure 48), results show a weaker tendency of nitrate concentration increase in the 4 aquifer sections. It is important to notice however, that as a consequence of intervention 1, nitrate concentrations in aquifer sections C and D reach the above Maximum Contaminant Level (MCL) for drinking water uses (MCL= 45 mg NO_3/l) about 10 years later than in the baseline scenario.

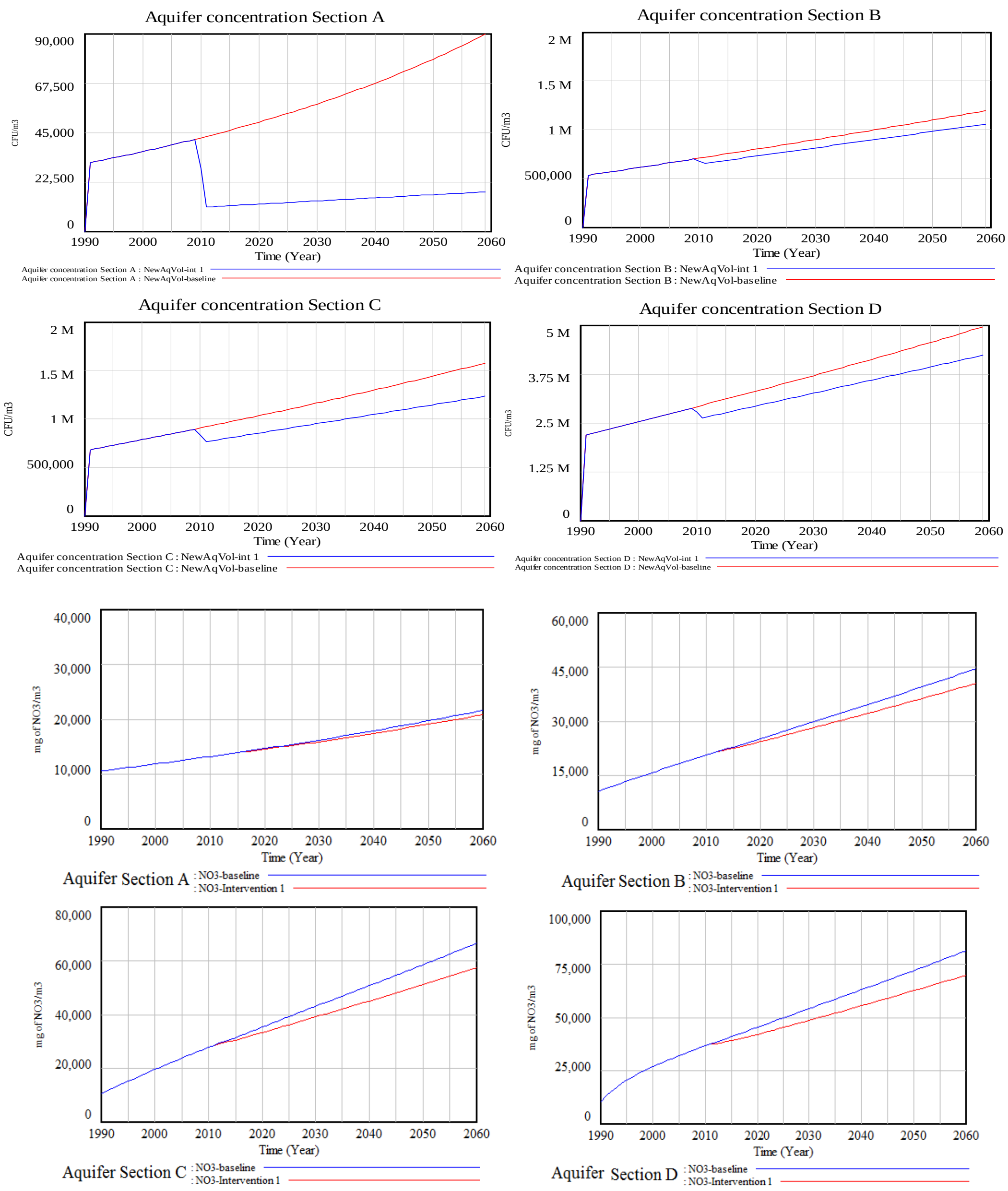


Figure 48 Intervention 1: Faecal coliform (FC-top) and nitrate (NO_3, bottom) concentrations comparison between baseline scenario and intervention 1, for each aquifer section.

Intervention 2. Connect all ST from DU and DR to WWTP (high cost). The aim of this intervention is to improve the overall quality of domestic effluents of existing ST, by building a sewerage system to connect these to the existing nearest WWTP for further treatment. Actions: There is the need to build drainage infrastructure at household level, in order to connect all ST to WWTP. Considering the karstic soil nature of the MAM, constructing such an infrastructure is expected to be challenging and therefore expensive. Nevertheless, the final outcomes of this intervention could justify the high costs. Outcomes: Final outcomes of this intervention are:

a. **FC removal.** FC removal increased from 1 log removal in existing ST to 3 log removal in existing WWTP. Figure 49 shows results and Figure 51-top shows a comparison with the baseline.

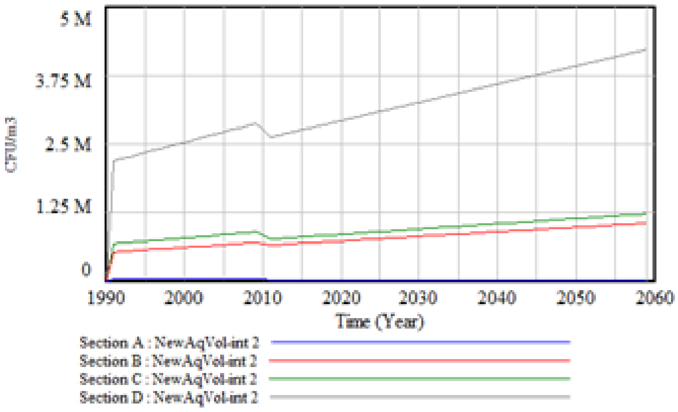


Figure 49 Intervention 2: FC concentrations in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of FC are: background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 3 log FC removals by connecting ST to WWTP in place for DU and DR; no treatment for LIV and AGR.

b. **NO3 removal.** Figure 50 shows results of this intervention for nitrate removal and Figure 51-bottom shows a comparison with baseline nitrate concentrations.

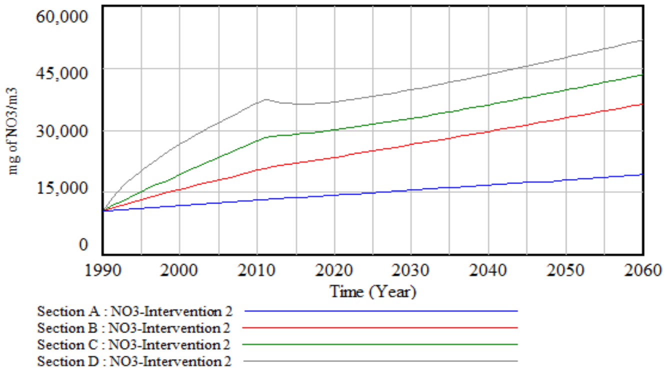


Figure 50 Intervention 2: Nitrate (NO_3) concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of NO_3 are: background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 80% NO_3 removal by connecting ST to WWTP in place for DU and DR; no treatment for LIV and AGR.

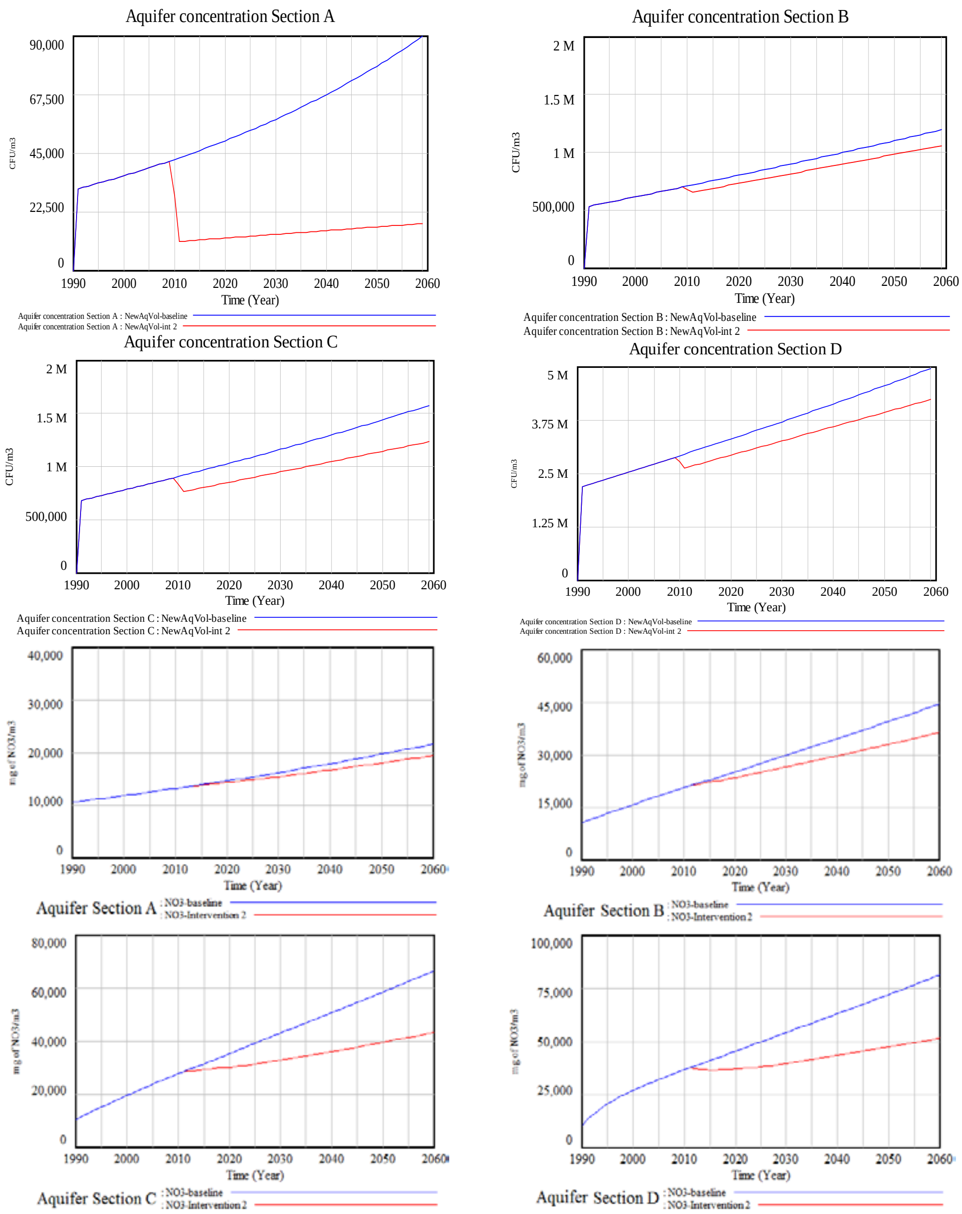


Figure 51 Intervention 2: Faecal coliform (FC-top) and nitrate (NO_3, bottom) concentrations comparison between baseline scenario and intervention 2, for each aquifer section of the MAM.

Intervention 3. Collect all ST from DU and DR to WWTP (low cost). The aim of this intervention is to reduce pollution from domestic wastewater by transporting with vacuum trucks the stored wastewater to nearby WWTPs for further treatment, eventually at a lower cost than with intervention 2. This improvement achieved by this intervention is highly dependent on the transport frequency. Even if pollutants concentrate in the sediment of the ST, and these sediments are removed once a year, as currently regulated in Merida at the municipality level, based on the national law (NOM-006-CAN-1997), the overall effect of nitrate and FC removal is not expected to be significant. Furthermore, complete transfer of released wastewater to treatment plants would, due to limited ST volume (typically ~1-2m^3, require a very high transport frequency of >50 per year. This may be not feasible in urban areas with high population density as in the MAM case study.

Actions: Provide an adequate infrastructure with vacuum trucks to collect and transport the domestic wastewater to the WWTP at the required frequency.

Outcomes: Final outcomes of this intervention would be the same as for intervention 2, given that all wastewater released into septic tanks is transported at high frequency to wastewater treatment plants. This may, however not be feasible in practice because of the high population density in the MAM area.

At low transportation frequency, only minor effects on pollutant reduction can be expected. But even an annual collection of the sediments in a ST would be beneficial in combination with intervention 1, since the transfer of sediments to the soil absorption system (SAS) would affect the proper function of the latter due to clogging of pores.

Intervention 4. The aim of this intervention is to improve the quality of domestic effluents from existing WWTP. Existing treatments are assumed to have 3 log removal of FC and 80% removal of NO_3. Actions: An improved performance of current WWTP is achieved by investing in qualified operators, continuous training, and a reliable monitoring system. Upon this intervention, the existing WWTP are assumed to operate at optimal conditions with the highest removal efficiency. Outcomes: These are:

FC removal. FC removal increases from 3 log removal to 6 log removal in existing WWTP with no changes in ST infrastructure. Modelling results are shown in Figure 52. Figure 54-top shows a comparison with baseline.

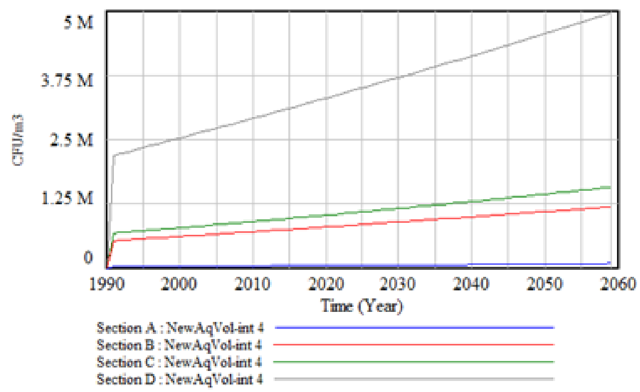


Figure 52 Intervention 4: FC concentrations in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of FC are: DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 6 log FC removal by WWTP in place for DU and DR. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

a. **NO_3 removal**. Increased from 80% to 100% removal by WWTP, with no changes in existing ST. Results are shown in Figure 53. Figure 54-bottom shows a comparison with baseline.

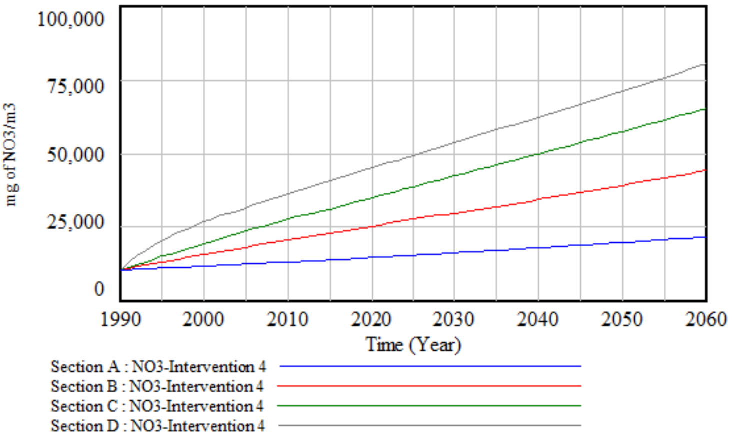


Figure 53 Intervention 4: Nitrate (NO_3) concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of NO_3 are: background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 100% NO_3 removal by improved WWTP for DU and DR. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

This intervention has very little effect since only a small portion of total wastewater from DU and DR is treated by WWTP. Consequently, improving these WWTP has little influence on the overall contaminant concentration in the aquifer

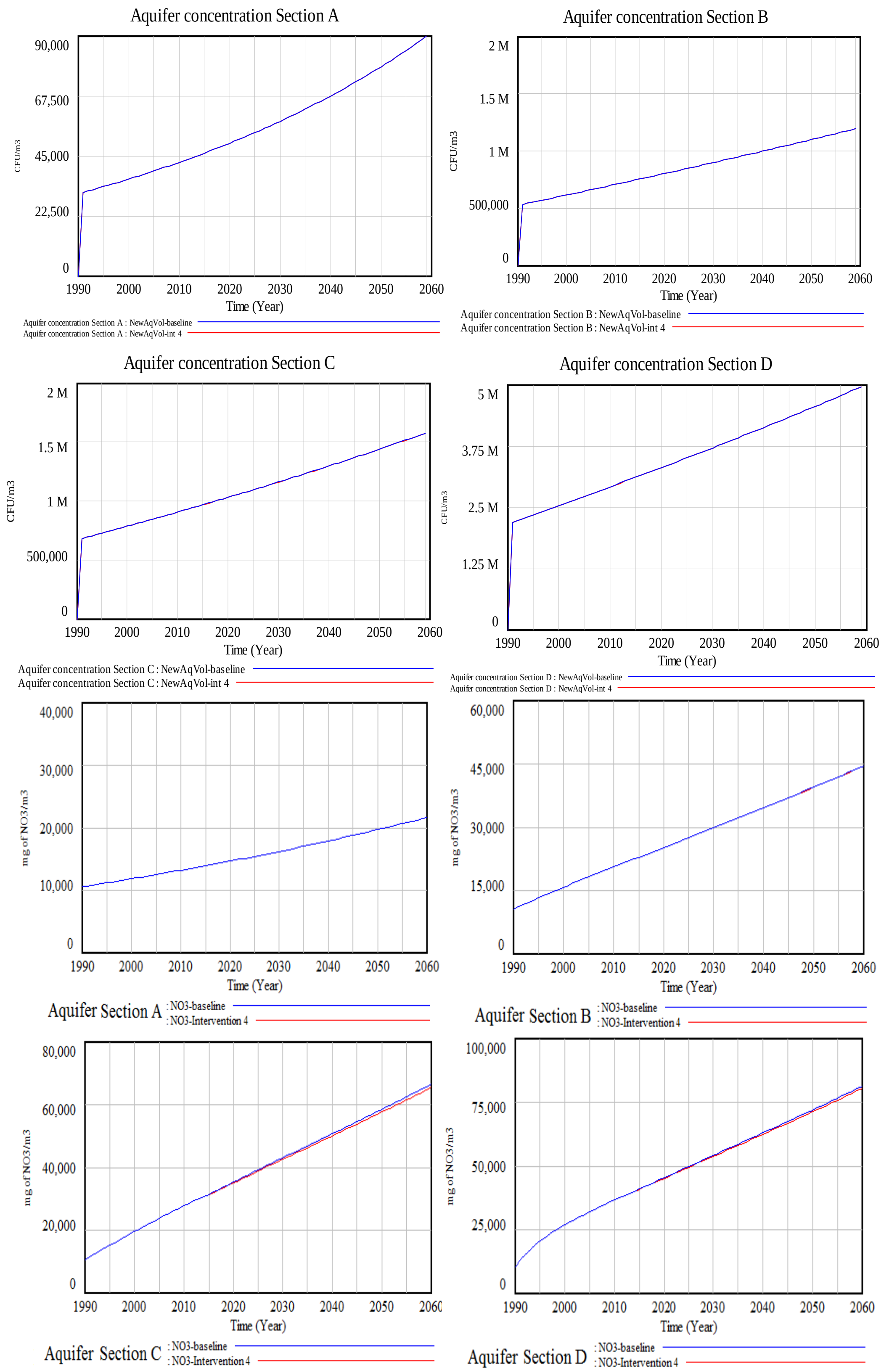


Figure 54 Intervention 4: Faecal coliforms (FC-top) and Nitrate (NO_3, bottom) concentrations comparison between baseline scenario and intervention 4, for each aquifer section of the MAM.

Intervention 5. The aim of this intervention is to improve the overall quality of domestic effluents by connecting all households to improved WWTP, and also increase the performance of existing WWTP in terms of pollutants removal efficiency.

Actions: There is a need to build new WWTP and to connect households to these WWTP by sewer pipes. In addition, the treatment efficiency of existing WWTP will be increased, as described for intervention 4. Outcomes: Final outcomes of this intervention are:

a. **FC removal.** FC removal increased from 1 and 3 log removal from ST and existing WWTP respectively, to 6 log removal due to wastewater collection and improved WWTP. Figure 55 shows results and Figure 57-top shows a comparison with baseline.

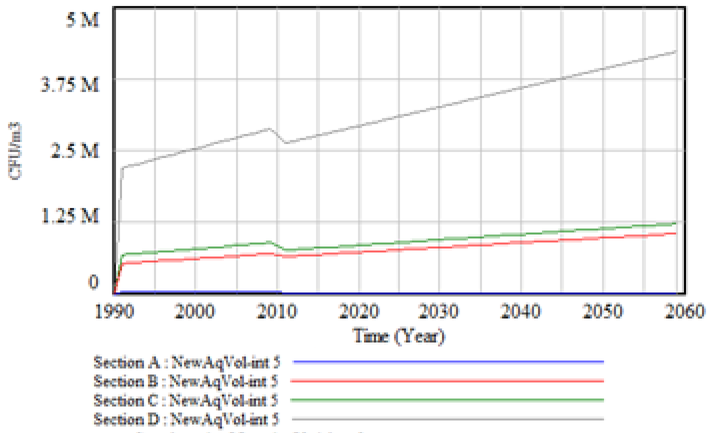


Figure 55 Intervention 5: FC concentrations in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of FC are: Domestic Urban, Domestic Rural, Agriculture, and livestock. Intervention starts in 2010: 6 log FC removals by wastewater collection from ST and improved WWTP for DU and DR. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

b. **NO_3 removal**. Increased from 0% in ST or 80% in existing WWTP respectively, to 100% removal due to wastewater collection and improved WWTP. Results are shown in Figure 56. Figure 57-bottom shows a comparison with baseline.

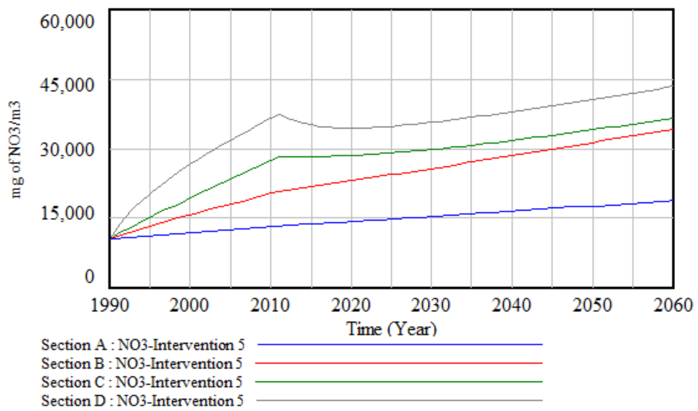


Figure 56 Intervention 5: Nitrate (NO_3) concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of NO_3 are: Aquifer background; Domestic Urban, Domestic Rural, Agriculture, and livestock. Intervention starts in 2010: 100% NO_3 removal by wastewater collection from ST and improved WWTP for DU and DR. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

There is a strong reduction of FC in section A where domestic sources are major contributors to FC contamination. The effect is relatively small in sections B-D where livestock is the major contributor to FC contamination. Importantly, nitrate levels in section C and D, which in the baseline scenario would exceed the MCL value 45 mg/L for drinking water within the next two decades, are kept below this value until 2060.

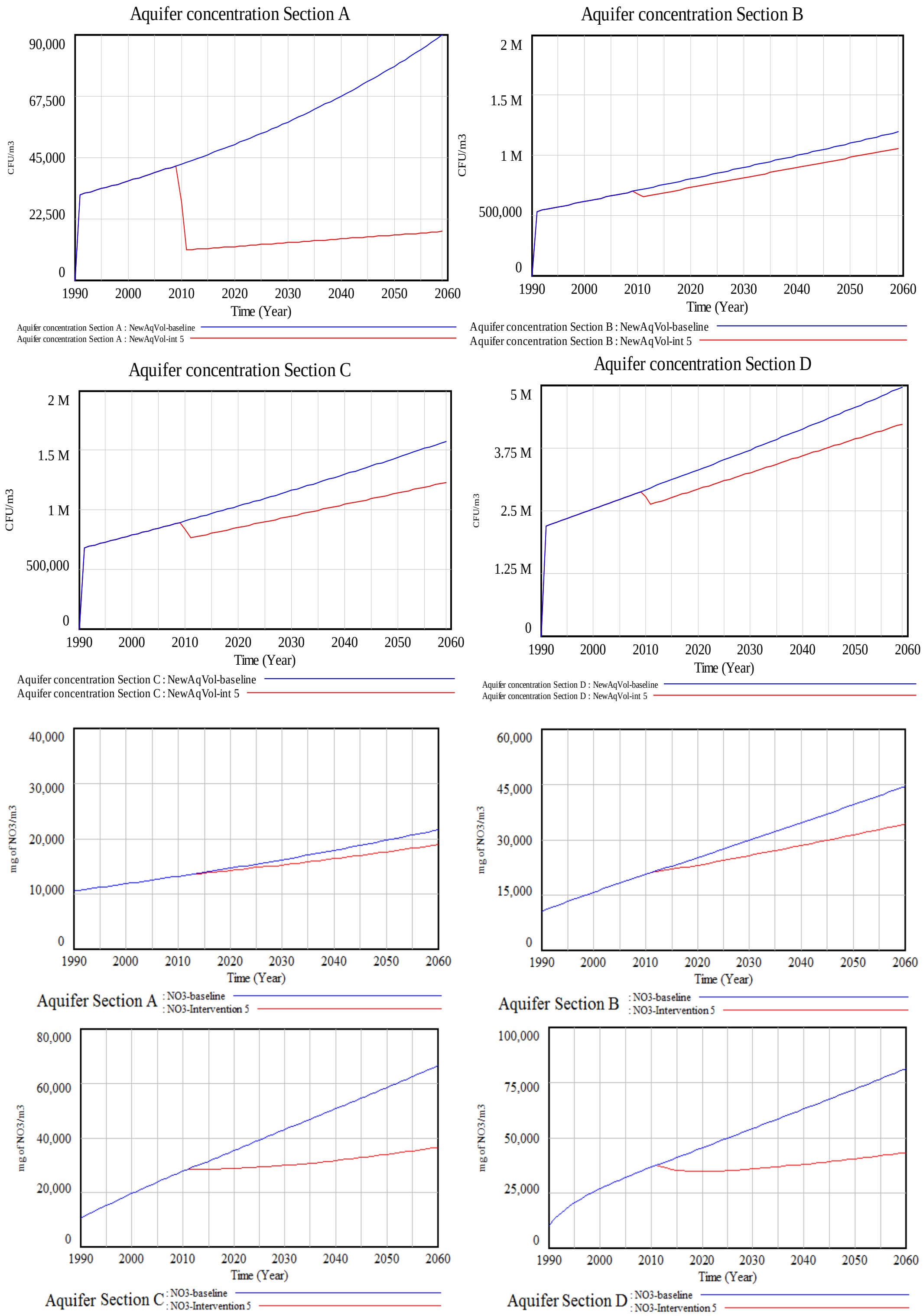


Figure 57 Intervention 5: Nitrate (NO_3) concentrations comparison between baseline scenario and intervention 5, for each aquifer section of the MAM.

Intervention 6. The aim of this intervention is to improve the overall quality of livestock effluents by creating WWTP for livestock, since there is no infrastructure currently in place for these effluents. Actions: There is the need to build WWTP infrastructure in place at farm level, in order to connect and treat all wastewater by WWTP. Considering the karstic soil conditions of the MAM, building the infrastructure required for this intervention could be expensive. Nevertheless, the final outcomes of this intervention could justify the costs at long term. Outcomes: Final outcomes of this intervention are:

a. **FC removal.** 6 log removals are assumed by WWTP. Modelling results are shown in Figure 58. Figure 60-top shows a comparison with baseline.

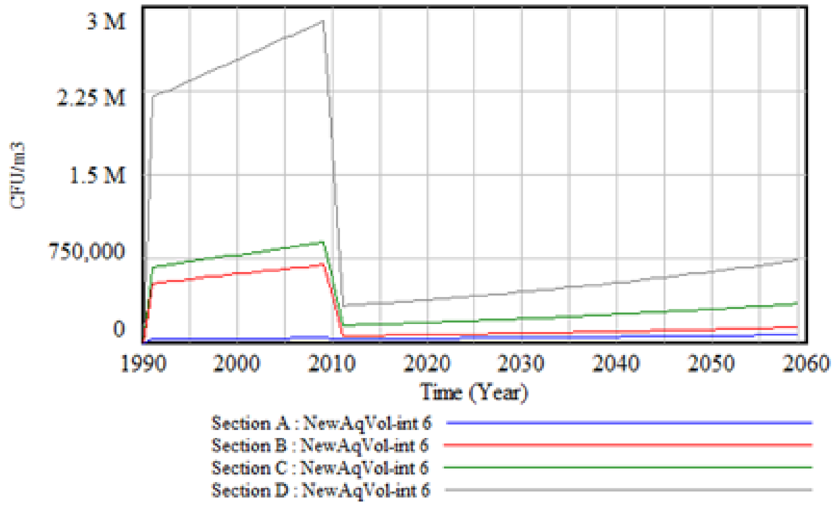


Figure 58 Intervention 6: FC concentrations in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of FC are: DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 6 log FC removals by WWTP in LIV. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

b. **NO_3 removal**. 100% removal is assumed by WWTP. Results are shown in Figure 59. Figure 60-bottom shows a comparison with baseline.

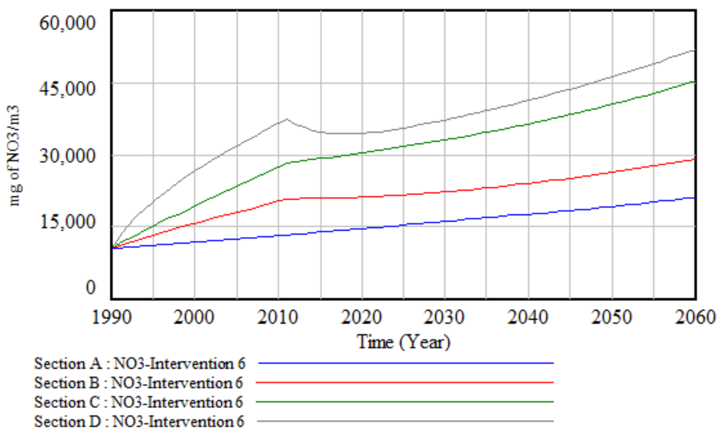


Figure 59 Intervention 6: Nitrate (NO_3) concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; sources of NO_3 are: Aquifer background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 100% NO_3 removal by WWTP. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

There is an instant >80% FC reduction in sections B-D where livestock is the major contributor to FC contamination. The effect is relatively small in section A where domestic sources are the major contributors to FC contamination. In addition, increase of nitrate levels in sections B-D slows down substantially, retarding the 45 mg/L MCL to 2045 (for D) or later.

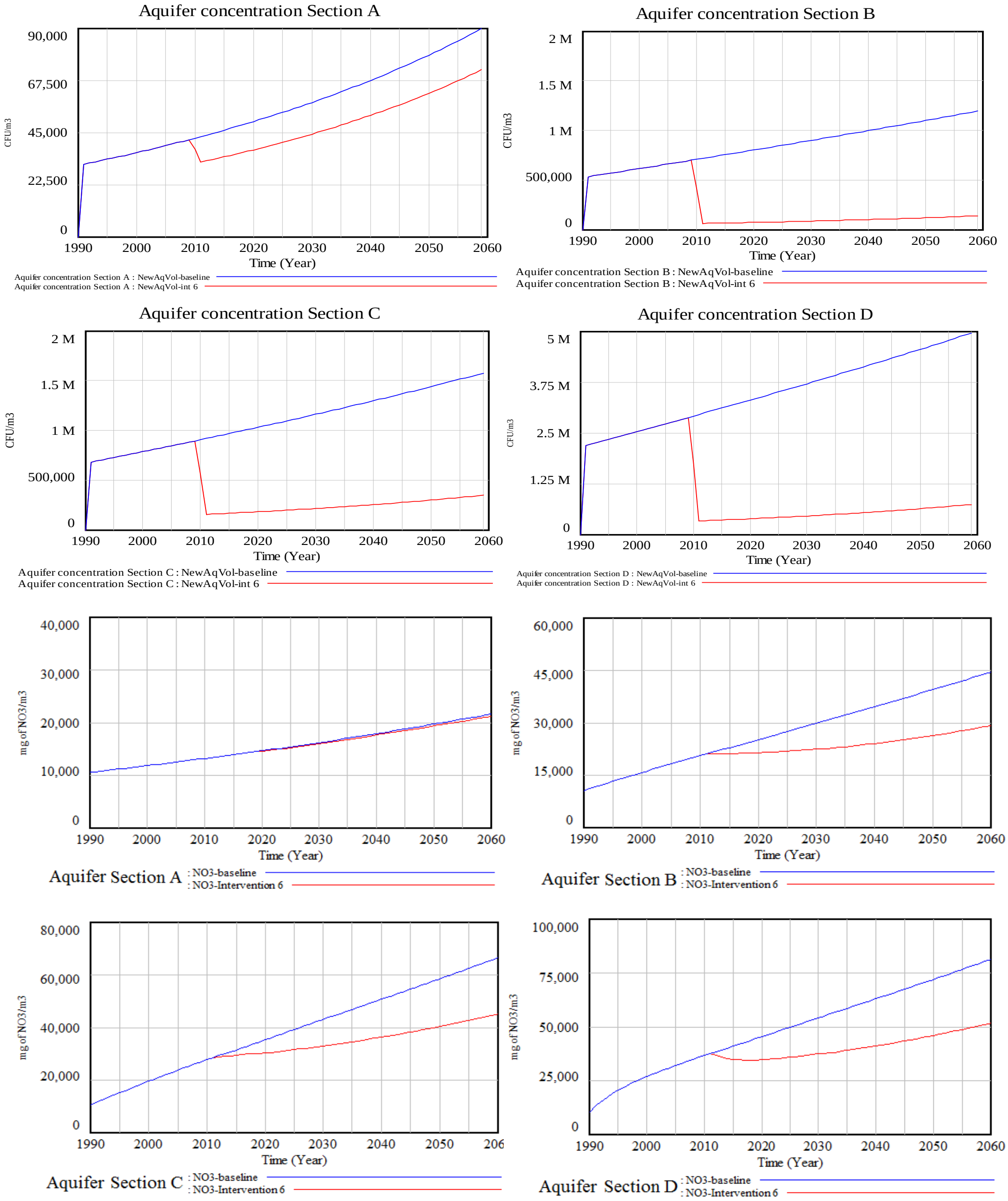


Figure 60 Intervention 6: Faecal coliform (FC-top) and nitrate (NO_3, bottom) concentrations comparison between baseline scenario and intervention 6 by aquifer section of the MAM.

Intervention 7. This intervention is focused on improving the management of nitrogen-based fertilizer use through the application of best management practices (BMPs) at the farming level. This is planned to reduce the overall leachate of nitrate to the aquifer from agriculture activity exceeding of 45 mg/L MCL to about 2045 (for D) or later. According to the Yucatan government statistics, there is no use of manure as fertilizer for agriculture; therefore there is no FC contamination from agriculture. Thus, this intervention is effective for nitrate removal only. Actions: Apply BMPs to nitrogen-based fertilizer use. Outcomes: Final outcomes of this intervention are:

NO_3 removal. According to literature reports (Shepherd and Chambers, 2007) through BMPs in agriculture at farm level, a 65% reduction of nitrate leaching can be achieved. Results are shown in Figure 61.

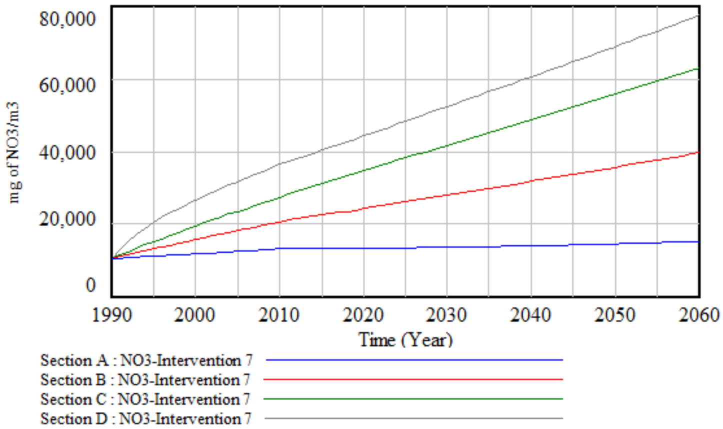


Figure 61 Intervention 7: Nitrate (NO_3) concentration in the 4 aquifer sections of the MAM. Conditions: constant population growth of 1.74%/y; source of NO_3 are: Aquifer background; DU: Domestic Urban, DR: Domestic Rural, AGR: Agriculture, and LIV: livestock. Intervention starts in 2010: 65% NO_3 removal by BMPs in agriculture. Other treatment practices as in baseline scenario: 1 log FC removal by ST for DU and DR; no treatment for LIV and AGR.

A comparison with baseline is shown in Figure 62. The nitrate levels in section A are almost kept at aquifer background levels, indicating that this intervention is very effective in this area where agriculture is the major contributor to nitrate contamination by wastewater. The intervention yields only insignificant results in section B-D where agriculture is only a minor contributor to nitrate contamination.

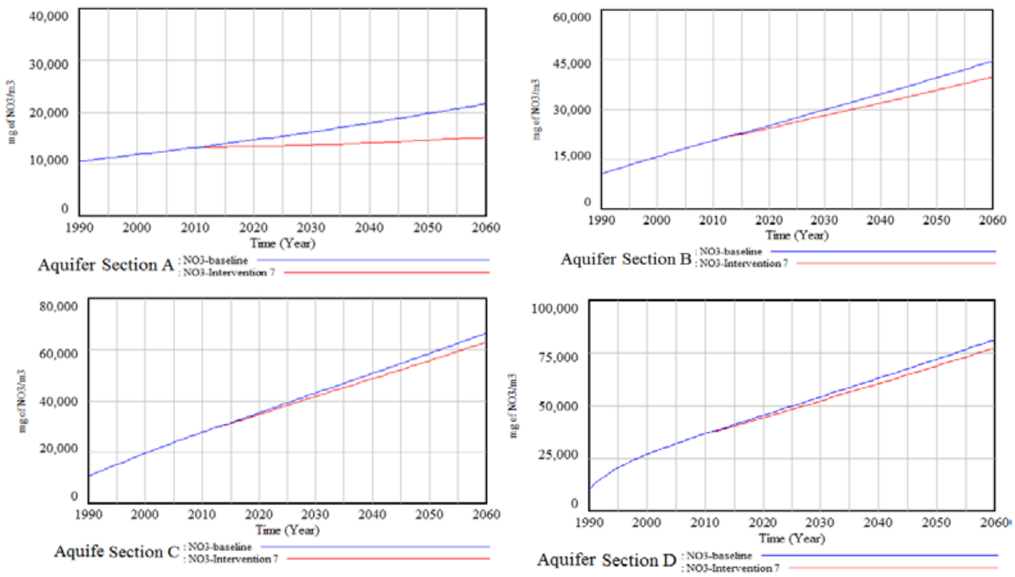


Figure 62 Intervention 7: Nitrate (NO_3) concentrations comparison between baseline scenario and intervention 7 by aquifer section of the MAM.

6.7. Comparison of interventions performance

In summary, the best results to reduce FC in the aquifer were obtained under the scenario of livestock intervention 6, after which 3 of the 4 aquifer sections have by far the lowest FC concentration at the end of the simulation period (Table 72). The only exception is aquifer section A what can be explained by considering the relatively insignificant livestock activity in this area. The highest reduction of nitrate in the aquifer is achieved under the scenario of intervention 5, where none of the 4 aquifer sections reach the MCL of 45mg NO_3/l at the end of the simulation period in 2060.

Table 72 Comparison of faecal coliforms and nitrate concentration at the end of the simulation period (2060) after interventions 1-7.

Intervention	Nitrate (mg/m^3)				Faecal Coliforms (CFU/m^3)			
	A	B	C	D	A	B	C	D
Baseline	2.17E+04	4.46E+04	6.66E+04	8.14E+04	9.09E+04	1.20E+06	1.59E+06	5.00E+06
1	2.09E+04	4.06E+04	5.75E+04	7.00E+04	1.81E+04	1.06E+06	1.24E+06	4.26E+06
2	1.96E+04	3.65E+04	4.35E+04	5.18E+04	1.81E+04	1.06E+06	1.24E+06	4.26E+06
3	1.96E+04	3.65E+04	4.35E+04	5.18E+04	1.81E+04	1.06E+06	1.24E+06	4.26E+06
4	2.17E+04	4.46E+04	6.56E+04	8.05E+04	9.09E+04	1.20E+06	1.59E+06	5.00E+06
5	1.90E+04	3.44E+04	3.67E+04	4.35E+04	1.79E+04	1.06E+06	1.24E+06	4.26E+06
6	2.13E+04	2.93E+04	4.54E+04	5.20E+04	7.30E+04	1.40E+05	3.53E+05	7.49E+05
7	1.52E+04	3.99E+04	6.31E+04	7.76E+04	N/A	N/A	N/A	N/A

*N/A: Not applicable

6.8. Selection of interventions for cost benefit analysis (CBA)

Interventions resulted in different levels of improvement in the aquifer water quality. The following graphs compare the performance of the 7 different interventions by individual aquifer sections. Based on Table 72, pollutants removal efficiency, display the following patterns:

For NO_3 reduction:

Aquifer section	Intervention
A	7 > 5 > 2&3 > 1 > 6 > 4
B	1 > 6 > 5 > 2&3 > 7 > 4
C	5 > 2&3 > 6 > 1 > 7 > 4
D	5 > 2&3 > 6 > 1 > 7 > 4

For FC reduction:

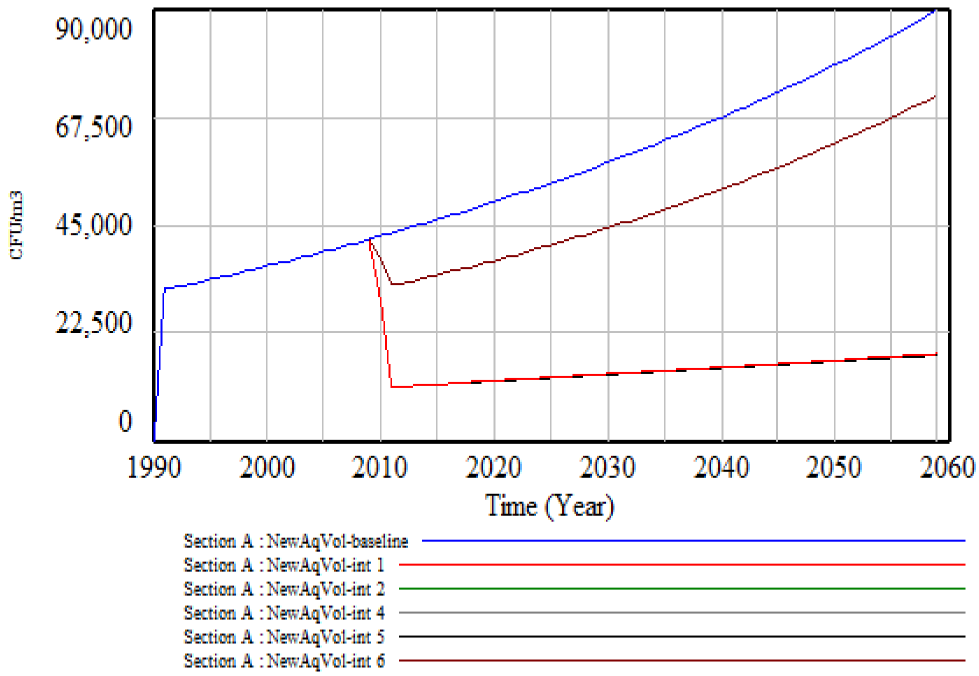
Aquifer section	Intervention
A	5 >1, 2&3 >6 > 4
B	6 > 5, 1, 2&3 > 4
C	6 > 5, 1, 2&3 > 4
D	6 > 5, 1, 2&3 > 4

For nitrate removal over all aquifer sections, interventions 5, 1 and 6 are the most effective. Meanwhile, for FC removal interventions 6, 5 and 1 in this order are the most effective. Considering their feasibility and long-term impact, the interventions 1, 5 and 6 were selected to carry out CBA in Chapter 7.

Aquifer section A.

a. FC reduction in groundwater is presented in Figure 63-top. Most effective intervention is intervention 6. Other interventions have limited effectiveness.

b. NO_3 reduction in groundwater is presented in Figure 63-bottom. Most effective intervention is 7 which aimed to reduce N-based fertilizers in agriculture.



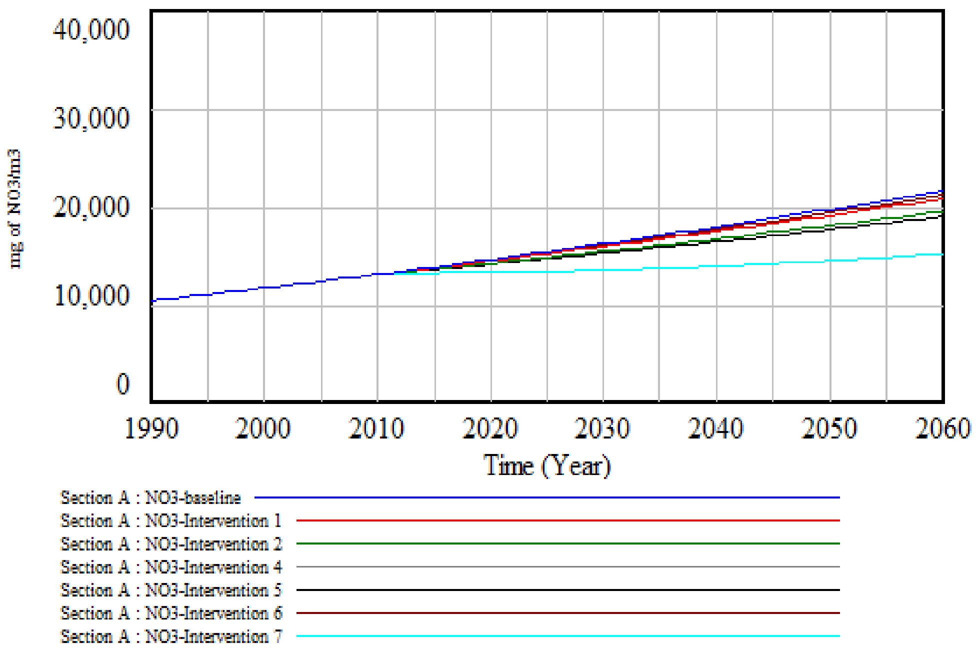


Figure 63 Aquifer section A. Faecal coliform (FC) and nitrate (NO_3) concentrations comparison between baseline scenario and the 6 relevant interventions in the aquifer section A of the MAM.

Aquifer section B

a. FC reduction in groundwater is presented in Figure 64-top. Most effective intervention is 6. Other interventions have limited effectiveness.

b. NO_3 reduction in groundwater is presented in Figure 64-bottom. Interventions reduce nitrate concentration in 2060 by up to 30%, with gradual differences in effectiveness. Most effective intervention is 6.

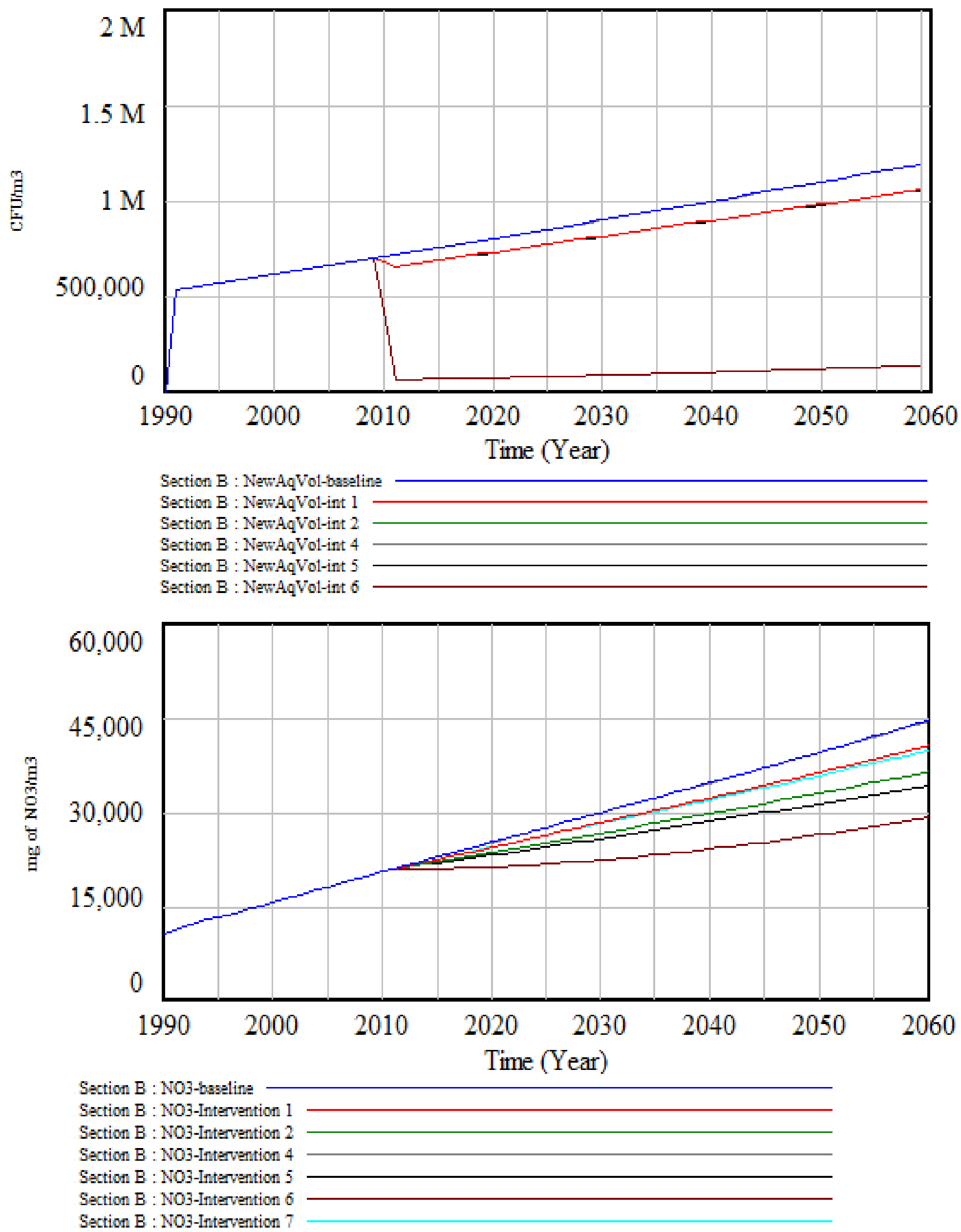
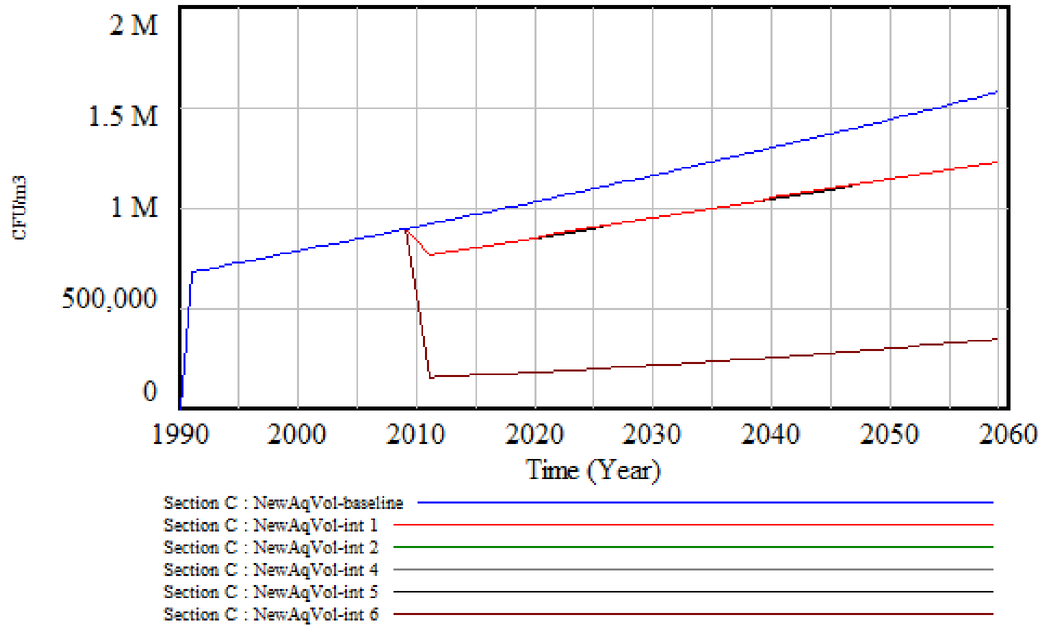


Figure 64 Aquifer section B. Faecal coliform (FC) and nitrate (NO_3) concentrations comparison between baseline scenario and the 6 relevant interventions in the aquifer section B of the MAM.

Aquifer section C

a. FC reduction in groundwater is presented in Figure 65-top. Most effective intervention is 6. Other interventions have limited effectiveness.

b. NO_3 reduction in groundwater is presented in Figure 65-bottom. Interventions reduce nitrate concentration in 2060 by up to 45%, with gradual differences in effectiveness. Most effective intervention is 5.



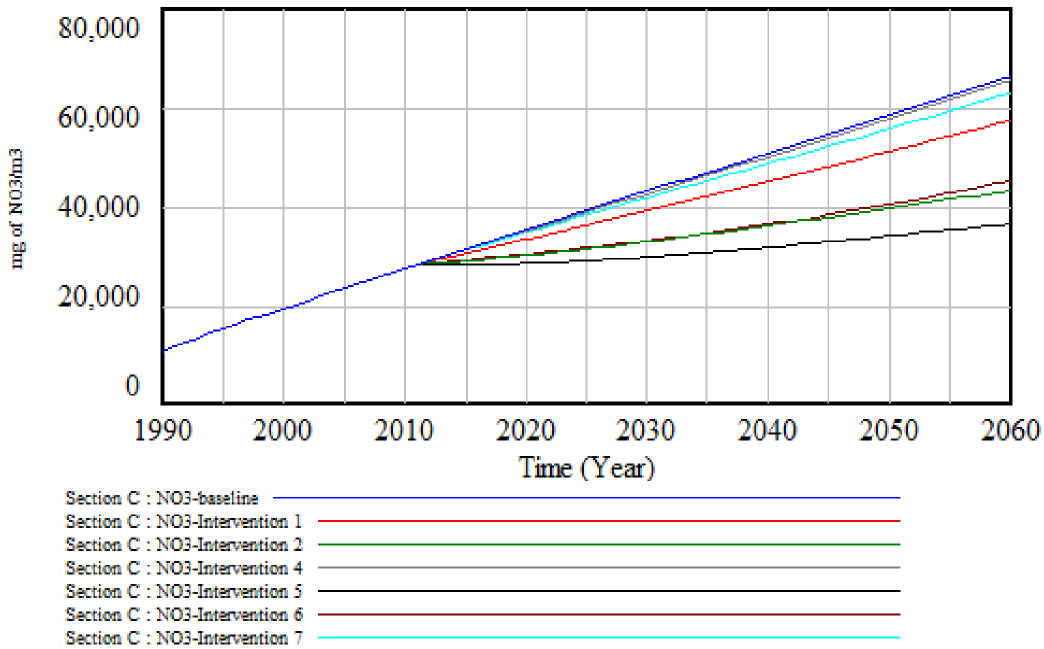


Figure 65 Aquifer section C. Faecal coliform (FC) and nitrate (NO_3) concentrations comparison between baseline scenario and the 6 effective interventions in the aquifer section C of the MAM.

Aquifer Section D

a. FC reduction in groundwater is presented in Figure 66-top. Most effective intervention is 6. All remaining interventions have limited effectiveness.

b. NO_3 reduction in groundwater is presented in Figure 66-bottom. Interventions reduce nitrate concentration in 2060 by up to 45%, with gradual differences in effectiveness. Most effective intervention is 5 for this aquifer section.

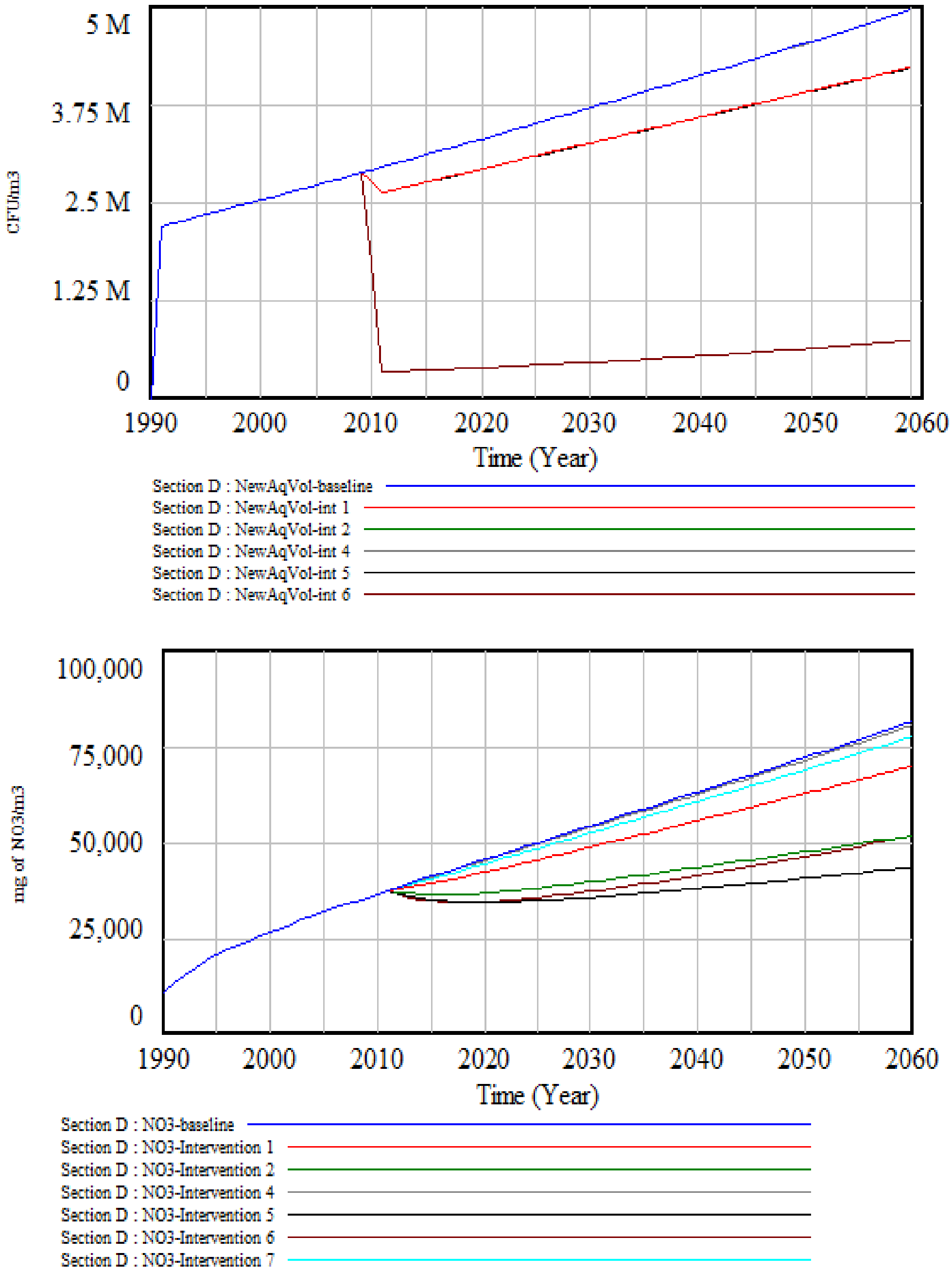


Figure 66 Aquifer section D. Faecal coliform (FC) concentrations comparison between baseline scenario and the 6 interventions in the aquifer section D of the MAM.

6.9. Analysis and discussion of model results

6.9.1. Nitrate and FC concentrations in the aquifer

The model simulates pollutant concentration in the karstic aquifer of the study area by assuming spontaneous homogeneous distribution of the wastewater-related pollutant load in the water volume of each aquifer section. For the conservative pollutant nitrate, this approach provides a reasonable approximation with the (limited) available field data in section A, B and C. The model underestimates nitrate levels in section D, which might be related to hydrogeological specificities of this aquifer section such as low thickness of the freshwater lens.

Modelling of Faecal Coliforms (FC) by the same approach initially gave a poor correlation with field data. Under consideration of the specific properties of this microbial pollutant, in particular the particle size and short lifetime, the aquifer volume was calibrated and a reduced volume, comprising 10% of the original volume of each aquifer section, applied to further modelling of FC concentration. This reflects the idea that FC is initially transported within small conduit volume and readily accessible fissures and larger pores. Their size and short lifetime does not allow FC to effectively diffuse through smaller pores into the limestone matrix, so that a major part of the aquifer volume is not accessible to FC. After volume calibration, the simulated FC levels are in good agreement with field data for sections B and C, and higher than field data for section A. Again, a poor agreement was found for section D.

6.9.2. Future scenario without interventions

The simulations already consider the limited existing infrastructure for wastewater management. Without additional wastewater management practice, the nitrate concentration is predicted to increase continuously by up to 1mg/L per year (section C) during the simulation time period, and exceed the MCL of 45 mg/L in section B by 2060 and in section C by 2035.

Increasing nitrate concentrations in groundwater are a worldwide problem. The general trend in Europe is an increase by about 0.4 mg/L per year (Beeson and Cook, 2004). Higher increases are reported at local sites, such as in the north of Valencia, Spain, where in average the nitrate level has increased by 2.6mg/L per year in the last decades (Perez et al., 2001).

FC concentration is predicted to almost double between 2010 and 2060 for aquifer section C of the study area. Increases of FC concentrations could lead to increases in disease outbreaks; nevertheless evidence to identify specific pathogenic pollutant is very limited when diarrhoea cases are reported. For instance, Craun et al., (2010) provided an overview of main pathogens causing outbreaks associated with drinking water in the United States from 1971 to 2006. Results of this review reported as non-legionella bacteria (which include the FC group) as main pathogen cause of a total of

113 outbreak from which, 105 outbreaks were directly caused by drinking water. From all outbreaks reported over the 35 years period of study, only 12 (11.4%) were identified for *E. coli* as the sole pathogen which caused the outbreak.

6.9.3. Effect of interventions

A long term reduction of NO_3 concentration in the aquifer by up to 45% is achieved by the most effective interventions 5 and 6, relating to nitrogen release from domestic and livestock sources. An important issue for decision makers is the delayed response of nitrate concentration on the interventions. For example, a reduction of nitrate level by 20% may take about 10 years (Figure 34). This delay is confirmed by Katz et al., (2014) for a karst aquifer in south-western Georgia, USA, where the simulation of flushing-out nitrate to halve its concentration takes several decades. Similar delay was predicted in the basalt aquifer in south-central Idaho, USA (Skinner and Rupert, 2012). Considering that the planning and implementation of the interventions may take several years, a long-term planning of interventions for nitrate reduction is of particular importance.

In contrast, response of FC levels to the interventions is fast, as expected from the short lifetime of these microorganisms. Model results for FC concentration follows similar pattern when comparing the effect of the different intervention as follows: for aquifer section B-D interventions effect is higher for intervention 6, relating to livestock wastewater, than for all the other interventions. The exception is in aquifer section A, where livestock activity is insignificant and the effect of the interventions is contrary to the above pattern, meaning other interventions results in higher reduction than intervention 6. This confirms the adequacy of the model approach used in this research. Nonetheless, FC results are subject to many objections due to the lack of knowledge in terms of processes that control bacteria survival and movement in the soil and in groundwater as discussed by Benham et al., (2006), which limits the applicability to a watershed-scale model.

Based on the literature, there are two main types of interventions widely used to improve drinking water quality for preventing diarrhoea: at the water source level and at the household level. Although the three interventions selected in this thesis to perform CBA only covers the household level intervention, this type of intervention has been documented as more efficient than interventions at the water source level (Chasen et al., 2007).

6.9.4. Seasonal variations

Seasonal variation of nitrate levels in deeper sections of the aquifer is considered insignificant in the simulation. This is confirmed by continuous monitoring of nitrate in an individual, deep water supply well in section C, with no evidence of a seasonal pattern (CONAGUA, 2008). In terms of seasonal variation significance, a similar study developed by Elci and Polat, (2011) for the karstic area of Turkey, examines the seasonal variability of the groundwater quality in this system. It was also concluded that no statistically significant changes were observed during season change from all different quality indicators monitored (electrical conductivity, nitrate, chloride, sulphate, sodium, and heavy metals). Nevertheless, a local and temporal variability of nitrate levels was observed. From these findings, two main conclusions could be drawn and are principally transferable to the interpretation of the model results for the MAM study:

1. Comparing single-site field data with model results has limitations, thus should be carefully interpreted.
2. Seasonal variability is case-specific, as mentioned by Elci and Polat, (2011) dry season not essentially results in higher concentration of all groundwater quality indicators due to the effect of other conditions that also plays an important role, such as water circulation times, lithology, recharge, and land use patterns.

Seasonal variation of FC levels in the MAM can be very significant according to field data, in particular in shallow wells. It was shown that a seasonal variation of FC- low in dry period and high in rain period- can in principle be modelled by relating FC load to precipitation, assuming that FC released on the surface is flushed more effectively in the aquifer by more rainfall. A practical implication for decision makers could, for instance, be the development of seasonal based effluent limits for wastewater treatment facilities, with season-dependent implementations of more effective disinfection procedure or even implement filter devices.

6.9.5. Intervention feasibility and timescale of implementation

As already indicated in section 6.8, feasibility and effectiveness are not the same for the seven suggested interventions. Interventions 2 and 3 have been evaluated by the Yucatan Government and the scientific community to some extent but are not considered feasible (Osorio, Personal communication, 2013). Building an extended drainage structure in the karstic soil of the MAM would be a project with unforeseeable costs. Concerning intervention 3, it is very questionable if a high-frequency transportation of wastewater from septic tanks to WWTP by vacuum trucks is a practical solution for the study area. Intervention 4 (improving efficiency of existing WWTP), resulted in too little benefit with respect to water quality based on the simulation results. Intervention 7 (BMPs for N-based fertilizers) improves groundwater quality in aquifer section A, but only related to nitrate levels. Section A is the area with the highest agricultural activity of the study area, and intervention 7 would have only little effect to the other aquifer sections. In conclusion, the most promising in terms of effectiveness and feasibility are interventions 1, 5 and 6, which were selected for further analysis.

Selected interventions 1, 5 and 6 have benefits for both public health and costs-saving due to nitrate pollution averted. In addition, these interventions were considered feasible from the viewpoint of:

- existing information for implementation of the interventions found on the literature (from similar cases studies)
- reasonable timescale to construct and start operating within the timescale of the proposed intervention (50 years), as show in Table 73
- available workforce, personnel training and raw material in place for implementation of the interventions
- The planning phase prior to effective implementation of the intervention was relatively short (considering some pilot projects such as for intervention 6)

Table 73 Typical timescale required to implement new infrastructure for selected interventions

Infrastructure stage	Time frame	Year of effective implementation of intervention (starting in 2010)
Planning stage to modify existing ST with SAS technology	3 month	
Installation of SAS	12 months	
Building connection to ST	2 months	2012 for intervention 1
Planning of WWTP construction	6 months	
Construction of WWTPs	23 months	
Interconnection with sewerage system	2 months	
Total time to implement WWTP	31 months	2013 for intervention 5 and 6

Source: Estimation based on CONAGUA, (2007); and USEPA, (1999).

For practical purposes to evaluate the different interventions at the same timescale, the effects (benefits) of all interventions were assumed to start in 2010.

6.9.6. Benefits for decision making and practical implications of interventions

In practice, all interventions scenarios simulated in this research are subject to a variety of factors including:

- Life-time of the project. The time scale of the project is 50 years, with interventions effectively starting in 2010. Majority of these interventions would require construction up to several years and consequently the benefits would be seen with a delay, but some would be accountable after the first year. For instance, intervention 7 where best management practice in agriculture would account after the first year, conditioned to crops season, harvesting period, and fertilization practice subject of change.
- As discussed in the costs estimation section, majority of these interventions would require capital cost coverage by the government, which would be subject to annual budget and change of government parties may also affect the

continuity of the projects. These would again results in an uncertainty about how feasible these interventions would be in practice.

Benefits for decision making are diverse, from adequate allocation of diverse land use socio economic activities, up to the control and preservation of natural water resources exclusively confined to water supply for primary uses (drinking water). Some of the main benefits for decision making are shown Table 74. These are achieved through the modelling approach used, which integrates the latest trends in hydro-economic models reviewed by Booker et al., (2012). These include among others the incorporation of modelling competing demands, governance and institutional conditions (i.e. laws, regulations and water quality standards), scenario analysis for the selection of engineering interventions, and economic analysis to identify the most cost-benefit.

Table 74 Benefits of implementing model framework for decision making

Area	Benefit	How to implement
Public urban	Planed urbanization through considering population growth	By projecting population growth and water demand, coupled by location of water resources and water supply capacity, decision makers could project the future urban development safeguarding adequate water supply quantity and quality
Water supply	Protection and prevention of water quality deterioration	Baseline conditions should be simulated together with an intervention, restricting the peripheral area of the main catchment system
Catchment	Guaranteed water table level, thus sufficient water quantity under climate change conditions	Water quantity could be projected under current conditions for baseline. Then an intervention for climate change conditions (extreme events) could be implemented

Although, if the main aim for other cases scenarios is to test adaptive solutions to climate change, a resilient approach could easily be implemented through new resilience scenarios simulations. Outcomes could serve for investment decisions using CBA as described by Nkomo and Bernard, (2006), by calculating among others: climate change damages, net benefits of adaptation, costs of preventive engineering interventions (cost of precaution), as well as net benefit of the interventions as follows:

a) Climate change damages. “Net economic loss in net welfare due to physical damages of climate change compared to the baseline case”.
b) Net benefits of adaptation. “Net reduction in climate change damages due to optimal capital investment for climate change interventions”. This is divided in:
    a. Climate change benefits: damages avoided by adaptation actions
    b. Climate change costs: costs of resources used for adaptation actions
c) Costs of precaution/caution. “Costs assuming climate will/ will not change, respectively and making the capital investment.
d) Net benefits of implementing adaptive interventions. “Measurement of the climate change damages avoided by adapting to climate change”.

Even though, resilience interventions for climate change adaptation would be planned under uncertainties (i.e. precise nature of climate variability, changes over long-term period), by using the framework developed by Callaway, (2004) to estimate costs and benefits of adaptive interventions to climate change (Nkomo and Bernard, 2006).

Chapter 7. Results of cost-benefit analysis of interventions

"The greatest risk to public health from microbes in water is associated with consumption of drinking-water that is contaminated with human and animal excreta, although other sources and routes of exposure may also be significant" (WHO, 2011).

7.1. Cost-benefit analysis

To analyse the relative cost-effectiveness of each intervention, model results were used to estimate health gains in the study population in each case. These could then be compared to the costs of each intervention. Health benefits were estimated in terms of disability-adjusted life years (DALYs) gained and quality-adjusted life years (QALYs) as result of improvements in water quality resulting from the proposed interventions.

A cost-benefit analysis (CBA) was carried out in order to compare the cost effectiveness of most promising interventions and propose the most suitable for the MAM region. Costs and benefits listed in Table 75 were considered over a 50-year period (2010-2060).

Table 75 Costs and benefits to calculate for the MAM case study

Costs	**Benefits**
Costs of Interventions- section 7.3.	Economic value of the health life gained due to reduced diarrhoea incidence –sections 7.2.1.
	Economic value of treatment saved for removing excessive nitrate, prior to use as drinking water- section 7.2.2.

The costs of interventions included capital costs (installation) and operation and maintenance costs (O&M), as mentioned in Figure 30 (Chapter 5). Capital maintenance was excluded due to lack of data. In the benefit analysis, Quantitative Microbial Risk Assessment (QMRA) was used to estimate the change in incidence of diarrhoea associated with each intervention. This was then converted to an estimate of Disability-Adjusted Life Years (DALYs) saved, which in turn was used to estimate the economic value of the health gains. Diarrhoeal disease incidence was assumed to be associated predominantly with the presence in water supplies of *E. coli*. Faecal coliforms (FC) are a widely accepted indicator of microbial water quality and faecal contamination (WHO 2011), and also considered a good indicator of bacterial pathogens*. E. coli* constitute about 95% of FC and for routine purposes, FC and *E. coli* may be regarded as generally equivalent indicators of faecal pollution (Dufour, 1997; Allen and Edberg, 1995; WHO, 1996; Howard et al., 2006). Thus, for practical purposes FC was used in this thesis as equivalent of pathogenic *E. coli* in the QMRA. In addition, the economic benefit from the savings for the removal of excessive nitrate in drinking water was considered.

The impact of interventions was measured in the third segment of the modelled aquifer (section C) that displays not only the highest population but also the highest concentration of domestic and livestock pollutant sources. Impacts were assessed for the whole population of this area assuming their exposure to drinking water of the same quality as the resultant aquifer water quality at baseline and after modelling each intervention. From the 7 interventions simulated in Chapter 6, intervention 1 (improve existing septic tanks), 5 (increase the number of domestic WWTP and connect all households), and 6 (create new WWTP for livestock) were selected for the cost-benefit analysis considering their feasibility and long-term impact on water quality in aquifer section C as discussed in section 6.8.

This chapter describes costs and benefits and estimations for the 50 years of intervention lifetime. At the end of this chapter, cost-benefit ratio (B/C) of each intervention provides the criteria (ratio >1) to select the most adequate intervention.

7.2. Benefits estimates

Benefits of the proposed interventions in this research are measured in terms of:

a) Economic value of the healthy life gained associated with a reduction in diarrhoeal disease. This benefit is measured through QMRA for pathogenic *E. coli* (assumed as equivalent of FC), and
b) Economic value of NO_3 removal treatment averted (when excessive NO_3 is present in the aquifer prior to use as drinking water). This benefit is estimated by calculating the cost of treatment that would be required to reduce nitrate concentration to an equivalent level prior to delivery in each case.

7.2.1. Benefit a) Economic value of the health gains by reduction in diarrhoeal disease

QMRA was applied to estimate diarrhoea diseases caused by faecal coliforms (FC) in drinking water (assuming direct water consumption from the MAM aquifer). As FC is the microbial indicator for microbial water quality (WHO, 2011) and field data and infectivity parameters are often expressed in *E. coli*, FC was used as equivalent to pathogenic *E. coli*, as discussed above (section 7.1.). Based on WHO, (2011), a risk assessment process for FC in the MAM comprises 4 steps (Table 76):

Table 76 Quantitative Microbial Risk Assessment (QMRA) for FC in the MAM

Step	QMRA
Hazards identification	Diarrhea incidence due to high concentration of FC in the aquifer
Exposure assessment	Aquifer FC concentration ingested as drinking water
Hazard characterisation	Dose-response equation
Risk characterization	Infection risks estimation

Note: Quantitative Microbial Risk Assessment (QMRA) FC was evaluated in term of *E. coli*.

Annual diarrhoea infection risk per person in the study area (hereby: aquifer section C) by exposure to FC taken up with drinking water was calculated through QMRA. Groundwater concentrations of FC modelling in Chapter 6 were used, with a baseline concentration in aquifer section C of 91 CFU/100 mL in 2010. Then, a “worst case” scenario was applied, considering use of groundwater as drinking water without prior treatment. This reflects to some extent the actual situation that an estimated 20% of the population, in particular in suburban and rural areas of the study area, directly abstract drinking water from nearby shallow wells rather than relying on public supply (Alonzo and Acosta 2003).

7.2.1.1. Daily dose of FC: dd(FC)

The daily faecal coliform dose dd(FC), ingested per person by consumption of drinking water, was calculated by the equation

$$dd(FC) = c(FC) * DWpppd \qquad \text{Equation 3}$$

dd(FC): daily FC dose per person, unit CFU/(person*day)

c(FC): groundwater FC concentration in aquifer section C, as modelled in Chapter 6, unit CFU/100 mL

DWpppd: average consumed drinking water per person and day, unit L/(person*day)

DWpppd was assumed to be 1.4 L/(person*day) based on Haas et al., (1999). Using the baseline concentrations of FC for 2010, a dd(FC) of 1274 CFU/(person*day), respectively, was calculated.

7.2.1.2. Annual infection risk: $P_I(d)$

The infection risk upon single exposure to FC was calculated using the β-Poisson dose response equation. Dose-response relationship depends on the pathogen and ingestion pathway (e.g. by drinking water). For pathogenic bacteria (i.e. *E. coli*), the β-Poisson model usually provides a better match with observed infection risk than other approaches. It expresses the probability distribution of an infection based on two main parameters, a median infectious dose (N_{50}) and a slope parameter (α) as follows (WHO, 2001):

$$P_I(d) = 1 - \left(1 + \frac{d}{N_{50}}\left(2^{1/\alpha} - 1\right)\right)^{-\alpha} \qquad \text{Equation 4}$$

$P_I(d)$: Infection risk of a person after single dose exposure

d: pathogen dose (here: dd(FC))

N_{50}: median infective dose, unit CFU

α: infectivity constant

Values used on the β-Poisson model are summarised in Table 77, and are specifically for diarrhoea infection due to pathogenic *E. coli* (considered equivalent to FC).

Table 77 Values used to estimate annual infection risk with the β-Poisson model

Parameter	Value (unit)	Reference
N_{50}	$8.6*10^7$ (CFU)	Haas et al., (1999)
α	0.1778	Haas et al., (1999)
d	1274 (CFU/person*day)	Estimated

Using these parameters, a single-dose infection risk of $1.27*10^{-4}$ is calculated. From that single (daily) dose response, the annual infection risk was derived using the equation:

$$P_{I(A)}(d) = 1 - [1 - P_I(d)]^n \qquad \text{Equation 5}$$

$P_{I(A)}(d)$: annual infection risk of a person

$P_I(d)$: Infection risk of a person after single dose exposure, as calculated above

n: number of exposures per year (here: 365)

By this equation, an annual infection risk $P_{I(A)}(d) = 4.52*10^{-2}$ is calculated for the reference year 2010. The calculated annual incidence of diarrhoea infections in study area aquifer section C (population 509419) is 23,080. The value is lower than the extrapolated incidence 33200 for aquifer section C, using the data of registered incidence of diarrhoea in Yucatan State (SSA, 2011). This may be explained by the fact that only a part of the registered diarrhoea cases origins from drinking water.

The calculations were repeated using water quality results from each run of the model, as a result of interventions 1, 5 and 6 and an equivalent annual infection risk was estimated in each case.

Table 78 shows the summary of water quality improved by each intervention, the associated number of diarrhoea infections, and the annual infection risk.

Table 78 Summary of water quality and the associated number of infections in 2010 by intervention

Intervention	Water quality (FC in CFU/m^3)	Number of infections	Annual infection risk PI(A)(d)
Baseline	9.1E+05	23080	$4.52*10^{-2}$
1	7.6E+05	19537	$3.82*10^{-2}$
5	7.6E+05	19490	$3.82*10^{-2}$
6	1.6E+05	4053	$7.93*10^{-3}$

7.2.1.3. DALYs

Disability Adjusted Life Years (DALY) for a disease or health condition are calculated as the sum of the Years of Life Lost (YLL) due to premature mortality in the population and the Years Lost due to Disability (YLD) for people living with the health condition or its consequences (WHO, 2011).

$$DALY = YLL + YLD \qquad \text{Equation 6}$$

YLL is obtained as the product of the number N of disease-related deaths, and lost years L between death age and life expectation:

$$YLL = N * L \qquad \text{Equation 7}$$

YLD is the product of the number I of disease incidence cases (derived from the annual infection risk $P_{I(A)}(d)$, as calculated above), the disability weight DW (also given as a "severity factor" of the disease), and the average duration of disability L:

$$YLD = I * DW * L \qquad \text{Equation 8}$$

For the cost-benefit analysis, it is reasonable to follow a recommendation of Hutton and Haller, (2004) and only consider the loss of economically productive years, i.e. the gap between death age and 60 years.

In view of a cost-benefit analysis, these general equations are usually adapted by implementing age-weighted disease incidence data and consideration of economically productive life years. The case study-specific evaluation of YLL and YLD was complicated by limited availability of such data. For Yucatan State, age-group specific data for diarrhoea incidence and diarrhoea related deaths are only fragmentary. Therefore, the available data for the average (over all age groups) diarrhoea-associated death risk in 2010, in combination with the mean age of Yucatan population (29 years) for 2010 and a 60 years cut-off age for economic productivity (Hutton and Haller, 2004), was used to derive YLL. The result was then adjusted by the use of an age weighting factor f, which was derived from the more complete country-wide Mexican database:

$$YLL = N * L = [(\text{population} * P_{I(A)}(d)) * (P_{D(A)})] * (60 - \text{mean age}) * f \qquad \text{Equation 9}$$

Population: population of aquifer section C in 2010 (509 419), (SSA, 2011)

$P_{I(A)}(d)$: annual diarrhoea infection risk of a person, as calculated above ($4.52*10^{-2}$ for aquifer section C in 2010)

$P_{D(A)}$: average annual death risk upon diarrhoea infection in Yucatan 2010 ($7.6*10^{-4}$)

Mean age: referring to the population of Yucatan state in 2010 (29 years)

f: age weighting factor (0.47)

When these Yucatan-specific values are substituted in the YLL equation, the latter modifies to:

$$YLL = [(\text{population} * P_{I(A)}(d)) * 7.6 * 10^{-4}] * 31 * 0.47 \qquad \text{Equation 10}$$

$P_{D(A)}$ was obtained as the ratio of registered diarrhoea-based deaths (94) and diarrhoea infections (124 424) in Yucatan State 2010 (SSA, 2011). The correction factor f was derived from the comparison of age-group specific, following the recommendations of Hutton and Haller, (2004), and age-averaged YLL calculations, using available data for the Mexican country and assuming a similar age-related distribution of diarrhoea incidence and diarrhoea-related deaths for Mexico and Yucatan. The age-averaged calculation overestimates the YLL since diarrhoea associated deaths is particular high in age groups 0-1 years and >65 years, which are not economically productive. YLD was calculated as follows:

$$YLD = I * DW * L = (population * P_{I(A)}(d)) * DW * (average\ duration\ of\ disease) \quad \text{Equation 11}$$

DW: disability weight or severity factor referring to diarrhoea, 0.105 (WHO, 2011)

Average duration of disease: referring to diarrhoea in Yucatan, 5.6 days or 0.0153 years (SEDUMA, 2009)

When these diarrhoea-specific values are substituted in the YLD equation, the latter modifies to:

$$YLD = (population * P_{I(A)}(d)) * 0.105 * 0.0153\ years \quad \text{Equation 12}$$

Here, the YLD for economically non-productive age-groups (<15 years and >60 years, about 30% of total population) was not excluded, taking into account that YLD will be overestimated. This is justified by the necessity to consider in the cost-benefit analysis also for these age groups the expenses for diarrhoea treatment, which make a significant contribution to the total disease-associated costs. In the disease-associated cost calculation, the overestimated loss of economic productivity will be down-corrected by using a correction factor for the economically productive portion of the population (see Equation 14).

Based on these equations and considerations, YLL = 225 years and YLD = 37 years are obtained for faecal coliform associated diarrhoea in aquifer section C for the reference year 2010 (Table 79).

Table 79 Summary of YLL and YLD estimates by intervention

Intervention	YLL	YLD
Baseline	255	37
1	216	31
5	215	31
6	44.8	6.5

7.2.1.4. Economic value gained by reducing diarrhoeal disease

For the calculation of diarrhoea-related economic value, YLL and YLD were considered. YLL was associated with an economic loss (salary loss), and YLD in addition with the treatment costs.

$$Economic\ value = (annual\ salary * YLL)+[(annual\ salary * YLD * f)+(treatment\ cost/year*YLD)] \quad \text{Equation 13}$$

Average salary per capita in Yucatan State was reported 150 Mexican Pesos/day (SEDUMA, 2009), corresponding to 54750 Mexican Pesos/year or 3997 USD/year, using a conversion factor 0.073.

Average treatment costs for a single diarrhoea infection sum up to 367 Mexican Pesos (SEDUMA, 2009). Considering 5.6 days duration of the disease, a treatment cost of 23917 Mexican Pesos or 1746 USD per year is assumed.

f is a correction factor that considers the “economically active” portion of the total population. In Yucatan, f = 0.64 (INEGI, 2012). This correction factor partially compensates the overestimated YLD value, as outlined above.

When these values are substituted in the economic value equation, the latter is modified as follows for the calculation of diarrhoea associated economic loss in the study area for a one-year period, reference year 2010:

$$\text{Economic value} = (3997\ \text{USD} * \text{YLL}) + [(3997\ \text{USD} * \text{YLD} * 0.64) + (1746\ \text{USD} * \text{YLD})] \quad \text{Equation 14}$$

Using YLL = 255 years and YLD = 37 years as calculated above, the annual diarrhoea-related economic value for aquifer section C (population 509 419) in 2010 sum up to 1,170,000 USD. Monetary benefit of interventions is given in Table 80.

Table 80 Annual economic loss by diarrhoeal disease in aquifer section C reference year 2010

Intervention	Economic loss (Million USD)	Economic benefit (Million USD)
Baseline	1.17	-
1	1.01	0.16
5	1.00	0.17
6	0.27	0.90

7.2.1.5. Diarrhoeal disease benefit by intervention

From the data estimate in section 7.2.1.4, the economic loss by diarrhoeal disease was extrapolated to the 50-years period of intervention (2010-2060), considering population growth-associated increase of baseline FC concentration in groundwater. For ease of comparison and consistency, constant 2010 prices were assumed in all calculations. Similarly, a constant salary of 3997 USD/year was applied for diarrhoeal related economic loss in the all intervention time period (2010-2060).

Table 81 Effect of interventions on faecal coliform concentration in groundwater c(FC) and on diarrhoea-related benefit for the 50-years period 2010-2060. Data refer to aquifer section C of the MAM study area, assuming an annual population growth by 1.7%

Intervention	c(FC) in groundwater CFU/m^3	Annual disease-related economic value (Million USD)	Disease-related benefits 2010-2060 (Million USD)
Baseline 2010	9.1E+05 (year 2010)	1.17	-
Baseline 2060	1.6E+06 (year 2060)	4.78	149
Intervention 1	1.2E+06 (year 2060)	3.67	31
Intervention 5	1.2E+06 (year 2060)	3.67	31
Intervention 6	3.5E+05 (year 2060)	1.1	115

Intervention 6 promises maximum economic benefit in the order of 115 Million USD, as summarised in Table 81. Disease-related benefits in this table were calculating as the cumulative benefit, when considering the average of annual economic values for 2010 and 2060 from Table 81 and multiplied it by the 50 years of intervention time period.

7.2.2. Benefit b) Economic value of NO_3 removal treatment averted

A second benefit from the selected interventions is the averted nitrate removal costs when nitrate concentration in the aquifer exceeds the national drinking water quality standards.

According to the International Agency for Research on Cancer (IARC), nitrate is classified in group 2A which group probable carcinogens to humans, specifically due to ingested nitrate or nitrite where endogenous nitrosation take place. However, there is no solid evidence in the literature yet to document the relationship between nitrate uptake and cancer risks in public health. Therefore, in this research nitrate pollution in drinking water has been evaluated from the view point of saving treatment investments for reducing nitrate concentration of the groundwater to an acceptable level prior to use as drinking water.

The approach taken to measure this benefit was based on the assumption that the government should guarantee water quality standards in supplying the population with drinking water. The exceeding nitrate concentration in the aquifer would be reached at different time under each intervention scenario as show in Table 82. Subtracting these times per intervention when nitrate concentration is exceeded from the time on the baseline scenario (or “no doing anything scenario”) when nitrate concentration is exceeded, the total years of nitrate removal treatment is obtained.

Table 82 Years of NO_3 concentration exceeding regulation by intervention (section C)

Intervention	Year when NO_3 concentration exceeds regulation	Number of years required to remove NO_3
Baseline	2032	28
1	2038	21
5	> 2060	0
6	> 2060	0

A rough cost estimate for the installation and operation of nitrate removal technologies has been done on the basis of Jensen et al., (2012). Electrodialysis reversal (EDR), as currently applied in California (USA), is a relatively cost effective nitrate removal option, but still requires average costs for both investment and operation and maintenance (O&M) of 1.6 USD per 1000 Gallons (or 3785 litres) of drinking water. Nitrate removal efficiency of 90% is assumed (GE, 2005). Due to the effectiveness of this technology in nitrate removal, treatment of only a portion of the drinking water would be required so that upon mixing of the purified water with untreated water, the MCL of 45mg/L can be maintained. This practice would minimize operation costs and was assumed to be applied for the case study. The volume fraction V_T/V_o of the water that needs to be treated to achieve an overall concentration below MCL increases along with increasing nitrate levels, according to the following equation:

$V_T/V_o = 1.11 - (1.11 * 45mg/L / C_o)$ Equation 15

V_o= total groundwater volume

V_T= treated groundwater volume

C_o= nitrate concentration in groundwater before treatment (>45 mg/L)

For section C, treatment would be initiated in the year 2032 when the MCL of nitrate is reached, and the portion of water that requires treatment according to the formula above increases from 2% in 2032 up to 40% in 2060. Considering a domestic water abstraction in 2010 of 2.2 m^3/s in section C, annual water consumption sums up to about 69 Billion litres. Along with population growth, it rises to 100 Billion litres in 2032 (onset of nitrate removal) and 160 Billion litres in 2060 (end of simulation). Considering treatment costs of 0.00042 USD/L, an economic loss of 840, 000 USD is predicted for 2032, and 27 Million USD for 2060, associated with nitrate removal. Over the 28 year period (2032-2060) of nitrate removal treatment, the economic loss would be sum up to 392 Million USD.

Table 83 summarizes the nitrate concentrations in 2060 depending on interventions at the wastewater level, and the economic benefits associated with intervention due to averted nitrate removal. The benefits are very significant for interventions 5 and 6 that keep nitrate levels below the MCL over the whole simulation period.

Table 83 NO_3 concentration in 2060 in aquifer section C, cumulated nitrate removal costs (onset when MCL is exceeded) and economic benefits associated with interventions

Intervention	NO_3 concentration in 2060 (mg/L)	Cost for NO_3 reduction (Million USD)	Economic benefit (Million USD)
Baseline	**70**	**392**	**-**
1	**58**	**108**	**284**
5	**38**	**-**	**392**
6	**44**	**-**	**392**

7.2.3. Summary of benefits

Table 84 shows the total benefits obtained by the three interventions selected. All interventions provide significant economic benefits of several hundred million USD. The overall most effective intervention is 6, which over the 50 years period of intervention, results in a total benefits of 507 Million USD.

Table 84 Summary of benefits from the selected interventions (over 50 years)

Intervention	Economic benefit gained by disease reduction (Million USD)	Economic benefit gained by avoiding NO_3 removal (Million USD)	Total economic benefit gained (Million USD)
1	31	284	315
5	31	392	423
6	115	392	507

7.3. Costs of interventions

General methodologies to estimate costs of engineering interventions have been documented by Vesilind and Rooke, (2003) and McGivney and Kawamura, (2008). The latter includes a more practical estimation by using nominal cost of different infrastructure. The method used in this research to estimate costs of interventions is based on the estimation of the full simulation period 2010-2060 and includes two main costs components to estimate: Capital cost, and Operation and Maintenance cost (O&M), all of these considering the time value of money, as described below.

- Capital cost: is one-time investment paid as bank loans, but because in the MAM case study these facilities are government-owned system, interests of the loans are neutralized by tax exemption for government projects.
- Operation and Maintenance costs (O&M). These includes labour costs, energy and supplies (i.e. chemicals).

To simplify costs calculations, zero discounting rate was applied for capital costs and O&M costs (Broome, 1992).

7.3.1. Summary of costs estimate for the selected interventions

Rough cost estimates were performed for the three selected interventions (1, 5 and 6) which for aquifer section C promise the most significant pollutant reduction over time, considering investment and O&M costs for a 50-year period between 2010 and 2060, without adjustment for inflation, as recommended by USEPA, (1999) for long lifetime wastewater projects.

The costs of intervention 1 (improvement of ST) consist on a capital investment for a soil absorption system (SAS) to connect to each ST, which has a unit cost of 400 USD (EPA, 1999). Assuming installation of the SAS in each household (4 people) of the MAM that has a ST, and taking into account population growth in aquifer section C, overall costs of 71 Million USD were derived for the full intervention time (2010-2060). O&M is not required for this technology, assuming adequate O&M of the ST.

The costs of intervention 5 (built new wastewater treatment plants and connect all households to new or existing plants) was simplified by associating the domestic wastewater release with a demand for modern treatment plants. This approach assumes a full replacement of the small number of less effective treatment plants currently in place. The costs for intervention 5 was thus derived from the domestic (1.73 m^3/s) wastewater produced in section C in 2010, and literature estimates for representative average costs by m^3 of treated water in a typical municipal treatment plant. OECD, (2006) provides such data and points out that the expenses for wastewater collection by connecting households to the plants (0.8 – 1.0 USD/m^3) are higher than the treatment costs itself (0.3 – 0.5 USD/m^3). The lower limit values 0.8 and 0.3 USD/m^3 of the given range were applied to the cost calculation in this research. With these data, annual costs for domestic wastewater treatment in 2010 are 60 Million USD, and 5.8 Billion USD for the 50-year period with consideration of 1.7% annual population (and consequently wastewater volume) increase. For wastewater treatment plants in Mexico with relatively large treatment capacity, lower values for treatment costs 0.15 USD/m^3 (Jimenez, 2008) and 0.13 USD/m^3 (Bartone, 2000) have been reported. Treatment costs per m^3 generally are the lower, the higher the capacity of the treatment plant is. It is particular difficult to estimate the costs for installation and maintenances of a pipe system that connects households in the study area to the treatment plants, since extra costs are expected due the karstic soil characteristics.

The costs estimation for intervention 6 is based on a study of the Yucatan government (SEDUMA, 2009), focusing on an projected central treatment plant for 40 pig farms (40000 pigs) in the MAM. Livestock wastewater is collected at the farm and transported by trucks to the treatment plant within a radius of 25 km. A lifetime of 50 years is assumed for the plant. Projected investment costs are 4.9 Million USD and operation and maintenance costs 1.8 Million USD/year. Since porcine is the major livestock activity in aquifer section C, the extrapolation of the data focused on porcine livestock and other activities were neglected. From the number of pigs (72000) in aquifer section C in 2010, it was concluded on a demand of 1.8 centralised treatment plants. The associated costs over the 50-year period, considering growth of livestock activity along with population growth (1.7%), are estimated 315 Million USD. Removal efficiencies for FC and nitrate are in the range of 99% (SEDUMA, 2009).

A summary of the estimated cost for the selected interventions in aquifer section C, over the 50-year period of intervention is presented in Table 85. The result is the net present value (NPV) of each intervention. Conversion to annual costs for financing of the interventions was estimated by dividing the total cost of the intervention by 50 years, the time period of the intervention.

Table 85 Costs estimate in Million USD over 50 years for the selected interventions

Parameter	Intervention 1	Intervention 5	Intervention 6
Capital costs ($)	71	2900	9
O&M costs ($)	-	2900	311
Total costs (2010-2060)	71	5800	320
Average annual costs	1.4	114	6.3

Note: costs are estimated assuming constant 2010 prices

For intervention 5 (new WWTPs) a 1:1 ratio of capital cost and O&M costs was used, based on data provided by The World Bank Group, (2015).

7.3.2. Per capita costs estimate for the selected interventions

Per capita costs for each selected intervention, both for the annual average and the cumulated 50 years costs (as given in Table 85) were estimated under the considerations mentioned above (neutralised inflation with government tax exemption) calculated for the 2010 population of aquifer section C (Table 86). Decreasing per capita costs with increasing population may represent a contribution to the Time Value of Money for the interventions investment.

Table 86 Per capita costs estimates (section C population 2010) for selected interventions, relating to averaged annual and 50 years cumulated costs of interventions

Intervention	Averaged annual per capita cost (in USD)	Cumulated 50 years costs (2010-2060) per capita (in USD)
1	2.7	139
5	230	11,159
6	13	626

7.3.3. Sensitivity analysis of costs estimate for the selected interventions

Sensitivity analysis was performed in a simplified approach for both the economic benefits and the costs associated with the selected interventions 1, 5 and 6 (Table 87 and Table 88). On the benefit side, the parameter under variation was the minimum wage which affects the economic gains by disease reduction. On the cost side, two parameters displayed in Table 85 were varied: capital costs and O&M costs of the selected interventions. Variations were within a range of ±10% and refer to the cumulated costs for the 50 years simulation period.

Table 87 Effect of variation of minimu wage on 50 years benefits of selected interventions

Variation	Intervention	50-years benefits (cost-savings) (Mio USD)		
		FC-related	NO_3-related	Total
Baseline	**1**	31	284	315
	5	31	392	423
	6	115	392	507
min wage -10%	**1**	28	284	312
	5	28	392	420
	6	104	392	496
min wage -5%	**1**	29	284	313
	5	30	392	422
	6	110	392	502
min wage +5%	**1**	32	284	316
	5	33	392	425
	6	121	392	513
min wage +10%	**1**	34	284	318
	5	34	392	426
	6	126	392	518

The sensitivity analysis from the benefit side is exemplified by the variation of the minimum wage in Table 87, indicating that a ±10% variation of minimum wage has a significant (±10%) effect on the benefits by avoiding FC-related disease, but only a minor effect (±2%) on the overall 50-year economic of the interventions that are driven by avoiding nitrate removal costs. The latter remains unaffected by variation of the minimum wage.

Similarly, results of sensitivity analysis from the costs side is exemplified by the variation of capital costs in relation to the 50 years costs of the interventions, which are presented in Table 88. Variations up to ±10% are observed.

Table 88 Effect of variation of capital costs and O&M costs on the 50 years costs of selected interventions

Variation	Intervention	50-years costs (Mio USD)	
		Capital cost variation	O&M cost variation
Baseline	1	71	71
	5	5800	5800
	6	320	320
cost -10%	1	64	71
	5	5510	5510
	6	320	284
cost -5%	1	68	71
	5	5655	5655
	6	320	299
cost +5%	1	75	71
	5	5945	5945
	6	321	330
cost +10%	1	78	71
	5	6090	6090
	6	322	345

Since the 50 year benefit exceed the costs only for interventions 1 and 6, the further analysis of uncertainty focused on these two promising interventions.

In the baseline scenario, benefits exceed costs for intervention 1 by 248 Million UDS and for intervention 6 by 192 Million USD. When all parameters under consideration (minimum wage, capital cost and O&M costs) are applied simultaneously with a variation of ±10% in the sensitivity analysis, the ranges (minimum to maximum) for these savings become 238-258 Million USD for intervention 1, and 149-235 Million USD for intervention 6. That means the uncertainty resulting from a combined effect of parameters is rather high, 86 Million USD for intervention 6, while lower, only 20 Million USD, for intervention 1.

7.4. Cost-benefit ratio

Costs and benefits estimations of the selected interventions for the 50-year period (2010-2060) are summarised in Table 89. The outcome of QMRA and saved nitrate removal treatment costs served to evaluate the benefits from the binomial cost-benefit analysis. CBA ratio of a given intervention provides the criteria (ratio >1 is feasible) to select the most adequate intervention for the MAM case study.

Table 89 Costs vs. benefits for interventions 1, 5 and 6, over 50 year (2010-2060)

Intervention	**50-years Costs (Mio USD)**	**50-years benefits (cost-savings) (Mio USD)**			**B/C ratio**
		FC-related	**NO_3-related**	**Total**	
1	71	31	284	315	4.4
5	5800	31	392	423	0.1
6	320	115	392	507	1.6

Summarising, the intervention 1 (improvement of septic tanks in households) and intervention 6 (built centralised wastewater treatment plants for livestock, sewage collection by trucks) result in a benefit/cost ratio > 1, and both provide significant saving in the order of 200 Million USD. Intervention 5 (built new wastewater treatment plants and connects all households to new or existing plants) results in a benefit/cost ration much lower than 1. This is related to the high costs for the installation, connection and operation of new wastewater treatment plants.

Chapter 8. Discussion of model framework and application beyond the study area

The framework developed in this research aims to identify main public health risks and subsequently evaluate potential engineering interventions of significant cost-benefit outcome. The methodology to create this framework was set out through combining different methodologies as described in Figure 67 which have been used and documented in the literature separately and are incorporated in this research and applied for integrated and sustainable water management on a case-specific approach.

This methodology is then a novel approach to help decision-makers to better overview the current and future threats, simulate potential interventions and foresee their effects in the short and long-term application, with the ultimate goal to prioritize those interventions that would result in a favourable cost-benefit action.

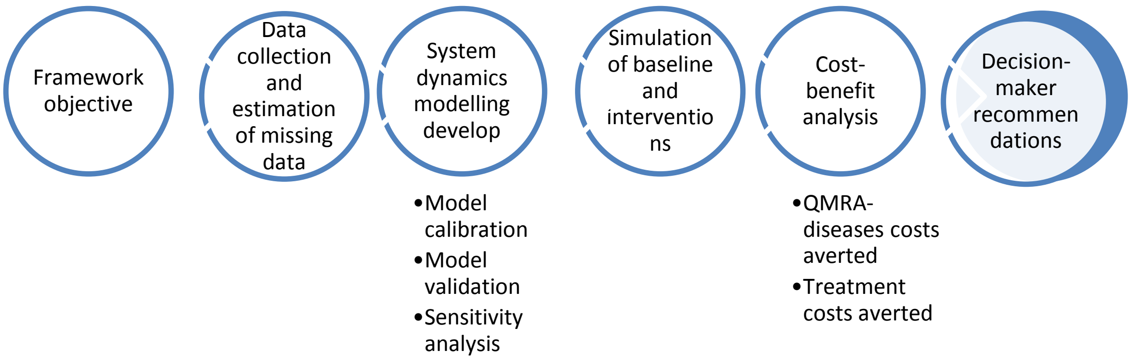


Figure 67 Schematic overview of model framework

A brief description of each framework step is discussed below:

- Framework objective. The objectives are case-specific oriented based on what is known of the current situation, what has been already tried to solve the issue and what are the decision-makers main awareness and potential contributions to solve the issues under study.
- Data collection and estimation of missing data. This step is fundamental for the full framework. Data quality and quantity would be the limiting factor to create model structure and foresee potential interventions. Thus, it is fundamental to invest enough time and resources to collect data of sufficient quality and adequate quantity.

- System dynamics modelling development. The SD modelling process requires a wide range of data available both from site-specific case study and from the general hydrodynamics conditions governing the aquifer system under study.
- Simulation of baseline and interventions. The baseline scenario requires to be tested through calibration and validation of the model, and eventually adaptations are required in order to reasonable match with field data. Simulations of interventions are the key outcome of this framework, in order to foresee future scenarios and apply cost-benefit analysis.
- Decision-makers recommendations. The final step of the framework is writing down the findings of the framework. After careful evaluation and analysis of the costs and benefits that results for each of the interventions tested, a recommendation or of a single or combined interventions is presented to the decision-makers.

8.1. Advantages and disadvantages of the proposed framework

This framework has advantages and disadvantages which are discussed in this section in order to identify the potential for application to other study areas. These are:

Advantage:

- Easy to use model, no advanced skills required by user, although development of the model requires a significant amount of data and specialist knowledge
- Highly versatile for application to other cases studies by modifying input data and/or some minor changes of the internal model structure if required, including other engineering interventions. An example is given below
- The model interface is illustrative for presentation to decision-makers
- Easy to interpret model outcomes, such as graphs and tables
- The general framework is fully used as a supply chain where outputs of one stage serve as inputs to the next stage (i.e. model results are used for cost-benefit analysis)
- Auto verification of model consistency in units and equations, combined with relatively fast simulation time
- Other water/wastewater quality parameters can be tested subject to parameter-specific modifications if required

Disadvantage:

- Limited to the availability of input data in the present case study. Approximations and extrapolations had to be used to derive missing data
- Uncertainty in the hydrogeological characteristics of the karst aquifer (although this is considered a general issue for karstic aquifers modelling).

- The model calculate average contaminant levels at regional scale for relatively large aquifer volume, thus it does not reflect local variations of contaminant concentrations within the region, such as expected in the proximity of point sources.
- Availability of new input data, such as additional field data for contaminant levels, is valuable but makes necessary an adaptation, revalidation and re-calibration of the model. This includes also an update of interventions costs, in order to maintain adequacy of final recommendations to decision makers.
- Simplified karstic heterogeneity model approach. Classic hydrogeological approaches are unable to model conduits of karst aquifers (Bakalowicz, 2005).

The SIWMM can be applied to other case studies without significant modifications if the following basic information is available, eventually after calibration of the model:

- Population development
- Catchment area and aquifer volume
- Recharge
- Flows of water abstraction and wastewater return for the specific socioeconomic activities
- Contaminant concentration in wastewater for the specific socioeconomic activities
- Efficiency of aquifer infiltration by the contaminant
- Information of lifetime or decay rate for microbial contaminants
- Groundwater flow direction and volume

8.2. Applicability of the proposed framework to comparable case studies

An example where the SIWMM can be readily applied for the simulation of contaminant concentration in a similar karstic aquifer is the Franconian Alb in Germany (Einsiedl et al., 2010). The hydrogeology and contaminant transport at this study site have been extensively explored by isotope and tracer tests, combined with mathematical modelling. The conceptual model describes a biphasic system of conduits and rock matrix, with fast water flow through the conduits and slow flow and transport in the matrix. The contaminant under study is the pharmaceutical diclofenac, considered (such as nitrate in the MAM study), as a conservative pollutant without significant biodegradation in the aquifer. Catchment area is clearly defined and associated with spring outflow. Diclofenac is released by wastewater treatment plants in know concentrations. These concentrations in the groundwater are consistent with significant dilution by storage in the large volume of the rock matrix. Importantly, this assumption reflects the similar approach used for modelling of nitrate levels by the SIWMM, based on homogeneous distribution of pollutant over the total aquifer volume of the MAM, even with the likely presence of a conduit-matrix structure. Table 90 shows a summary of data input from the Franconian Alb study case required for modelling through the SIWMM. Remarkably, the SIWMM calculates for an equilibrium state (considering continuous infiltration of the same concentration for 30 years, as suggested in Einsiedl et al., (2010).

Table 90 Simulation of Diclofenac concentration Franconian Alb, case study at two independent sites

Input parameter	Site 1	Site 2
Population	Constant	Constant
Catchment	18 km^2	23 km^2
Aquifer volume	$2.3x10^7$ m^3	1.5x107 m^3
Recharge	230 mm/year	230 mm/year
Wastewater (WW)	80 m^3/s	80 m^3/s
Groundwater flow	136 L/s	174 L/s
Diclofenac concentration in WW	~1 µg/L	~1 µg/L
Infiltration efficiency	100%	100%
Contaminant lifetime	infinite	Infinite
Diclofenac concentration in aquifer: field data (average of 2 samples)	12 ng/L	3 ng/L
Diclofenac concentration in aquifer: simulated data for equilibrium state	7 ng/L	5 ng/L

Diclofenac concentrations are in reasonable agreement with field data. Once the baseline concentration of contaminant is simulated, the effect of interventions might be included in the modelling such as implementation of specific technologies in the wastewater treatment plants for removal of the pharmaceutical of concern. Even if in this case study example, the low concentrations of the pharmaceutical in groundwater are not considered a health concern, a quantitative risk assessment might in principle be applied to the baseline and interventions scenarios, and the scenarios might be compared in a cost-benefit analysis following the suggested model framework.

Applicability of this framework to other similar case studies is possible considering the following limitations:

- The objective of the framework is related to the specific issues of the case study
- The SIWMM is a generic model as discussed in the methodology chapter. Thus the SIWMM itself is subject to data availability for the case study under analysis, including the adjustment required to adequately reproduce the hydrodynamic conditions governing the aquifer under study.
- Costs and benefits of the interventions under evaluation are also subject to data availability at regional level

8.3. Applicability of the proposed framework to non-comparable case studies

For non-similar cases, it is important to highlight those considerations of the SIWM model proposed here for its transferability as described below.

- Modify the multidisciplinary modelling approach. As reviewed by Scheibe et al., (2015), hydrologic models have a rigorous interest for the multi-scale nature of the aquifer under study, resulting in a necessary simplification of models parameters (as the proposed in the SIWMM), which consequently may question models predictability. In this respect, experts on hydrogeological conditions of the study area (catchment), general context of social and economic factors, and complexity of the water supply system (policy makers), might be consulted prior to use the model framework for implementation of adaptive measurements
- Water users and polluters sectors review. Depending of the given urban/rural area, these sectors should be reviewed, in order to redefine and/or modify as appropriate those social and economic activities included or excluded of the model structure, after identification of the significant biological and chemical pollutants
- Sectionalisation of the model framework. The series of steps in the methodology of the model framework could be used independent. For example, for specific case study, data for catchment hydrogeology are not available; pollutants concentration at steady state conditions could be estimated.
- Evaluate model transferability through calibration, validation and sensitivity analysis. If the case study is significantly different to the present study, rigorous analysis of data input and output could be assessed through the model interface in order to identify those specific parameters that could be used to simulate and report for further evaluation

Summarizing, for comparable cases studies, the framework could be easily adapted by considering the above limitations. For non-comparable cases studies, it is important to highlight that through system dynamics modelling as a step of the framework, allows to test and evaluate whether or not there are some logical and numerical relationships between parameters of the model, which facilitates the identification of possible variables interconnections to be used for modelling purposes. Furthermore, calibration, validation and sensitivity analysis would adequately evaluate whether or not the model is able to reproduce the real scenario of the given case study.

Chapter 9. Conclusions and recommendations

A sustainable integrated water management model was developed and applied to the case study of the Metropolitan Area of Merida (MAM) Yucatan, Mexico in order to predict the effect of water management interventions on groundwater contaminant levels and ultimately identify interventions of substantial economic and health benefits.

The MAM covers an area of about 5000 km^2 in the northwestern part of the Yucatan Peninsula, with a population of about 1 Million in 2010. The aquifer underneath the MAM has been recently declared by the RAMSAR Convention as a wetland site of international importance due to its hydrogeological nature. The MAM karstic aquifer presents a high hydraulic conductivity but a low hydraulic gradient, substantial recharge by precipitation, with absence of surface runoff and a continuous groundwater flow directed toward the coast. The karstic aquifer is the only source of drinking water in the MAM (Chapter 3).

The porous soil matrix makes the karstic aquifer very vulnerable to contaminants that directly infiltrate the groundwater along with, for instance, untreated wastewater. Wastewater management in the MAM is insufficient; the majorities of households and of livestock farming are not connected to wastewater treatment plants. Pollutant load is expected to increase along with a rapid population growth, and improved wastewater management practices are urgently needed. These issues are exacerbated in the MAM area by drastic climate change phenomena such as hurricanes. In addition, the complex hydrogeology of the karstic aquifer makes the prediction of spatial and temporal patterns of groundwater contaminants a challenging task. This and a rather limited database for groundwater contaminants complicate the planning of effective wastewater management interventions with respect to public health and economic benefits.

The present thesis project attempts to address this issue by developing a novel sustainable integrated water management model that aims to quantify the effect of interventions not only on groundwater contaminant reduction but also on the improvement of public health and on economic benefits. System dynamics modelling (Vensim software) is applied to predict future levels of selected contaminants in the groundwater of the MAM. The model divides the aquifer in 4 interconnected (by groundwater flow) sections, and considers water inflow (rain, groundwater, and wastewater) and outflow (water abstraction, groundwater). A simplification that largely facilitates the estimation of contaminant levels is the assumption of a spontaneous, homogeneous distribution of the contaminants in the groundwater of each section. The model projects the effect of various interventions that reduce the contaminant load of inflowing wastewater on the concentration of the contaminants (Chapter 6).

The focus of this research was on two indicator contaminants: Faecal coliforms as a microbial indicator of water quality and representing the non-conservative pollutants, which are characterized by a relatively fast decay, and nitrate as a chemical indicator of water quality and an example of a conservative pollutant that may persists in the groundwater for decades. Among the considered socioeconomic activities, domestic and livestock were identified as major sources of faecal coliform and nitrate contamination due to wastewater release into the groundwater. The model is readily applicable to other contaminants that may origin from different socioeconomic activities, such as heavy metals from industrial sources.

Faecal coliforms in groundwater trigger a major risk of diarrhea infection. The latter was analysed using Quantitative Microbiological Risk Assessment (QMRA), and disease associated costs were calculated on the basis of Disability Adjusted Life Years (DALY).

Finally, a preliminary cost benefit analysis (CBA) for several interventions was performed for the most important section of the MAM (section C). The costs associated with the intervention include capital investment, operation and maintenance (O&M). The benefits are defined as cost-savings of expenses, either associated with a reduced burden of waterborne diarrhea or with removal of excessive nitrate in the aquifer for drinking water use in order to comply with the Maximum Contaminant Level (MCL) of 45 mg/L nitrate.

9.1. Conclusions

Major outcomes of the modeled scenarios are:

- Modeled nitrate concentrations in groundwater of aquifer sections A, B, and C are in reasonable agreement with averaged field data for deep parts of the aquifer were drinking water is abstracted (45 m depth).
- Nitrate will exceed the Maximum Contaminant Level according to international standards for drinking water in the most populated sections (Section C) of the study area within the next 20 years.
- Wastewater management interventions have a delayed effect on nitrate levels; it takes years to reduce the nitrate level by 5 mg/L.
- Modeled faecal coliform concentrations required a calibration of the aquifer volume that is effectively accessible to these microorganisms. A reasonable match with field data was achieved for aquifer sections A, B, and C.
- Wastewater management interventions have an “instant” effect on faecal coliforms levels in the groundwater.

- Among various interventions, the treatment of wastewater from livestock is overall the most effective with a long-term reduction of nitrate levels by 30% and faecal coliforms by 80% in the most populated section of the study area.
- Improving existing domestic septic tanks by adding a soil-absorption system is a relatively cheap intervention with nitrate and FC contaminant reductions in 2060 mostly between 10 and 20%.
- The interventions, in particular those for livestock, would significantly reduce diarrhea burden and may even save individual life. The savings estimated by QMRA and DALY for reduced disease burden over a 50-years period are, however, much lower than the costs of the interventions.
- The intervention for livestock would keep nitrate levels in the most populated section of the MAM below the Maximum Contaminant Level for the next 45 years. Given that modern (expensive) technologies would be applied to remove excessive nitrate from drinking water in the absence of interventions, a costs-benefit analysis predicts benefits in the range of 390 Million USD over a 50-year period.
- Even the improvement of existing domestic septic tanks would bring a benefit of about 280 Million USD associated to the reduction of nitrate levels.
- Based on the cost-benefit ratio, intervention 1 (improvement of ST) is the most effective intervention.

9.2. Recommendations

Future work may apply this water management tool to other emergent pollutants and other case studies of karstic aquifers.

Further work is needed for creating field data to assess the health impact of elevated nitrate concentrations, which in this study could not be derived due to lack of statistic data for the study area.

The sustainable integrated water management model identifies the treatment of wastewater from households (by implementing soil absorption systems into septic tanks) and livestock (by centralized wastewater treatment plants) as an interventions with substantial positive impacts on groundwater quality and public health and, in addition, substantial economic benefits. Installing soil absorption systems and treatment plants for livestock, as exemplified by a projected centralized pilot plant including a wastewater transportation infrastructure in the MAM, are priority recommendations to the water authorities of the Metropolitan Area of Merida.

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Appendix A

Drinking Water Maximum Contaminant Level (MCL) Standards. Those highlighted are officially regulated in Mexico.

Pollutant	Type	Unit	USEPA	EU	WHO	Mexico
Microorganisms						
Cryptosporidium	1	% removal	99			
Giardia lamblia	1	% removal	99.9			
Heterotrophic plate count (HPC)	1	CFU/million	≤500			
Total Coliforms	**1**	**CFU/100ml**	**≤5**	**0**	**0**	**0**
Faecal coliform (*E. Coli*)	**1**	**CFU/100ml**				**0**
Viruses (enteric)	1	% removal	99.9			
Turbidity	**1**	**NTU**	**≤5**			**5**
Disinfection Byproducts						
Haloacetic acids (HAA5)	1	µg/l	60			
Chlorate		µg/l			700	
Chlorite	1	µg/l	1000		700	
Bromate	1	µg/l	10	10	10	
Total Trihalomethanes (TTHMS)		**µg/l**	**80**	**10**	**1**	**200**
Disinfectants						
Chlorine dioxide as CLO_2	1	µg/l	800			
Chlorine as CL_2	**1**	**µg/l**	**4000**		**5000**	**200-1500**
Chloramines as CL_2	1	µg/l	4000			
Inorganic chemicals						
Ammonium		**µg/l**		**500**		**500**
Antimony	1	µg/l	6	5	20	
Arsenic	**1**	**µg/l**	**10**	**10**	**10**	**50**
Asbestos (fiber >10 micrometers)	1	Million/l	7			
Barium	**1**	**µg/l**	**2000**		**700**	**700**
Beryllium	1	µg/l	4			
Boron		µg/l		1	500	
Cadmium	**1**	**µg/l**	**5**	**5**	**3**	**5**
Chromium (total)	1	µg/l	100	50	50	50
Copper	**1**	**µg/l**	**1300**	**2**	**2000**	**2000**
Cyanazine		µg/l		0.6		
Cyanide (as free cyanide)	**1**	**µg/l**	**200**	**50**	**70**	**70**
Cyanogen chloride		µg/l			70	
Fluoride	**1**	**µg/l**	**4000**	**1.5**	**1.5**	**1500**
Formaldehyde		µg/l			900	
Lead	**1**	**µg/l**	**15**	**10**	**10**	**25**
Mercury	**1**	**µg/l**	**2**	**1**	**1**	**1**
Selenium	1	µg/l	50	10	10	
Nitrate (as Nitrogen)	**1**	**µg/l**	**10,000**	**50**[1]	**50**[1]	**10,000**
Nitrilotriacetic acid		µg/l			200	
Nitrite	**1**	**mg/l**	**10 (as Nitrogen)**	**50**	**50**	**50**
Thallium	1	µg/l	2			
Organic chemicals						
Acrylamide (of treatment dose)	1	%	≤0.05	0.1	0.5	
Alachlor	1	µg/l	2	20		
Aldicarb*	1	µg/l			10	
Aldrin and dieldrin*	**1**	**µg/l**			**0.03**	**30**
Atrazine*	1	µg/l	3	2		
Benzene	1	µg/l	5	1	10	
Benzo (a)pyrene (PAHs)	1	µg/l	0.2	0.01	0.7	
Bromodichloromethane		µg/l		60		
Bromoform		µg/l		100		

Carbofuran*	1	µg/l	40		7	
Carbon tetrachloride	1	µg/l	5		4	
Chlordane*	**1**	**µg/l**	**2**		**0.2**	**300**
Chlorobenzene	1	µg/l	100			
Chloroform		µg/l			200	
Chlorotoluron*		µg/l		30		
Chlorpyrifos*		µg/l			30	
2,4-D (Dichlorophenoxy acetic acid)	**1**	**µg/l**	**70**	**30**		**50,000**
2,4-DB (Dichlorophenoxy butyric acid)				90		
Dalapon	1	µg/l	200			
1,2-Dibrome-3-chloropropane (DBCP)*	1	µg/l	0.2		1	
1,2-Dichlorobenze		µg/l			1000	
1,4-Dichlorobenzene		µg/l			300	
o-Dichlorobenzene	1	µg/l	600			
p-Dichlorobenzene	1	µg/l	75			
1, 2-Dichloroethane	1	µg/l	5	3	30	
1,1-Dichloroethene		µg/l			30	
1,2-Dichloroethene		µg/l			50	
1, 1-Dichloroethylene	1	µg/l	7			
cis-1, 2-Dichloroethylene	1	µg/l	70			
trans-1, 2-Dichloroethylene	1	µg/l	100			
DDT (Dichlorodiphenyltrichloroethane)		**µg/l**		**1**		**1000**
Dichloromethane	1	µg/l	5		20	
Dimethoate*		µg/l			6	
Dichlorprop*		µg/l		100		
1,2-Dichloropropane	1	µg/l	5		40	
1,3-Dichloropropene*		µg/l			20	
Di (2-ethylhexyl) adipate	1	µg/l	400			
Di(2-ethylhexyl) phthalate	1	µg/l	6	8		
Dibromoacetonitrile		µg/l		70		
Dibromochloromethane		µg/l		100		
Dinoseb	1	µg/l	7			
Dioxin (2,3,7, 8-TCDD)	1	µg/l	0.00003			
Diquat	1	µg/l	20			
Edetic acid (EDTA)		µg/l			600	
Endothall	1	µg/l	100			
Endrin*	1	µg/l	2		0.6	
Epichlorohydrin	1	%	≤0.01 of dose to treat water	0.1	0.4	
Ethylbenzene	1	µg/l	700		300	
Ethylene dibromide	1	µg/l	0.05			
Fenoprop*		µg/l		9		
Gryphosate	1	µg/l	700			
Heptachlor	1	µg/l	0.4			
Heptachlor epoxide	**1**	**µg/l**	**0.2**			**300**
Hexachlorobenzene	**1**	**µg/l**	**1**			**10**
Hexachlorobutadiene		µg/l		0.6		
Hexachlorocyclopentadiene	1	µg/l	50			
Isoproturon*		µg/l		9		
Lindane*	**1**	**µg/l**	**0.2**		**2**	**2000**
MCPA* (2-methyl-4-chlorophenoxyacetic acid)		µg/l			2	
Mexoprop*		µg/l		10		
Methoxychlor*	1	µg/l	40		20	
Metolachlor*		µg/l		10		
Metoxichlorine		**µg/l**				**20,000**

Microcystin-LR		µg/l		1		
Molinate*		µg/l		6		
Molybdenum		µg/l			70	
Monochloramine		µg/l			3000	
Monodhloroacetate		µg/l		20		
Oxamyl (Vydate)	1	µg/l	200			
Polychlorinated biphenyls (PCBs)	1	µg/l	0.5			
Pendimethalin*		µg/l		20		
Phenol		**µg/l**				**1**
Pentachlorophenol	1	µg/l	1		9	
Picloram	1	µg/l	500			
Polycyclic aromatic hydrocarbons		µg/l		0.1[3]		
Pyriproxyfen*		µg/l		300		
Simazine*	1	µg/l	4		2	
Styrene	1	µg/l	100		20	
Terbuthylazine*		µg/l		7		
Tetrachloroethylene	1	µg/l	5	10	40	
Toluene	1	µg/l	1000		700	
Toxaphene	1	µg/l	3			
2,4,5-TP (Silvex)*	1	µg/l	50	9		
2,4,6-Trichlorophenol		µg/l			200	
1,2,4-Trichlorobenzene	1	µg/l	70			
1,1,1-Trichloroethane	1	µg/l	200			
1,2,2-Trichloroethane	1	µg/l	5			
Trichloroacetate		µg/l			200	
Trichloroethylene	1	µg/l	5	10	70	
Trifluralin*		µg/l		20		
Vinyl chloride	1	µg/l	2	0.5	0.3	
Xylenes (total)	1	µg/l	10,000		500	
Radionuclides						
Alpha particles	**1**	**pCi/l**	**15**			**0.1[4]**
Beta particles and photon	**1**	**mRem/y**	**4**			**1[4]**
Radium 226 and Radium 228	1	pCi/l	5			
Tritium		Bq/l		100		
Uranium	1	µg/l	30		15	
Aluminum	**2**	**µg/l**	**50-200**	**200**		**200**
Chloride	**2**	**µg/l**	**250,000**	**250,000**		**250,000**
Color	**2**	**units**	**10**			**20**
Hardness (CaCO3)		**µg/l**				**500,000**
Zinc	**2**	**µg/l**	**5000**			**5000**
Corrosivity	2		noncorrosive			
Foaming Agents	**2**	**µg/l**	**500**			**500**
Iron	**2**	**µg/l**	**300**	**200**		**300**
Manganese	**2**	**µg/l**	**50**	**50**	**400**	**150**
Nickel		µg/l		20	20	
Odor	**2**	**Unit**	**3**			**good**
pH	**2**	**Unit**	**6.5-8.5**			**6.5-8.5**
Silver	2	µg/l	100			
Sodium		**µg/l**		**200,000**		**200,000**
Sulfate	2	µg/l	250,000	250,000		
Total Dissolved Solids	**2**	**µg/l**	**500,000**			**1000,000**
Total Organic Carbon				No change		

Appendix B

Karstic aquifers worldwide.

	Location	
Africa	Anjajavy Forest, western Madagascar	Madagascar dry deciduous forest, western Madagascar
	Ankarana Reserve, Madagascar	Tsingy de Bemaraha Strict Nature Reserve, Madagascar
Asia	Area around Guilin and Yangzhou in Guangxi Zhuang Autonomous Region, China	Jiuzhaigou and Huanglong National Park (UNESCO World Heritage Site), Sichuan, China
	Zhangjiajie National Forest Park, forming part of the Wilingyuan scenic area, Zhangjiajie Prefecture, Hunan, China	Zhi Jin Dong in Gui Zhou Province, China
	The Stone Forest (called the South China Karst by UNESCO), Yunnan Province, China	Arabika Massif (including Voronya Cave- the world’s deepest cave), Abkhazia, Georgia
	Bantimurung, Indonesia	Ofra region, Israel
	Akiyoshi plateau, Japan	Vang Vieng, Laos
	Gunung Mulu National Park, Malaysia	Kilim Karst Geoforest Park, Langkawi, Malaysia
	Kinta Valley, Perak, Malaysia	El Nido, Palawan, Philippines
	Coron, Palawan, Philippines	Sagada, Mountain Province, Philippines
	Chocolate Hills, Bohol, Philippines	Negros and Gigante Islands, Negros Oriental, Philippines
	Krabi región, Thailand	Phangnga Bay Area, southern Thailand
	Kenting National Park, Taiwan	Taseli plateau, Turkey
	Halong Bay, Vietnam	Phong Nha-Ke Bang, Vietnam
	Tam Coc-Bich Dong in Ninh Binh Province, Vietnam	
Europe	Eastern region of the Northern Limestone Alps in the provinces of Salzburg, Upper Austria, Styria and Lower Austria, forming huge limestone plateaus such as Steinernes Meer, Hagengebirge, Tennengebirge, and Hochschwab, Austria	Area around Graz, Styria, Austria
	Central Rhodope karst (including Trigad Gorge), Bulgaria	Devnya Valley, Varna Province, Bulgaria
	Dragoman marsh, Bulgaria	Cadi mountain range, Catalonia
	Regions of Dalmatia (including Zagora), Lika, Gorski kotar, Kvarner and the islands in Croatia	Garraf Natural Park area, Catalonia
	Moravian Karst, Czech Republic	Bohemian Karst, Czech Republic
	Tuhala karst area, Estonia	Ares de l’Anie, in the southernmost part of Baretous valley, southwest France
	Causes of the southern Massif Central, France	Honnetal at Balve, Germany
	Swabian Alb region in the federal state of Baden-Wuerttember, Germany	Region of the Mecsek Mountains in Hungary
	Bukk, a plateau in northeastern Hungary	The Burren in County Clare, Ireland
	Kras, a plateau in northeastern Italy and southwestern Slovenia	Murge, in Apulia and Basilicata, southern Italy
	Herzegovina region of Montenegro and Bosnia-Herzegovina	Polish Jura Chain (Jura Krakowsko-Czestochowska), Poland
	Holy Cross Mountains (Gory Swietokrzyskie) with the Jaskinia Rai, Poland	Tatra Mountains including the Jaskinia Wielka Sniezna (Great Snowy Cave) – the longest cave in Poland
	Slovak Paradise, Slovak Karst and Muranska planina, Slovakia	Apuseni Mountains, Romania
	Kras, a plateau in southwestern Slovenia and northeastern Italy	Region of Inner Carniola, Slovenia
	Picos de Europe and Basque mountains, northern Spain	Ciudad Encantada in the Cuenca province, Castilla-La Mancha, Spain
	White Peak of the Peak District, around Matlock, Castleton (including Thor’s Cave), England, United Kingdom	El Torcal de Antequera nature preserve, southern Spain

	Yorkshire Dales (including Malham Cove), England, United Kingdom	Southern region of the Brecon Beacons National Park, Wales, United Kingdom
	Eastern foothills of Maya Mountains including parts of the Cockscomb Basin Wildlife Sanctuary, Belize	Great Blue Hole near the center of Lighthouse Reef, Belize
North America	Monkman Provincial Park in the Northern Rockies	Nahanni region in the Northwest Territories
	Wood Buffalo National Park in Alberta and the Northwest Territories	Niagara Escarpment, Ontario
	Marbel Canyon, British Columbia	Northern Vancouver Island, British Columbia
	Mogotes in Vinales Valley	Los Haitises National Park
	Cockpit Country region	Cenotes of the Yucatan Peninsula
	Sótanos of the Sierra Gorda, Querétaro	Cacahuamilpa grottos Guerrero
	Karst forest, Puerto Rico	Mountains of northwestern Puerto Rico
	Kosciusko Island, southeaster Alaska	Mitchell Plain and uplands of southern Indiana
	Great Valley of Appalachia (Huntsville, Alabama to northeast Pennsylvania)	Shenandoah Valley, Virginia
	Driftless Area of southwest Wisconsin, southeast Minnesota, northeast Iowa and northwest Illinois.	Florida península
	Mammoth Cave area and the Bluegrass region of Kentucky	Illinois Caverns State Natural Area and Illinois Sinkhole Plain in Monroe County,
	Ozark Plateau of Missouri and Arkansas	Kamas Ranch and Alabaster Cavern area of Oklahoma
	Cumberland Plateau in Middle Tennessee	Grassy Cove Karst Area, Tennessee
	Hill Country of Texas and its northern extensions, including the Palo Pinto Mountains	Carlsbad Caverns National Park, New Mexico
	Central Pennsylvania	Presque Isle County near and around Rogers City in northern Michigan
	Germany Valley Karst Area, West Virginia	Swago Karst Area, West Virginia
	Leeuwin-Naturaliste National Park, near Margaret River, south west western Australia	Cutta Caves National Park and Kintore Caves Conservation Park, Katherine, Northern Territory
Oceania	Northern Swan Coastal Plain, Perth, Western Australia	Naracoorte Caves National Park, South Australia
	Jenolan Caves, New South Wales	Wombeyan Caves, New South Wales
	Mole Creek Karst Conservation Area, Tasmania	Takaka Hill, South Island
	Waitomo, Oparara regions	Nakanai Mountains, East New Britain