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# Gestão de Água Subterrânea através do Planejamento em Escala de Paisagem: incorporação das mudanças de uso do solo e provimento de informações

# Groundwater Management through Landscape Scale Planning: incorporation of land-use changes and provision of information

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To my mother, who always has been giving me the strength to keep believing, I dedicate this work.

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"If you have God on your side, everything becomes clear" - Aryton Senna

# Abstract

Groundwater has a fundamental role in the health of ecosystems and is the primary supply source of approximately two billion people. The pressuring demand for groundwater led to the depletion of the water levels throughout the world. Moreover, groundwater management has proven to be a challenging endeavour. Most of the traditional approaches have failed to sustain a long-term balance. Hence, integrated approaches that include the social, economic, cultural, and institutional settings of each system have been advocated; besides, information for groundwater management has always been a challenge to analyse outside the hydrogeological sphere. Groundwater and land use are intrinsically connected. Land use can influence the recharge and the water demands of the aquifer, and changes in the aquifer can impact the land use. Furthermore, land-use changes act as a mirror of socio-economic development and ecosystem change. Therefore, groundwater management needs to be linked to land use. Landscape Scale Planning is an approach for land use planning defined as an integrative framework based on evidence and centred on landscapes. This approach offers the opportunity for integration of cross-sectoral planning and management. Part of this lies in the adoption of integrative units defined as landscape units. These units are the basis for data gathering, analysis and interpretation. The objective of this thesis is to analyse the incorporation of the land-use changes into groundwater management by the application of the Landscape Scale Planning framework in a cultural landscape at the coast of Northeast of Brazil. To this end, it was investigated: how the Landscape Scale Planning and its connection with land-use changes have the potential to provide information to groundwater management; to what extent landscape units can be integrated to provide a whole-of-landscape analysis using a numerical groundwater model; and the incorporation of temporal dynamics into the analysis of groundwater management. This research has found an intrinsic connection between Landscape Scale Planning and groundwater management. Hence, the challenge lies in the frame of analysis of groundwater management from the Landscape Scale Planning perspective. Underpinned by these findings, guidelines for groundwater management based on this framework were suggested. Also, landscape units were defined and delimited using the following criteria: topographical/hydrological, hydrogeological, water demand, land use and census sector. The landscape units were applied to analyse the groundwater budget. As results, it was possible to determine the groundwater recharge distribution in the system and among the landscape units, as well as to identify the influence of the land use in the recharge of two adjacent landscape units. The application of landscape units also provided a whole-of-landscape integrated analysis of the spatial dynamics of the system that led to the identification of inter-basin groundwater flow (IGF) between two river basins in the study area. A possible reason for this IGF is the different land uses and consequent characteristic water demand. Finally, a framework for the determination of the importance of the transience to groundwater management using the basin time constant was applied. Three landscape units presented a time constant of less than 30 years, indicating the changes, such as in land use, can have more than 60% of its disturbances adjusted within a management timeframe; a possible consequence would be alteration in the groundwater availability for allocation.

Keywords: Spatial dynamics, temporal dynamics, groundwater recharge, management units

**Gestão de Água Subterrânea através do Planejamento em Escala de Paisagem: incorporação das mudanças de uso do solo e provimento de informações**


## Resumo

As águas subterrâneas possuem um papel fundamental na manutenção dos ecossistemas e são a principal fonte de abastecimento para dois bilhões de pessoas. A crescente demanda por este recurso levou a uma depleção dos níveis freáticos ao redor do mundo. Ademais, a gestão de água subterrâneas é um trabalho desafiador. A maioria das abordagens tradicionais têm falhado na promoção de balanço sustentável a longo prazo. Desta forma, abordagens integradas que incluem os aspectos sociais, econômicos, culturais e a as configurações institucionais de cada sistema têm sido defendidas; além disso, fora da hidrogeologia, a análise das informações para gestão de água subterrânea é um desafio. Água subterrânea e uso do solo estão intrinsicamente conectados. O uso do solo influencia a recarga e a demanda de água no aquífero, e mudanças no sistema aquífero podem impactar o uso do solo. Adicionalmente, mudanças no uso do solo atuam como um espelho do desenvolvimento socioeconômico e das mudanças ecossistêmicas. Desta forma, a gestão de águas subterrâneas precisar estar ligada ao uso do solo. O Planejamento em Escala de Paisagem é uma abordagem para planejamento do uso do solo, definido como um arcabouço integrativo baseado em evidências e centrado nas paisagens. Esta abordagem oferece a oportunidade para integração entre setores de planejamento e gestão. Parte disso reside na adoção de unidades integrativas definidas como unidades de paisagem. Estas unidades são a base para coleta, análise e interpretação dos dados. O objetivo desta tese é analisar a incorporação das mudanças de uso do solo na gestão das águas subterrâneas pela aplicação do arcabouço do Planejamento em Escala de Paisagem em uma paisagem cultural na costa do Nordeste do Brasil. Para este fim, foi investigado: como o Planejamento em Escala de Paisagem e suas as conexões com as mudanças do uso do solo tem o potencial para fornecer informações para a gestão de águas subterrâneas; em qual extensão as unidades de paisagem podem ser integradas para fornecer uma análise integrada da paisagem usando modelos numéricos de água subterrânea; e a incorporação da dinâmica temporal na análise da gestão das águas subterrâneas. Esta pesquisa encontrou uma intrínseca conexão do Planejamento em Escala de Paisagem com a gestão de águas subterrâneas. Desta forma, o desafio reside no enquadramento da análise da gestão pela perspectiva do Planejamento em Escala de Paisagem. Assim, diretrizes para a gestão de água subterrânea baseadas neste arcabouço foram sugeridas. Ademais, unidades de paisagens foram delimitadas usando os seguintes critérios: topográfico/hidrológico, hidrogeológico, demanda de água, uso do solo e setores censitários. As unidades de paisagem foram aplicadas para analisar o balanço hídrico subterrâneo e, como resultado, foi possível determinar a distribuição da recarga no sistema e em cada unidade. A aplicação das unidades de paisagem também forneceu uma análise integrada da dinâmica espacial do sistema levando à identificação de um fluxo subterrâneo (IGF) entre duas bacias na área de estudo. Uma possível razão para este IGF são os diferentes usos do solo e consequente características da demanda hídrica. Finalmente, uma avaliação da importância da dinâmica temporal para gestão de águas subterrâneas baseado na constante temporal da bacia foi aplicada. Três unidades de paisagem apresentaram constante temporal menor que 30 anos, indicando que mudanças no uso do solo podem ter 60% dos seus impactos sentidos dentro do horizonte de planejamento, uma possível consequência disto seria a alteração da disponibilidade de água subterrânea para alocação.



Palavras-chave: Dinâmica especial, dinâmica temporal, recarga de água subterrânea, unidades de gestão

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# Chapter One: Introduction

## 1.1 Research motivation

Renewable natural resources refer to natural supplies that, after exploited, can return to previous levels through natural processes of replenishment. Among the vital natural resources are water, land and biodiversity (Westhoek et al., 2016). Water resources are essential to life, and the development of humankind has happened and continues to happen, near rivers courses (Arsel, 2011; Richey et al., 2015; Zilov, 2013). However, these resources are different from the others, because they are highly localised with occurrence varying in time, and with some degree of uncertainty (Qi and Altinakar, 2011; Westhoek et al., 2016). These aspects of water resources are even more important in concerns to groundwater resources.

As groundwater generally means the infiltrated water that reaches the saturated zone, its occurrence is dependent on geological characteristics that favour the storage and the movement of such resource (Margat and Gun, 2013). The renewability of groundwater resources is a small amount from the total storage and highly dependent on the recharge due to rainfall (Jasechko and Taylor, 2015). In some regions, the groundwater might not be rechargeable due to geological constraints (Shrestha et al., 2018), while in others, more pronouncedly the confined aquifers, the long exploitation without sufficient recharge can result in the dewatering and loss of the compaction, reducing the total recharge capacity (García et al., 2017). These cases are considered as "groundwater mining" (Custodio et al., 2017), and the groundwater resources are actually non-renewable (Jakeman et al., 2016).

Groundwater resources have a fundamental role in the maintenance of baseflow to rivers, as a consequence, in the health of ecosystems (Bertrand et al., 2012), but it is also the largest stock of accessible water, is the primary supply source of approximately two billion people, and supplies at least half of the world's demand for irrigation purposes destined to food production (Famiglietti, 2014; Richey et al., 2015; Westhoek et al., 2016). Furthermore, groundwater is a key reserve for drought resilience and water security in places like United States, Australia, Northeast of Brazil and Ethiopia (MacDonald et al., 2019; Taylor et al., 2013), while in others, such as France, Germany and United Kingdom, is responsible for more the 60% of the public water supply (Margat and Gun, 2013).

This pressuring demand for groundwater, added by the reliability of the groundwater resources along the year and the advances in pumping technologies have led to a considerable rise in the amount of groundwater abstracted, especially in India and North Africa (Margat and Gun, 2013; Massuel et al., 2017). On top of that, on the one hand, the increasing population growth, economic development and the natural human response to withdraw more water during droughts have increased even more the demand for groundwater in the past 30 years (Famiglietti, 2014; Wada et al., 2012). On the other hand, the climate changes due to anthropogenic pressures altered the rainfall patterns reducing groundwater recharge (Gorelick and Zheng, 2015; Konikow and Kendy, 2005). As a consequence, depletion of the water levels in the aquifers throughout the world increased by 22% from 2000 to 2010 (Dalin et al., 2017; Famiglietti, 2014). This depletion brings more unwanted situations such as the increasing cost of pumping (Foster et al., 2017), reduction of baseflow (White et al., 2016), and, in coastal regions, the salinisation of water (Stein et al., 2019). Such negative consequences of long-term groundwater abstraction higher than the aquifer renewable capacity can be defined as groundwater overexploitation (Custodio, 2002).

With the overexploitation of aquifers occurring throughout the world, the need for careful management of the groundwater resources became extremely important (Gleeson et al., 2012). Groundwater resources management deals with the balance of groundwater exploitation as well as the demand by water and land users with the aquifer's capacity of recharge, seeking to mitigate the negative impacts of its development (Foster and Chilton, 2017; White et al., 2016). Besides, groundwater management has to address not only the quantitative aspect but also the qualitative aspect (Jakeman et al., 2016), while ensuring long-term sustainability (Alley and Leake, 2004). The implementation of groundwater management has started, and continues until today, with the application of simple measures, such as the volumetric allocation of the abstractions based on the concept of safe yield or sustainable yield, trigger-levels was a warning measure, and buffer zones to the protection of groundwater from contaminants or nearby ecosystem from depletion (Alley and Leake, 2004; Konikow and Leake, 2014; Noorduijn et al., 2018; Zhou, 2009). With time, the groundwater management became more complex applying measures in combination (as a strategy) (FAO, 2016), with the institutionalisation of groundwater management plans (that proposes measures to be implemented in certain conditions) (White et al., 2016) or through the application of technical approaches to increase the aquifer availability (such as managed aquifer recharge) (Dillon et al., 2018).

However, groundwater management has proven to be a challenging endeavour. Most of these technical approaches have failed to sustain a long-term balance. Groundwater management objectives (Aeschbach-Hertig and Gleeson, 2012), as well as the groundwater management measures, have also failed in avoiding overexploitation (Nabavi, 2018). Groundwater management plans have low or unclear effectiveness (White et al., 2019), especially if applied in a command and control approach that ignores the links between social and environmental systems (Bouchet et al., 2019). Foster and Chilton (2017) have advocated that managing groundwater does not regard only to the hydrogeological understanding of the system under stress, but it requires, in the same level, the understanding of the water and land users as wells. Therefore, groundwater management needs to comprise the social, economic, cultural, and institutional settings of each system (Aeschbach-Hertig and Gleeson, 2012). In this direction, the overarching principle of Integrated Water Resources Management has been developed, followed by a range of frameworks such as the Water Framework Directive (European Commission, 2000) and Global Groundwater Governance Framework for Action (FAO, 2016). However, some challenges remain, as the changing needs and demands from an ever-changing society (Ross, 2016).

The society has changed considerably in the past 100 years, and most of this change came with economic development and further advances in technologies. As a consequence, humankind has altered natural areas and ecosystems all over the world (Foley et al., 2011). The way that society uses and alters the biophysical cover can be defined as land use (Fritz et al., 2017). The present earth's ice-free surface has been heavily transformed: almost 40% of natural land use was converted into agricultural land uses (Meiyappan and Jain, 2012). In the first 50 years of the century, the changes were concentrated in North America, Europe and Asia; after that, the changes have been happening more in the tropical areas of America and Asia (Foster, 2018). Nowadays, the land-use changes are even happening due to drivers located from a distance of the area under change, in the process of globalisation of land use (Meyfroidt et al., 2013). These land-use changes lead to changes in the ecosystems (Arnell et al., 2019), the hydrological processes (Li et al., 2018), especially groundwater recharge (Han et al., 2017), and also changes in demand for water (Bryan et al., 2018).

Groundwater and land use are intrinsically connected. Land use can influence the recharge and the water demands of the aquifer (Han et al., 2017), and changes in the aquifer can impact the land use such as the severe reduction of water level leading to land subsidence.

In a spatial perspective, land-use change can transform the physical landscape by altering the flow to rivers due to peak discharges of reduction of baseflow (Remondi et al., 2016), while in the temporal perspective, groundwater quality can take decades to recover from contamination of agricultural land uses (Aquilina et al., 2012). Therefore, groundwater management needs to be linked to land use (Foster, 2018). The connection of land use with groundwater management can bridge the gap between the changes in society and its consequent pressures in groundwater resources, given that land-use changes act as a mirror of socio-economic development and ecosystem change (Li et al., 2018).

Such incorporation of land-use change into groundwater management needs to be developed with the aid of the land use planning, given that future land uses have to take into account the possible impacts to groundwater resources (Scanlon et al., 2007). Furthermore, the analysis and planning for the land use and groundwater must include a 'theory of place' – referring to the understanding of land use and groundwater interactions under a particular context, and a 'theory of change' – referring to the understanding of how the system can be influenced by the users and management arrangements (Wiegant and van Steenbergen, 2017).

According to Foster (2018), among the main barriers for this integration are the legal and institutional constraints. Two different sectors generally operate land use planning and groundwater management and, many times, the policymakers, managers, stakeholders have a poor understanding of the geological concepts and the existing connection between land use and groundwater (Wiegant and van Steenbergen, 2017). To effectively communicate and to support a planning process comprising different sectors under such institutional constraint, the provision of sound information is fundamentally important (Cuadrado-Quesada and Gupta, 2019; Neuendorf et al., 2018).

Lack of information has been considered an obstacle to sustainable management practices in diverse sectors (Nakasone and Torero, 2016). The provision of more information reduces uncertainty and facilitates the decision-making process (Galioto et al., 2020). Especially for groundwater management (van der Gun, 2017), but not only regarding the aquifer. Information regarding the socio-economic aspect is equally essential (Knüppe, 2011). Information comes from data, and the knowledge is built on top of information (Rowley, 2007), but data and information about groundwater are generally restricted due to its hidden aspect (Albrecht et al., 2017). In some regions, the scarcity of data is a severe issue that constrains the application of tools to generate information, such as groundwater modelling, and results in elevated levels of uncertainty (Delottier et al., 2017). Furthermore, the collection of data and further analyses of information from natural and social settings seeking the monitoring and assessment of groundwater management must be conducted in the appropriate scale taking into account its cost-effectiveness (Varady et al., 2016), as well as the integration with the land use planning (Cerutti et al., 2019), given that there is a mismatch between the boundaries of land use planning and the water resources management (Renouf et al., 2018; Serrao-Neumann et al., 2019).

Land-use planning can be a fundamental tool to solve the conflicts from the competing land uses, environmental concerns and sectoral interests (Metternicht, 2018). An emergent approach for land use planning is the Landscape Scale Planning. Landscape Scale Planning is defined as an integrative framework based on evidence and centred on landscapes (Selman, 2006). This framework applies the concept of landscapes as defined by the European Landscape Convention, where a landscape results from the action and integration of natural and human factors (Council of Europe, 2000) and focuses on the cultural landscape – those that have been modified by anthropogenic factors. Hence, the landscape concept is not restricted to its biophysical aspect, but also includes the socio-economic and cultural ones (Selman, 2006).

Furthermore, at the core of Landscape Scale Planning is the opportunity for the integration of cross-sectoral planning and management through policies. Part of this opportunity comes from the adoption of integrative units with an internal similarity of its multitude of aspects, defined as landscape units (Selman, 2006). These landscape units are the basis for data gathering, analysis and interpretation; hence it enables a more holistic synthesis of the area to be managed, including the dynamic forces driving its changes (Selman, 2006). Therefore, the Landscape Scale Planning has the potential to guide a planning and management process based on information that embraces the spatial, temporal and modification dynamics, while seeking for the multifunctionality of landscapes through the connection of their different aspects (Selman, 2009, 2006; Selman and Knight, 2006).

This thesis hypothesises that the Landscape Scale Planning framework can underpin groundwater management by the incorporation of land-use change while aiding the provision of information.

In order to test this hypothesis, a cultural landscape with a sedimentary aquifer located at the coast of the Northeast of Brazil was chosen. This landscape is the central economic region of the State of Paraíba with approximately 1 million inhabitants, covering an area of 1,032 $km^2$.

This landscape has been altered from pristine conditions to mainly agricultural and urban land uses. The aquifer in this landscape has a complex formation presenting one phreatic subsystem that partially overlaps one confined subsystem. Indications of groundwater overexploitation have been found in previous studies (Batista et al., 2011; Braga et al., 2015). Therefore, this cultural landscape represents a typical situation of stress on coastal groundwater resources from urban and agricultural demands. Similar setting can be found in other regions of Brazil (Bertrand et al., 2016), as well as in other regions of the world, such as in Indonesia (Adyasari et al., 2019) and California (USA) (Hagedorn et al., 2018).

## 1.2 Aim and Objectives

The overall aim of this thesis is to analyse the incorporation of the land-use changes into groundwater management by the application of the Landscape Scale Planning framework in a cultural landscape at the coast of Northeast of Brazil.

The specific objectives are:

i. Investigate to what extent the multiple dimensions of Landscape Scale Planning and their connections with land-use changes have the potential to provide information to groundwater management;

ii. Suggest guidelines for the application of Landscape Scale Planning as a basis for groundwater management;

iii. Analyse to what extent spatial landscape units can be integrated to provide a whole-of-landscape analysis to support groundwater management;

iv. Examine to what extent the temporal dynamics of the groundwater system can be incorporated into the analysis of groundwater management.

## 1.3 Thesis outline

This thesis is distributed into seven chapters. The present (first) chapter presents the research motivation, hypothesis, overall and specific objectives guiding this research. Chapter Two brings a systematic review guided by groundwater management and Landscape Scale Planning to investigate the theoretical relationship among these concepts. Chapter Three is dedicated to present the study area, a cultural landscape at the Northeast of Brazil, from the biophysical aspects to the socioeconomics ones and including the institutional settings referring to land use and groundwater management. Chapter Four shows the process of setting up a conceptual and numerical groundwater model tool under the restriction of the lack of data; this tool is applied to obtain information about the study area. Chapter Five depicts the application of quantitative methods to increase the knowledge to base groundwater management in the study area by overcoming a data-scarcity situation. Chapter Six defines, delineates and applies the concept of landscape units to provide information and support groundwater management. Finally, Chapter Seven has the concluding remarks and perspectives for further research.

# Chapter Two[1] - Groundwater management in Coastal Areas through Landscape Scale Planning: A systematic literature review

## 2.1 Introduction

Groundwater is the water supply for a large part of the world's population and has a fundamental role in ecosystem health and conservation (Gorelick and Zheng, 2015). When used unrestricted, overexploitation can occur, especially when the natural recharge is incompatible with increased anthropogenic demands (Molle et al., 2018). Managing groundwater systems present a complex challenge (Konikow and Kendy, 2005). Groundwater systems encompass the aquifer, including the flow boundaries, sink (discharges and withdrawal) and sources (recharges) (Alley et al., 2002).

Groundwater management 'seeks to balance and mitigate the detrimental impacts of development, with plans commonly used to outline management pathways' (White et al., 2016, p. 4863). Such plans usually comprise a set of actions implemented to achieve the sustainability of a groundwater system. It is expected that those actions keep the balance between the exploitation of the resource with the demands of water and land users. Consequently, the management of groundwater involves managing both water and land use, from a socio-economic perspective, and managing the resource behaviour under stress, from a hydrogeological perspective (Foster and Chilton, 2017). To effectively deliver this outcome, information is of paramount importance to inform the management process (van der Gun, 2017). This includes information about the groundwater systems, potential interactions and threats to the system, and the functions and benefits of groundwater for stakeholders (van der Gun, 2017). Hence, adequate management requires proper understanding of the systems and their influence on society, and vice-versa.

Jakeman et al. (2016), taking an Integrated Water Resources Management (IWRM) perspective, suggest that one of the priorities of groundwater management is balancing

[1] This chapter is an adapted reproduction of the following paper: Braga, A.C.R., Serrao-Neumann, S. & Galvão, C. O. Groundwater Management in Coastal Areas through Landscape Scale Planning: A Systematic Literature Review. Environmental Management 65, 321–333 (2020)

groundwater exploitation with demand from natural processes, the economy and society. Intrinsically related to IWRM is the concept of groundwater sustainability, defined by Alley and Leake (2004, p. 13) as the 'development and use of groundwater in a manner that can be maintained for an infinite time without causing unacceptable environmental, economic, or social consequences'. There is an ongoing debate about how thresholds should be determined to guide groundwater exploitation (Molle et al., 2018), whilst sustaining the health of surrounding terrestrial ecosystems (Ross, 2016). While the primary source of recharge to aquifers is precipitation, groundwater management requires consideration of the total water cycle because this is affected by both anthropogenic pressures and biophysical characteristics (Minnig et al., 2018). Hence, the concepts of groundwater recharge, availability, and sustainability are at the core of groundwater management.

Coastal regions are influenced by and influence on the water cycle. Coastal regions comprise many water systems such as rivers, wetlands, floodplains and estuaries, often connected by aquifers (Momtaz and Shameem, 2016). More than half of the human population and activities are concentrated in these regions (Chatton et al., 2016). Consequently, coastal regions have been strongly influenced by social and economic development which affects the sustainability of groundwater systems (Chatton et al., 2016). Generally, in highly urbanised areas, anthropogenic pressures may affect the aquifers in many ways, including, but not limited to, altering the recharge due to impermeable surfaces, point source contamination due to leaks, and overexploitation due to uncontrolled increase of extractions (Minnig et al., 2018; Tam and Nga, 2018). Other groundwater related issues challenging coastal urbanised regions include, *inter alia*, saltwater upconing, saltwater intrusion and reduction of submarine discharge to ecosystems (Michael et al., 2017; Petelet-Giraud et al., 2018). These regions are therefore complex landscapes that require more integrated management approaches.

Changes in the biophysical cover affects the recharge process and available groundwater for human consumption and environmental functions (Tam and Nga, 2018). Additionally, as anthropogenic pressures increase they may compromise the sustainability of these systems (Wiegant and van Steenbergen, 2017). There is a connection between how land use influences and is influenced by groundwater systems. While land use can influence the recharge and the water demands on groundwater systems, changes in the groundwater system can impact the land use such as the rising of water tables leading to floods (Han et al., 2017). However, land use is generally planned and managed following anthropogenic rather than natural criteria.

Thus, to achieve groundwater sustainability, groundwater management and land use planning must be integrated (Foster, 2018).

Landscape Scale Planning is an emerging approach that takes a holistic view of land use planning and management by focusing on how human intervention altered landscapes. Landscape Scale Planning is defined as an integrative framework based on evidence and centred on landscapes (Selman, 2006). Landscapes have both physical and informational flows (e.g. environmental, economic, societal and cultural flows), along with dynamic forces driving their change. Hence, the application of Landscape Scale Planning requires the integration of diverse datasets from different land uses so that changes (or the relationship between biophysical and socio-economic conditions) are better understood by the planning process (Selman, 2006). Landscape Scale Planning offers an opportunity for the integration of cross-sectoral policies using the landscape as a unit of analysis to achieving sustainable resource management (Selman, 2006). This includes making decisions based on information that takes into account competing land use interests but also their multiple functions, values and ecosystem services (Antrop and Van Eetvelde, 2017; Plieninger and Bieling, 2012; Selman, 2009). Landscape Scale Planning embraces three dimensions: spatial, temporal and modification. The spatial dimension centres on the recognition of distinct landscape units defined as integrative units that have more internal resemblance compared with the surrounding regions; ii) the temporal dimension entails the first use of a landscape through to its sustainable use by future generations; and, iii) the modification dimension involves the anthropogenic alterations that affected and will affect the landscape and its features along the spatial and temporal dimensions (Selman, 2006). Hence, Landscape Scale Planning enables integrated planning for differing systems (natural, forest, agricultural, urban), including the incorporation of conflicts and changes that happen within them.

The concept of landscape underpinning this approach follows the definition from the European Landscape Convention as ‘an area, as perceived by people, whose character is the result of the action and integration of natural and/or human factors’ (Council of Europe, 2000). From this definition, landscapes can be divided into three types: i) those purely natural; ii) those resulting from the action of natural and human factors; and, iii) those purely altered by human factors (Selman, 2006). Our paper focuses on the last two types because these landscapes go beyond the traditional biophysical aspect. They include societal and cultural aspects (i.e.

anthropogenic drivers) that have altered their pristine condition and created non-natural land uses. Hence, such alterations and drivers must be understood by any management process.

In addition to land use planning (Hawkins and Selman, 2002), Landscape Scale Planning has been used in environmental management (Andersen et al., 2019), and in conjunction with other approaches such as the ecosystem approach (Morrison et al., 2018). To date, not many applications of Landscape Scale Planning have been found in groundwater management. This paper uses a systematic literature review to investigate to what extent the multiple dimensions involved in Landscape Scale Planning (spatial, temporal and modification), and its connection with land use changes, have the potential to provide the much needed data and evidence to inform groundwater management decisions (Vadiati et al., 2018) under both a total water cycle and IWRM perspectives.

## 2.2 Methods

Literature reviews aim to identify the current state of knowledge on a selected topic or subject and provide insights or directions for future research (Paré et al., 2015). Literature reviews can follow traditional (narrative) or systematic methods. Although narrative literature reviews are still widely applied and informative, systematic literature reviews have been increasingly applied due to its rigour, reproducibility and transparency by following predetermined steps (Mallett et al., 2012). This is an important advantage, especially when a review is applied to inform environmental management and planning based on evidence (Bilotta et al., 2014). On the other hand, the systematic literature review can be too restrictive due to its rigour, not comprising publications gathered though the predetermined steps. Also, given that such studies usually vary among a wide range of methodologies or different locations, the interpretation and summarisation of results can be controversial (Garg et al., 2008).

Systematic literature review (SLR) is a method for locating, appraising and summarising evidence on a given topic. It started in the Health Sciences but its application has extended to other fields due to the wide range of published articles available (Haddaway and Bilotta, 2016). While there is a raft of methods to carry out systematic reviews of literature (Haddaway and Bilotta, 2016), this paper adopts the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) (Moher et al., 2009), which has been applied to many investigations concerning environmental management (Baum and Bartram, 2017; Tseng et al.,

2019) and spatial planning (Boulton et al., 2018; Song et al., 2018). Following the PRISMA method, the following steps were used to carry out the review: search strategy; screening and eligibility criteria; and, content and result analysis (Moher et al., 2009).

The search strategy was based on an iterative process to refine the use of keywords to identify selected publications whilst maintaining a balance between comprehensiveness and relevance to the research question (Bilotta et al., 2014). The screening and eligibility criteria step sought to determine which studies would be included in the review to ensure transparency and avoid biases in the selection process (Bilotta et al., 2014). Finally, the content and result analysis extracted and assessed the valid data and information through a structured process to ensure they were supported by evidence so as to answer the research question guiding this paper (Haddaway and Macura, 2018).

### 2.2.1 Search strategy

Searches using the combination of the terms “Groundwater Management” and “Landscape Scale” did not return any results. Consequently, the two concepts were further unpacked to derive other search terms and proxies. Relative to “Groundwater management” proxy was “Groundwater Recharge”. Relative to “Landscape Scale Planning” proxies were “landscape scale”, “land use planning”, and “land use change”. These keywords were joined in the query searches using Boolean operators (“AND”, “OR"). The protocol for conducting the searches involved queries in the Web of Science (category: “Topic”) and Scopus (category: “Title, abstract, and keywords”) databases, of peer-review journals and book chapters, published in the English language up to December 2018. This resulted in 480 publications (Figure 2.1). Subsequently, a search among the results was conducted using the keyword “coast*” to narrow down the results to those related to coastal areas. This resulted in 55 publications. Information on the delimitation of the keywords can be found in appendix A.

### 2.2.2 Screening and eligibility criteria

The screening of search results followed three steps. Firstly, the duplicate articles or those that did not have a full-text available were removed. Secondly, a screening of the title and abstracts was carried out to remove publications unrelated to the research question guiding the paper – that is, if the multiple dimensions involved in Landscape Scale Planning (spatial, temporal and modification) have the potential to provide data and evidence to inform groundwater management decisions taking into account land-use changes under both a total water cycle and IWRM perspectives. The screening evaluated whether: (i) analysis or provision of information to groundwater management was one of the objectives of the article; and, (ii) publications analysed land use patterns or changes. Finally, articles that passed this first screening were then analysed for content if:

- at least one of the three Landscape Scale Planning dimensions was analysed in the paper (i.e. spatial, temporal and/or modification dimensions), and/or
- type(s) of land use covered by the paper had at least one non-natural use (urban, agricultural, forest).

In total, 28 publications passed the screening phase. These were analysed to identify the countries where the studies were developed and other general aspects such as date of publication. Publications were then classified based on: (i) which dimensions of the Landscape Scale Planning approach they were related to; (ii) what methods were used to investigate such dimensions, given that this represents the procedure used to transform raw data into information for management purposes; (iii) and, how they could contribute to the objectives of this paper. An in-depth content analysis was then carried out to investigate how spatial, temporal and modification dimensions aspects were addressed, how land use changes were considered, and methods used for these (see Table 2.1). Methods used by the publications were considered in the analysis to explore the extent to which they supported the investigation of spatial, temporal and/or modification dimensions. Analysis results were then contrasted with the existing literature on groundwater management, land use changes and Landscape Scale Planning approaches.

Table 2.1 - Criteria used to classify the publications and content analysis.

| **Criteria** | **Description of the classification criteria of the publications** |
|---|---|
| Country/continent | The location where the study was conducted |
| Spatial Dimension | Study site, catchment or hydrogeological unit |
| Temporal Dimension | Past, present or future analyses |
| Modification Dimension | Current land uses or land use changes across time (past and future) |
| Methods applied | The methods applied in each publication to address differing dimensions |

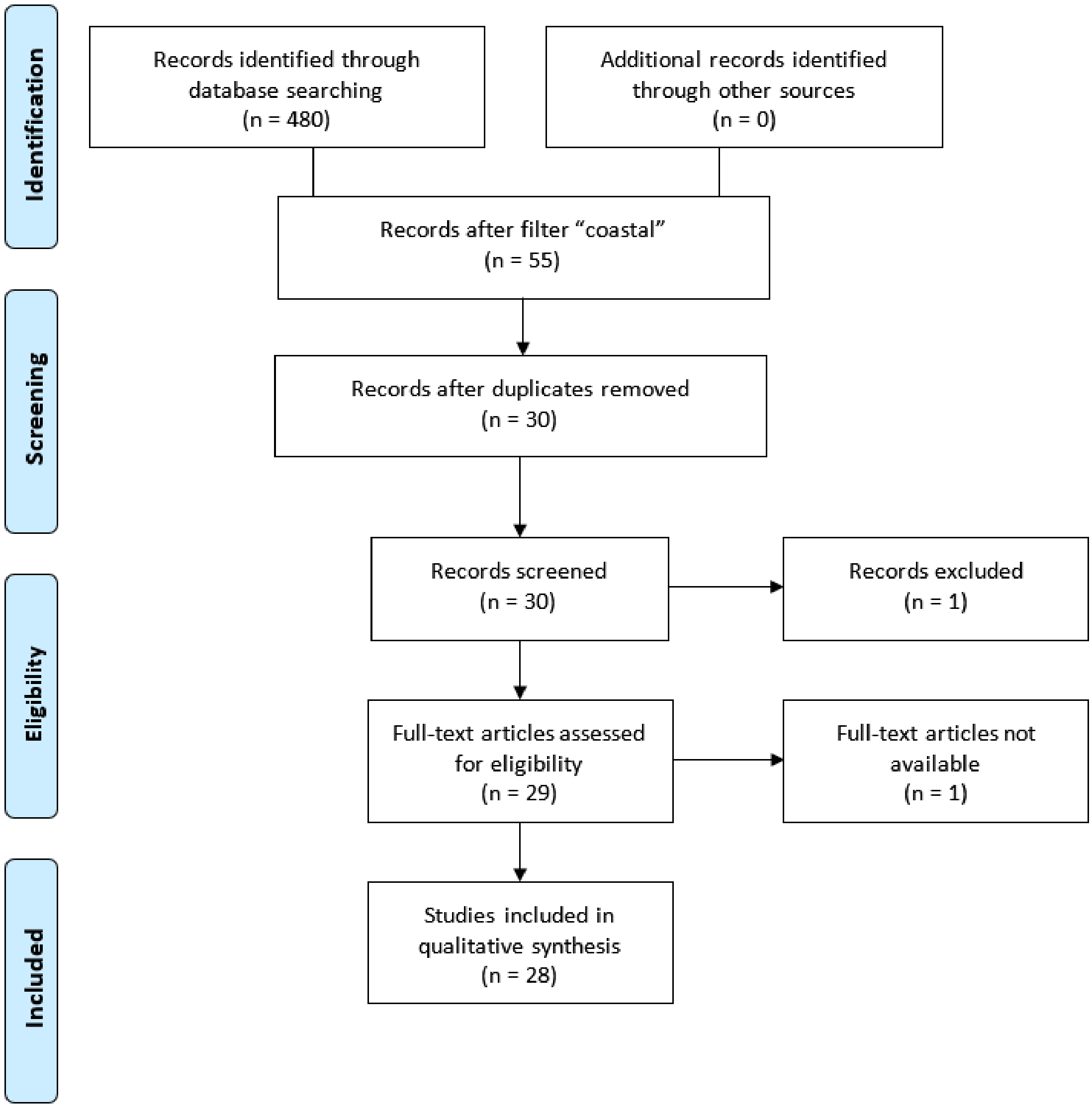


Figure 2.1 - PRISMA Flowchart

## 2.3 Results

### 2.3.1 Overview

Among the 28 selected publications, eight were from the Americas (seven in North and one in South America), seven from Asia, six from Africa, six from Europe and three from Oceania. Only one of these publications had case study areas from multiple countries. The timespan of the publications ranged from 2001 to 2018. The year with the highest number of publications was 2012. However, there was no significant trend in the number of publications per year. The spatial dimension, especially spatial boundaries used to delimit the studies, was identified in all selected papers. The modification dimension regarding current land use and land use changes was present in 23 papers and the temporal dimension in only 16. A complete list of the selected publications can be found in the Table 2.2.

With respect to methods, the majority of papers adopted some sort of modelling, followed by an analysis of groundwater quality or quantity through chemical methods (Table 2.3). Other methods included conceptual approaches, spatial analysis, multicriteria analysis, physical methods, and sensitivity analysis. 21 publications presented potential evidence to inform groundwater management based on at least one dimension of Landscape Scale Planning, while 16 showed perspectives that could lead to a better understanding of groundwater management issues. The content analysis revealed publications predominantly focused on two emerging groundwater management issues: alterations in groundwater recharge and groundwater contamination. Table 2.4 shows a general overview of the connection between the Landscape Scale Planning dimensions and these two issues.

Table 2.2 - Selected publications for systematic review

| Item | Publication Reference |
|---|---|
| 1 | Allouche, N., Maanan, M., Gontara, M., Rollo, N., Jmal, I., Bouri, S., 2017. A global risk approach to assessing groundwater vulnerability. Environmental Modelling & Software 88, 168-182. |
| 2 | Aquilina, L., Vergnaud-Ayraud, V., Labasque, T., Bour, O., Molenat, J., Ruiz, L., de Montety, V., De Ridder, J., Roques, C., Longuevergne, L., 2012. Nitrate dynamics in agricultural catchments deduced from groundwater dating and long-term nitrate monitoring in surface- and groundwaters. Sci Total Environ 435-436, 167-178. |

Table 2.2 - Selected publications for systematic review (continued)

| Item | Publication Reference |
|---|---|
| 3 | Batelaan, O., De Smedt, F., Triest, L., 2003. Regional groundwater discharge: phreatophyte mapping, groundwater modelling and impact analysis of land-use change. Journal of Hydrology 275, 86-108. |
| 4 | Brauman, K.A., Freyberg, D.L., Daily, G.C., 2012. Land cover effects on groundwater recharge in the tropics: ecohydrologic mechanisms. Ecohydrology 5, 435-444. |
| 5 | Calderon, H., Uhlenbrook, S., 2016. Characterizing the climatic water balance dynamics and different runoff components in a poorly gauged tropical forested catchment, Nicaragua. Hydrological Sciences Journal 61, 2465-2480. |
| 6 | Callahan, T.J., Vulava, V.M., Passarello, M.C., Garrett, C.G., 2012. Estimating groundwater recharge in lowland watersheds. Hydrological Processes 26, 2845-2855. |
| 7 | Collin, M.L., Melloul, A.J., 2001. Combined land-use and environmental factors for sustainable groundwater management. Urban Water 3, 229-237. |
| 8 | Daraio, J.A., 2017. Potential Climate Change Impacts on Streamflow and Recharge in Two Watersheds on the New Jersey Coastal Plain. Journal of Hydrologic Engineering 22. |
| 9 | Dawes, W., Ali, R., Varma, S., Emelyanova, I., Hodgson, G., McFarlane, D., 2012. Modelling the effects of climate and land cover change on groundwater recharge in south-west Western Australia. Hydrology and Earth System Sciences 16, 2709-2722. |
| 10 | Gondwe, B.R.N., Merediz-Alonso, G., Bauer-Gottwein, P., 2011. The influence of conceptual model uncertainty on management decisions for a groundwater-dependent ecosystem in karst. Journal of Hydrology 400, 24-40. |
| 11 | Guan, H., Love, A.J., Simmons, C.T., Hutson, J., Ding, Z., 2010. Catchment conceptualisation for examining applicability of chloride mass balance method in an area with historical forest clearance. Hydrology and Earth System Sciences 14, 1233-1245. |
| 12 | Guinn Garrett, C., Vulava, V.M., Callahan, T.J., Jones, M.L., 2012. Groundwater-surface water interactions in a lowland watershed: source contribution to stream flow. Hydrological Processes 26, 3195-3206 |

Table 2.2 - Selected publications for systematic review (continued)

| Item | Publication Reference |
|---|---|
| 13 | He, B., Wang, Y., Takase, K., Mouri, G., Razafindrabe, B.H.N., 2008. Estimating Land Use Impacts on Regional Scale Urban Water Balance and Groundwater Recharge. Water Resources Management 23, 1863-1873. |
| 14 | Hu, X., Ma, C., Qi, H., Guo, X., 2018. Groundwater vulnerability assessment using the GALDIT model and the improved DRASTIC model: a case in Weibei Plain, China. Environ Sci Pollut Res Int 25, 32524-32539. |
| 15 | Koh, E.-H., Lee, S.H., Kaown, D., Moon, H.S., Lee, E., Lee, K.-K., Kang, B.-R., 2017. Impacts of land use change and groundwater management on long-term nitrate-nitrogen and chloride trends in groundwater of Jeju Island, Korea. Environmental Earth Sciences 76. |
| 16 | Kurtzman, D., Scanlon, B.R., 2011. Groundwater Recharge through Vertisols: Irrigated Cropland vs. Natural Land, Israel. Vadose Zone Journal 10 |
| 17 | Mair, A., Hagedorn, B., Tillery, S., El-Kadi, A.I., Westenbroek, S., Ha, K., Koh, G.-W., 2013. Temporal and spatial variability of groundwater recharge on Jeju Island, Korea. Journal of Hydrology 501, 213-226. |
| 18 | Mattas, C., Voudouris, K., Panagopoulos, A., 2014. Integrated Groundwater Resources Management Using the DPSIR Approach in a GIS Environment Context: A Case Study from the Gallikos River Basin, North Greece. Water 6, 1043-1068. |
| 19 | McFarlane, D., Strawbridge, M., Stone, R., Paton, A., 2012. Managing groundwater levels in the face of uncertainty and change: a case study from Gnangara. Water Science and Technology: Water Supply 12, 321-328. |
| 20 | Melloul, A., Wollman, S., 2003. Qualitative hydrological and land-use planning tool for the Israel Coastal aquifer. The Science of The Total Environment 309, 1-17. |
| 21 | Melloul, A.J., Collin, M.L., 2001. A Hierarchy of Groundwater Management, Land-Use, and Social Needs Integrated for Sustainable Resource Development. Environment, Development and Sustainability 3, 45-59. |
| 22 | Melloul, A.J., Collin, M.L., 2003. Harmonizing water management and social needs: a necessary condition for sustainable development. The case of Israel's coastal aquifer. Journal of Environmental Management 67, 385-394. |

Table 2.2 - Selected publications for systematic review (continued)

| Item | Publication Reference |
|---|---|
| 23 | Priyantha Ranjan, S., Kazama, S., Sawamoto, M., 2006. Effects of climate and land use changes on groundwater resources in coastal aquifers. J Environ Manage 80, 25-35. |
| 24 | Ragab, R., Bromley, J., D'Agostino, D.R., Lamaddalena, N., Luizzi, G.T., Dörflinger, G., Katsikides, S., Montenegro, S., Montenegro, A., 2012. Water Resources Management Under Possible Future Climate and Land Use Changes: The Application of the Integrated Hydrological Modelling System, IHMS, Integrated Water Resources Management in the Mediterranean Region, pp. 69-90. |
| 25 | Righini, G., Raspini, F., Moretti, S., Cigna, F., 2011. Unsustainable use of groundwater resources in agricultural and urban areas: a persistent scatterer study of land subsidence at the basin scale, Ecosytems and Sustainable Development VIII, pp. 81-92. |
| 26 | Rodríguez-Rodríguez, M., Benavente, J., Alcalá, F.J., Paracuellos, M., 2011. Long-term water monitoring in two Mediterranean lagoons as an indicator of land-use changes and intense precipitation events (Adra, Southeastern Spain). Estuarine, Coastal and Shelf Science 91, 400-410. |
| 27 | Tsutsumi, A., Jinno, K., Berndtsson, R., 2009. Surface and subsurface water balance estimation by the groundwater recharge model and a 3-D two-phase flow model. Hydrological Sciences Journal 49. |
| 28 | Zhu, J., Sun, G., Li, W., Zhang, Y., Miao, G., Noormets, A., McNulty, S.G., King, J.S., Kumar, M., Wang, X., 2017. Modeling the potential impacts of climate change on the water table level of selected forested wetlands in the southeastern United States. Hydrology and Earth System Sciences 21, 6289-6305. |

Table 2.3 - Methods used to investigate spatial, temporal and modification dimensions.

| Dimension | Models | Chemical methods | Spatial analyses | Conceptual approaches | Sensitivity analyses | Physical methods | Multicriteria analyses |
|---|---|---|---|---|---|---|---|
| Spatial | 16 | 11 | 8 | 5 | 5 | 4 | 4 |
| Temporal | 11 | 8 | 4 | 3 | 0 | 2 | 0 |
| Modification | 12 | 10 | 6 | 4 | 3 | 4 | 4 |

(i) Models - mathematical representation of natural processes (Beven, 2009); (ii) Chemical methods – use of chemical tracers to monitor groundwater (Healy, 2010); (iii) Spatial analyses - analytical techniques that generate information taking into account its locations, attributes and relationships (Longley and Batty, 2003) (iv) Conceptual approaches - analyses using conceptual models or frameworks, drawing conclusions through existing knowledge and information. (v) Physical methods - based on measurements that monitor the physical process (Healy, 2010); (vi) Sensitivity analyses - evaluation of the output uncertainty based on the change of a determined input parameter (Beven, 2009) (vii) Multicriteria analyses – a decision-aid process involving the evaluation of different attributes in a transparent and structured form (Langemeyer et al., 2016)

Table 2.4 - Connection between the Landscape Scale Planning dimensions and the main groundwater management issues found in the publications

| Dimension | Alteration in recharge | Contamination of aquifers |
|---|---|---|
| Spatial | o The recharge happens with different intensities within the aquifer, this depends on the topography, location and land use (Calderon and Uhlenbrook, 2016; Guinn Garrett et al., 2012).<br>o Hydrogeological units have uncertainties such as spatial boundaries or hydrogeological characteristics that need to be taken into account in the estimates of recharge (Gondwe et al., 2011; Priyantha Ranjan et al., 2006). | o The contamination mainly depends on the physical aspects such as vadose zone depth, hydraulic conductivity or topography (Allouche et al., 2017; Hu et al., 2018).<br>o Groundwater flow direction and precipitation can spread the contaminant to rivers through baseflow (Aquilina et al., 2012). |
| Temporal | o Temporal variability needs to be taken into account in groundwater management, because the recharge process does not happen equally throughout the year (Calderon and Uhlenbrook, 2016; Daraio, 2017)<br>o Climate change can lead to alteration in the recharge process due to changes in precipitation pattern and evapotranspiration (Daraio, 2017; Ragab et al., 2012) | o Due to the physical characteristics of groundwater systems, contamination from agricultural land uses can take up to 25 years until it starts to be detected in the rivers (Aquilina et al., 2012).<br>o The time required for mitigation of contaminants depend on the hydrogeological setting. Hence, it is necessary constant monitoring to understand the behaviour of different parameters (Melloul and Collin, 2003). |

Table 2.5 - Connection between the Landscape Scale Planning dimensions and the main groundwater management issues found in the publications (continued)

| **Dimension** | **Alteration in recharge** | **Contamination of aquifers** |
|---|---|---|
| Modification | ○ Changes in land use directly affect the recharge of the aquifer altering processes of the water cycle such as evapotranspiration and infiltration (Brauman et al., 2012; Tsutsumi et al., 2009).<br>○ Urban land uses can provide indirect sources of recharge such as leakage of  pipe systems (Callahan et al., 2012)whereas the increase of impervious cover minimises infiltration and can lead to a decrease in the recharge (Guinn Garrett et al., 2012). | ○ Agricultural land use can cause the degradation of groundwater quality due to chemical fertilisers (Koh et al., 2017) and cause alteration in salinity levels due to irrigation leaching (Kurtzman and Scanlon, 2011).<br>○ Urbanisation can bring a positive effect to contamination if the change in land use was from agricultural use (Koh et al., 2017). However, there is also the possibility of degradation due to untreated waste disposal, usage of landfills and other urban anthropogenic activities (Mattas et al., 2014) |

## 2.4 Spatial dimension

All the 28 selected publications covered one or more case study areas. In total, 50 case study areas were identified, some papers analysed multiple cases. Three types of case study areas were found: i) study site refers to a point or local sites such as boreholes, wetlands (4 publications); ii) catchments are defined by the topography of a region (9 publications); and, iii) hydrogeological units correspond to geological formations capable of water storage and transfer (15 publications). The delimitation of 43 case studies was mostly influenced by visible natural features such as their geomorphology. From the 28 publications, 10 included coastal features or processes as criteria to define the study area.

The importance of analysing study sites at a local level from a spatial dimension perspective is that these are essential starting points to understand changes across larger spatial levels. Four publications had evidence of the relationship of the spatial dimension of Landscape Scale Planning with groundwater systems and groundwater management (e.g. effects of upstream land use changes can be identified in downstream discharge areas). Two studies used wetlands as a representative outlet of groundwater flow. Rodríguez-Rodríguez et al. (2011) and Zhu et al. (2017) applied a water balance to analyse the dynamics between water bodies and the aquifer. Other two publications analysed study sites that had specific characteristics that could be used to define landscape units, and focused on the physical recharge process. Kurtzman and Scanlon (2011) studied a typical soil type of the Israeli Coastal Aquifer, and Brauman et al. (2012) studied different land cover types in Hawaii.

Catchments are the spatial unit through which hydrological processes can be analysed and modelled, including groundwater systems. Of the nine publications that focused on one or more catchments as their study area, two had groundwater as the main objective. Callahan et al. (2012) analysed groundwater recharge, and Guinn Garrett et al. (2012) analysed the river-aquifer interaction. The other seven publications studied groundwater systems processes as one of their objectives, but they were not the primary topic of the publication. Hence, catchments were generally adopted as the spatial boundary when other factors were also taken into account such as streamflow (Ragab et al., 2012) or surface runoff (Calderon and Uhlenbrook, 2016). Among these seven publications, only Guan et al. (2010) considered the possibility that a subsurface flow between catchments might occur. Groundwater recharge (Calderon and Uhlenbrook, 2016; Tsutsumi et al., 2009), assessment of groundwater quality (Aquilina et al.,

2012) and climate change impacts (Daraio, 2017; Ragab et al., 2012) were analysed to provide information for groundwater management.

Most publications adopted hydrogeological units, or parts of it, as their spatial boundaries. As these hydrogeological units generally are large formations that can extend across several catchments (Dawes et al., 2012; Gondwe et al., 2011), other characteristics were used to define the spatial boundary. For example, He et al. (2009), Hu et al. (2018), and Righini et al. (2011) delimited their study areas by using the intersection of the hydrogeological unit with the overlying coastal plain, and Collin and Melloul (2001) combined the hydrogeological unit with social aspects. Koh et al. (2017) and Mair et al. (2013) conducted their studies on an island (i.e. Jeju Island, Korea). Mair et al. (2013) studied the occurrence and recharge of groundwater and Koh et al. (2017) investigated the effects of land use change on groundwater quality. Both publications described Jeju Island as having several catchments in their hydrogeological unit, but their analyses did not consider individual catchments.

## 2.5 Temporal dimension

Evidence of the temporal dimension was found in 16 publications. Past analyses of land uses were done in 13 publications. The identification of which publications carried out past analyses comprised not only the timespan of the data set but also if the data were discussed or correlated with the past modifications. Future estimates were analysed in 5 publications. Only Dawes et al. (2012) and Aquilina et al. (2012) considered both past and future land use changes.

In the publications that analysed past data, the time length used varied according to the objective of the study. While Calderon and Uhlenbrook (2016) used a short period of three years to analyse groundwater recharge and climate water balance in a catchment, Aquilina et al. (2012) analysed a nitrate concentration series of 38 years to evaluate the alterations due to agricultural land use in Brittany. Both analysed groundwater systems and the implication of these processes to groundwater management. Koh et al. (2017) analysed 11 years of data using chemical methods to evaluate the effectiveness of groundwater management measures to prevent saltwater intrusion. The results started to become visible after six years of implementation. When analysing a sparse time series, other publications could go even further back. The longest analysis was made by Kurtzman and Scanlon (2011) who compared the changes in chloride levels between the years of 1935 and 2007 due to extensive settlements and modern cultivation on an Israeli coastal aquifer. These publications analysed aspects or measures of groundwater management under a historical perspective based on past data.

The five publications that analysed future conditions investigated the response of groundwater systems to determine changes. Four publications applied scenarios related to climate change and Global Climate Models. Starting from past data analysis, Dawes et al. (2012) analysed future scenarios including climate and land use change up to 2030 and Ragab et al. (2012) analysed land use and climate change scenarios until 2100. Zhu et al. (2017) and Daraio (2017) applied scenarios of climate change until 2100. Aquilina et al. (2012) were the only ones to approach the water quality aspect under a future perspective. These futures estimates were based on trends detected. The future scenarios analysed did not include changes due to other factors such as socio-economic pressures.

## 2.6 Modification dimension

Aspects of anthropogenic modification of land use were present in 23 of the 28 selected publications. For example, three main types of land use changes were shown in the publications: agricultural, urban and forestry. While agricultural land use was present in all 23 publications, urban land use was present in 18 publications, and forestry land use was present in 15 publications. These publications connected the impacts of these land uses with groundwater quantity (Brauman et al., 2012; Callahan et al., 2012) and quality (Hu et al., 2018; Koh et al., 2017). Current land uses within the study area were analysed by ten publications, whereas past land use change processes were analysed in 15 publications. Only (Collin and Melloul, 2001) and (Dawes et al., 2012) analysed both current and future impact of land use changes on groundwater resources.

When current land uses were evaluated, the publications studied mostly agricultural or urban land uses and their impacts on groundwater systems, and possible implications for groundwater management. The main anthropogenic impact detected was the contamination of aquifers, especially due to agricultural activities. This included presence of nitrate (Allouche et al., 2017; Mattas et al., 2014) and phosphate (Mattas et al., 2014) above the desired levels for drinking water consumption. Issues directly related to coastal areas were not much explored. Only four papers studied directly land use changes and saltwater intrusion. For example, Priyantha Ranjan et al. (2006) estimated the correlation between deforestation and aridity index losses in fresh groundwater due to saltwater intrusion. Where the impacts of urban land uses were investigated, high levels of chloride were detected in groundwater (Melloul and Collin, 2003). The main type of land use change observed was from pre-development (natural) conditions to agriculture and from agricultural to urban (residential) uses. These land use changes were found to alter groundwater recharge (He et al., 2009), water quality (Aquilina et al., 2012) and aquifers dynamics (Guan et al., 2010). Saltwater intrusion and overexploitation of groundwater were also related to changes in land use (McFarlane et al., 2012; Melloul and Collin, 2003).

## 2.7 Methods applied by publications

Modelling was the most applied method by the publications to analyse the spatial dimension. At the site level, Kurtzman and Scanlon (2011) analysed recharge through physical methods and modelling, and Brauman et al. (2012) used water balance to identify the primary components of the recharge. At the catchment level, six out of the nine publications applied some modelling technique, with three applying scenario analysis. Regarding hydrogeological units, the most common method applied was also modelling. Numerical models were applied to analyse alteration in recharge (He et al., 2009); effects of climate change on groundwater resources (Dawes et al., 2012); the uncertainty to define the actual hydrogeological areas (Gondwe et al., 2011); and to evaluate alteration in groundwater quality (Priyantha Ranjan et al., 2006). Other methods were spatial analysis (Righini et al., 2011), multicriteria analysis (MCA) (Melloul and Wollman, 2003), and case study analysis using a conceptual approach (McFarlane et al., 2012).

Modelling was used by 11 publications to analyse the temporal dimension, including scenario analysis (7 publications). Models based on the water balance equation were the mostly used. Modelling based on past data was used by Mair et al. (2013) (18 years of data) and Calderon and Uhlenbrook (2016) (3 years of data). Chemical methods were used in eight publications. The shortest period used for the application of chemical methods was nine years (Guan et al., 2010). Chloride and nitrate were the most used indicators for chemical methods (e.g. Calderon and Uhlenbrook, 2016; Guan et al., 2010). Other methods applied were case study analyses through a conceptual approach (Mattas et al., 2014; McFarlane et al., 2012), spatial analysis with remote sensing (Righini et al., 2011), and physical methods (Aquilina et al., 2012; Hu et al., 2018). To analyse future conditions, five publications applied modelling techniques to simulate future scenarios. Aquilina et al. (2012) were the only ones to approach the water quality aspect under a future perspective.

To investigate the modification dimension, 13 publications applied modelling to analyse land use impacts on groundwater resources, including scenario analysis (8 publications). Models based on the water balance equation were applied eight times. Models based on the flow equation were applied in four publications (Dawes et al., 2012; Kurtzman and Scanlon, 2011). While the water balance based models were generally applied to evaluate impacts on groundwater recharge due to changes in land use, flow equation models were applied to analyse impacts not only on recharge but also on the behaviour of the aquifer. Only Daraio (2017) applied a surface hydrologic model. He analysed the groundwater component of the water cycle taking into account the current land uses in the study area. Another frequent method applied was the analysis through chemical methods, used in ten publications (e.g. Allouche et al., 2017; Rodríguez-Rodríguez et al., 2011) to estimate alterations in groundwater recharge (Guan et al., 2010; Kurtzman and Scanlon, 2011) and contamination of aquifers (Aquilina et al., 2012; Koh et al., 2017). Chloride, nitrates and electrical conductivity were the indicators mostly used. Other methods used were spatial analysis (Dawes et al., 2012; Mair et al., 2013), physical methods (Callahan et al., 2012; Kurtzman and Scanlon, 2011), conceptual approaches (Mattas et al., 2014; McFarlane et al., 2012), MCA (Collin and Melloul, 2001; Hu et al., 2018), sensitivity analysis (Allouche et al., 2017), and statistical methods (Daraio, 2017; Koh et al., 2017).

## 2.8 Discussion

Balancing groundwater exploitation with demands from natural processes and anthropogenic uses comprises the key challenge for groundwater management (Jakeman et al., 2016). This challenge is further compounded by ongoing population and urbanisation growth (Tam and Nga, 2018), along with climate change impacts (Cuthbert et al., 2019). The search for a framework capable of dealing with such aspects is still a challenge that needs to be overcome for groundwater management (Vadiati et al., 2018). Additionally, coastal urbanised areas relying on groundwater supplies present a set of issues that are both complex and difficult to address without a more holistic perspective to groundwater management (Michael et al., 2017). One such holistic perspective is the Landscape Scale Planning approach (Selman, 2006), which not only takes into consideration the distinctive features and characteristics of landscape units but also spatial and temporal changes in land uses and inherent impacts caused on groundwater resources. The implementation of the Landscape Scale Planning approach however, requires groundwater resources related information that can address the complexity of issues affecting them - especially information that sheds light on the spatial, temporal and modification dimensions. Findings from this study indicate that current groundwater science are not thoroughly dealing with such complexity.

Fisrtly, findings indicated that from the spatial dimension, groundwater systems are usually studied and managed at only one level, as the catchment or hydrogeological basin, in an attempt to analyse the region into self-contained units. However, according to (Sanz et al., 2015), the occurrence of groundwater does not always coincide with these units. Part of what forms a landscape are the spatial differences. These differences such as topography, geology and land use can influence the aquifer, and as consequence, the recharge process or the contaminant path in the aquifer. Although evidence of this relationship was present in selected publications (Calderon and Uhlenbrook, 2016; Hu et al., 2018), many times these differences within the study area were disregarded. Recently, other scholars (Fan et al., 2018) have also shown that different parts of the hydrological system, including groundwater, have differing spatial characteristics that need to be taken into account. This implies that the knowledge required for managing groundwater systems needs to be built from the local scale to the whole landscape scale. Therefore, the use of landscape units underpinned by physical, socioeconomic, cultural and other factors can aggregate similar spatial characteristics that are critical for informing groundwater management. One of the benefits of considering the Landscape Scale Planning approach would be the possibility of connecting multiple features of coastal landscapes such as shorelines, beaches and estuaries to the whole groundwater management. Submarine groundwater discharge, base flow in estuary regions and saltwater intrusion are coastal groundwater management issues that affect such features (Coelho et al., 2018; Robinson et al., 2018) but had little attention in the selected publications.

Secondly, from a temporal dimension, a significant amount of publications focused only on the current state of a groundwater system, or on the provision of information for groundwater management for the current situation. When analysing the temporal dimension, many publications sought to establish an average situation of the system based on past data as a baseline (Mair et al., 2013), and identified additional disturbance as external drivers (Mattas et al., 2014). Building on this information, some publications developed future scenario for analysis based on detected trends (Zhu et al., 2017) or future drivers (Aquilina et al., 2012). Consideration of future changes to groundwater systems, however, was not a predominant trend in the publications analysed. This presents a gap in knowledge because these systems are, and will continue to be, confronted with future uncertain changes in rates of population and urbanisation growth, demands from natural processes and climate change impacts. Additionally, there was no application of participatory approaches to determine the future scenarios to be analysed. Workshops with stakeholders can be applied to generate scenarios regarding future situations (Faysse et al., 2018). The impacts of these drivers should be incorporated to the provision of information, or strategies, for groundwater management. Recently, Currell et al. (2016) proposed a framework to guide the determination of the importance of transient behaviour to the management timeframe based on hydraulic response times, diverse hydrological drivers (natural and anthropic) and spatiotemporal interest. The application of this framework could be used to determine the timescale of the impacts to the aquifer. Therefore, management actions could consider more long-term, strategic planning horizons.

Thirdly, from a modification dimension, the publications gathered accounted for many analyses of land use changes but without fully considering more dynamic aspects. While some of these dynamics were included in analyses of groundwater management issues based on the historical evolution (Mattas et al., 2014), they were not considered in analyses exploring future situations. The manner in which different pressures alter land use patterns and groundwater systems under a longer planning horizon is often disregarded, despite its importance. For instance, Gashaw et al. (2018) and Kundu et al. (2017) noted that when the process of change is considered throughout the planning horizon, the difference can be of up to 20% of the groundwater balance. Teklay et al. (2019) found that management approaches considering the dynamic nature of land use change can significantly result in differing outcomes because it can increase the spatial and temporal accuracy of the hydrological information provided. Landscapes can be formed through a matrix of different and competing land use patterns. Thus, the dynamic aspect of changes needs to be acknowledged as well as the relationship between these land uses (Mercau et al., 2015) not only regarding their physical aspects but also the socioeconomic or cultural ones. Coastal areas have become centres with high levels of urbanisation, agriculture, and seasonal population increases due to tourism (Michael et al., 2017). These drivers are causing an increase in saltwater intrusion, alterations in both groundwater availability and quality (Petelet-Giraud et al., 2018). Including these dynamics in groundwater management could support a move towards a more interdisciplinary approach - one of the objectives of the IWRM (Foster and Ait-Kadi, 2012).

Finally, findings from this study indicate that most of the limitations associated with providing information for managing groundwater under a Landscape Scale Planning approach appear to be related to existing methods of analysis. However, some methods have the potential to overcome some of these limitations. For example, to address the spatial dimension, water table fluctuation can be spatialized to a broader scale (Healy, 2010), while groundwater flow models can be applied to understand the behaviour of the whole aquifer (Anderson et al., 2015a). Chemical or physical methods have both been applied to increase the understanding of the study area characteristics and problems. Although these methods are point-based, they can serve as part of a monitoring effort to provide information through the detection of alteration in groundwater quality and quantity for management purposes (Rohde et al., 2017; Thomas, 2018). For this, their outputs need to be scaled up from a point-based to the whole landscape. Similarly, they can deal with different spatial levels and provide information and support to management processes (Pezij et al., 2019). From a temporal perspective, these methods cannot simulate future conditions. In order to do that, models, which can be underpinned by chemical or physical methods, need be applied. However, the uncertainty inherent to all these methods cannot be removed and this needs to be acknowledged by management processes (Neuendorf et al., 2018). Regarding the modification dimension, some proposed methods are capable of encompassing the dynamics of change in land use for future scenarios (Dang and Kawasaki, 2017; Han et al., 2015; Lima et al., 2011). Yet, these methods have not been widely used, especially to inform groundwater management.

## 2.9 Guidelines for groundwater management based on Landscape Scale Planning

Underpinned by the three dimensions of Landscape Scale Planning, the following management guidelines for groundwater are suggested. These dimensions have to be considered simultaneously. They present a continuous approach seeking to incorporate both the dynamics of change through land use and the dynamics of the groundwater systems to inform groundwater management. Additionally, the Landscape Scale Planning approach does not conflict with any of the methods applied or developed in previous groundwater management studies. Therefore, the challenge to adopt the Landscape Scale Planning approach is more related to how to frame the analysis to effectively provide information for groundwater management decisions rather than a methodological barrier (Figure 2.2 and Figure 2.3):

i) Spatial Dimension – this dimension can better inform groundwater management through the application of landscape units. These landscape units are determined using the hydrogeological, hydrological, morphological and land use patterns of a landscape. The inclusion of land use to determine the boundaries of landscape units is a proxy for the inclusion of socio-economic and cultural aspects, given that these aspects directly influence both groundwater management and land use changes (Li et al., 2018). Therefore, these landscape units do not need to obey the political boundaries and do not need to coincide with the catchments' limits. This is because there could be groundwater flow between adjacent surface catchments, but still within the same aquifer. Local-scale analysis at study sites using physical and chemical methods can provide valuable information to increase the understanding of the groundwater system, while spatial analysis brings together this information to define landscape units.

ii) Temporal Dimension – the management should be based on the evaluation of past and current data in the area to be managed, using both data physical, such as water table depth, and socioeconomic, such as water demand, population growth, and institutional constraints, in order to develop possible storylines aiming at different management objectives. Using this analysis, it is possible to determine how the information should be provided for groundwater management, through a series of steady state or transient models. These models can simulate past behaviour of the groundwater system considering the diverse historical drivers that led the system to the current state, from this point multiple possible future situations could be evaluated.

iii) Modification Dimension – for an effective management of groundwater both land use changes and anthropogenic pressures must be included in the management process. The dynamic aspect of change has to be considered in management decisions. The drivers that brought the system to the current state also have to be analysed, and future changes along the planning horizon such as the future land uses must be included. A participatory process involving a range of stakeholders can be applied to develop future perspectives and storylines for a scenario analysis that incorporates the societal pressures. This inclusion would aid the conceptualisation of both the final situation and how the diverse land uses would change until this situation is achieved.

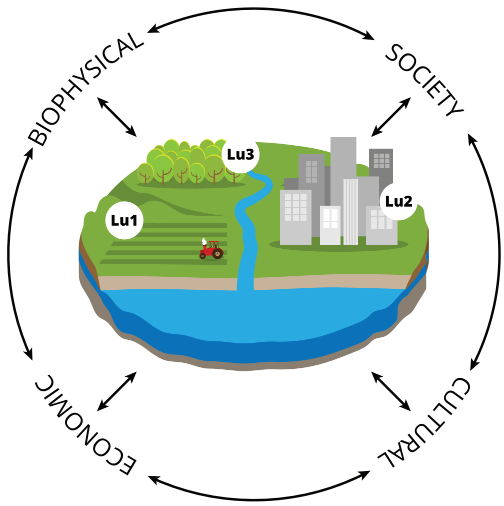


Figure 2.2 - The landscape unit concept

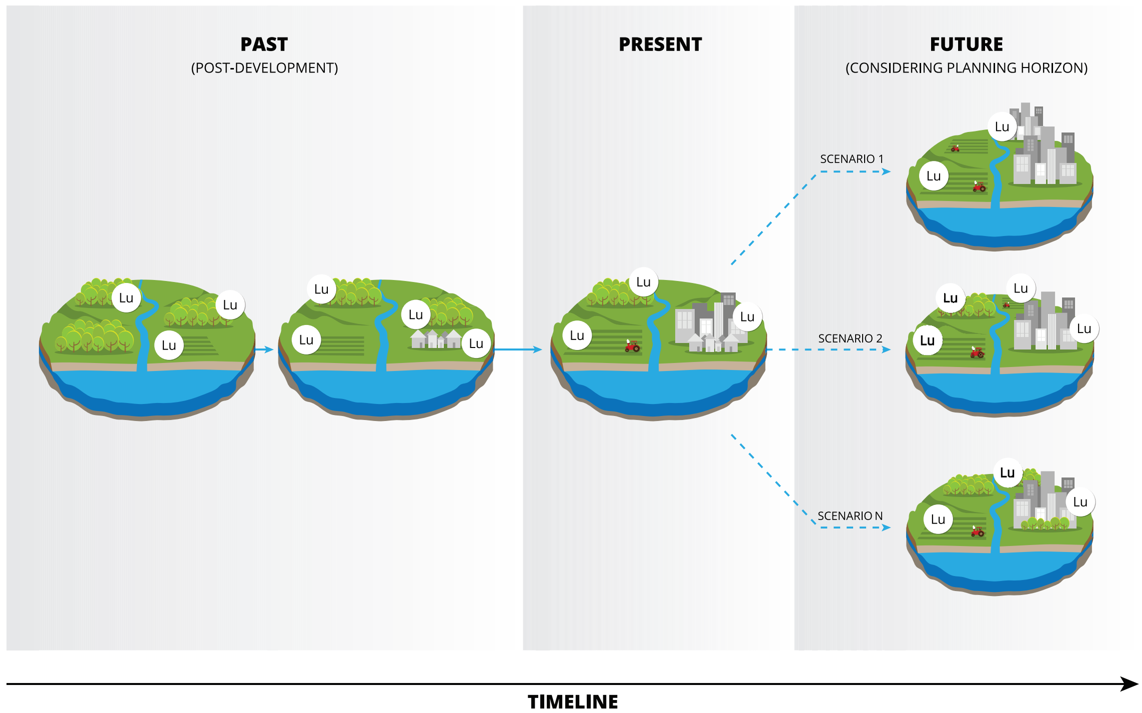


Figure 2.3 - The dimensions and dynamics of Landscape Scale Planning for Groundwater Management

## 2.10 Conclusion

Using a systematic literature review, this paper sought to analyse how the understanding of the groundwater management and land use change in coastal areas could be improved through the application of a Landscape Scale Planning approach. From the selected publications, it was noted that the spatial, temporal and modification dimensions have an intrinsic connection with groundwater management. This connection was confirmed by how each dimension relates to characteristics of groundwater systems, including the different spatial and temporal distribution of recharge as well as the strong influence that land use modifications cause on the aquifer. Furthermore, this study identified a gap regarding the inclusion of land use change dynamics to inform current groundwater management approaches. This consideration is important because it can provide more accurate information to management processes. Applying a Landscape Scale Planning approach to frame this connection can therefore lead to more comprehensive outcomes and improve groundwater management processes. Landscape Scale Planning can be applied as an integrative framework because it connects different sectors and perspectives. This can aid groundwater management to take into account differing spatial and temporal characteristics of the aquifer.

Two main issues that emerged from the review were alterations in recharge and contamination of the aquifer. These issues happen in a heterogeneous way within the aquifer and are influenced by several aspects such as topography, geology, and current, past and future land uses. Landscape units can bring together these distinct aspects and characteristics of the aquifer to improve groundwater management and secure the aquifer's future sustainability. Issues such as saltwater intrusion and submarine groundwater discharge had little attention in the publications gathered. These issues could be better addressed under a Landscape Scale Planning approach because it enables the integration with other sectors, such as estuarine planning.

For the application of the proposed guidelines, the groundwater system has to be divided into landscape units. These landscape units are determined according to the hydrological and hydrogeological characteristics, based on which management boundaries could be determined to consider the existing land use patterns. With the information organised within landscape units, management objectives could be determined and evaluations from the historical to the current state conducted. The possible changes to the system would be incorporated through analysis of multiple future scenarios generated based on the societal, economic and cultural pressures. These scenarios would allow the exploration of different alternatives to provide information for groundwater management, aiming at including stakeholders and seeking a self-reinforcing relationship among stakeholders and the groundwater system.

Suggestions for further research include the evaluation of groundwater management strategies in non-coastal environments underpinned by all the three dimensions of Landscape Scale Planning. Other studies could analyse the effects of the dynamic aspects included in the modification dimension of typical coastal groundwater management issues such as saltwater intrusion. Finally, given that landscape units would be determined according to evolving land use changes, they are able to capture both the groundwater system dynamic and changes through time. But this comprises another research challenge.

# Chapter Three - Mismanagement of Groundwater in a Cultural Landscape: the João Pessoa case study

## 3.1 Introduction

Groundwater recharge in tropical areas has its maximum during the wet season (Jasechko et al., 2014), whereas the demand also has its seasonality aspect (Wada, 2008). Hence, a higher rate of groundwater recharge does not necessarily mean that demands will be sustained. Furthermore, anthropic global warming can alter the precipitation patterns causing changes in the rates of groundwater recharge (Jasechko and Taylor, 2015). Besides, the sustainability of groundwater systems is dependent on the effective management of their resources; however, only recently this has been acknowledged by the legislations (White et al., 2019) and applied as institutional tools for groundwater management (Albrecht et al., 2017).

As mentioned in the previous chapter, coastal regions also have groundwater systems facing sustainability issues due to anthropogenic pressures caused by unplanned social and economic development. Among the diverse issues, groundwater depletion and saltwater intrusion are rising concerns in many of these regions (Michael et al., 2017). Moreover, almost every coastal city is expected to continue growing in the upcoming decades (GRID-Arendeal and UNEP, 2016). Therefore, there is an urge to analyse, inform and support sustainable groundwater management in areas subjected to these conditions, not only for the increase of water security to society but also to ensure a healthy status to groundwater-dependent ecosystems (Albrecht et al., 2017; Erostate et al., 2020).

An example is a cultural landscape located at the coastal area of Paraíba State, Northeast of Brazil, delimited by part of the intersection between Paraíba-Pernambuco Sedimentary Coastal Basin and both the Gramame river basin and part of the Paraíba river basin (Figure 3.1). This area includes the state capital João Pessoa and several surroundings municipalities (Alhandra, Bayeux, Cabedelo, Conde, Cruz do Espírito Santo, Pedras de Fogo, Santa Rita e São Miguel de Taipu), representing the leading economic region of the state with more than one million inhabitants and with an area covering 1,032.31 km². Batista et al. (2011) and Braga et al. (2015) have detected indications of groundwater over-exploitation problems in this area through the application of hydrogeological modelling. Furthermore, there have been significant changes in land use over the last decades, such as a considerable increase in the urban area, and changes from natural cover (forest and mangrove) to farming (agriculture and pasture).

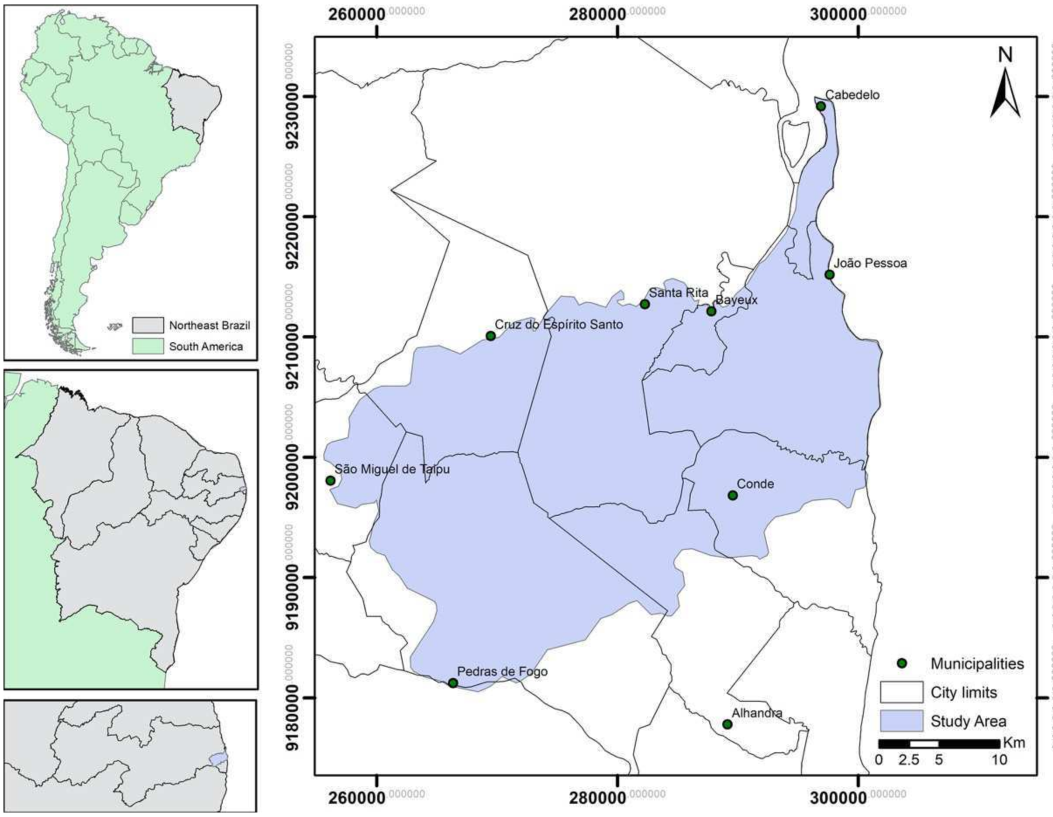


Figure 3.1 - Selected Study Area for the Thesis

Therefore, the objective of this chapter is to present the characteristics of this cultural landscape, including the natural and anthropogenic drivers and the institutional framework that guides the land use and groundwater management in the region. Afterwards, it is shown perspectives for the integration between land use planning and groundwater management, and the existing issues from the current groundwater management arrangement that have been leading the aquifer to a possible over-exploitation. Finally, possible alternatives to increase groundwater sustainability are presented.

## 3.2 Previous studies

The first hydrogeological studies in Brazil started to appear at the end of the 1940's and beginning of the 1950's, as part of the characterisation of some water systems in the North region, conducted by the *Instituto Agronômico do Norte* (Sioli, 1951, 1946), and in the South region, conducted by the *Instituto Agronômico do Sul* (Vassao, 1952). However, more complex and dedicated hydrogeological studies started at the 1960's with projects conducted by the Northeast Development Bureau (Superintendência do Desenvolvimento do Nordeste -

SUDENE). These projects were motivated by the guidelines of the first SUDENE's director plan. Among these guidelines was the improvement of the harnessing of natural resources in the Northeast region, having as one of the fields of work the rational use of water resources, with investments allocated to hydrogeology (Mota, 1963). These studies were published between 1959 and 1989. Special attention must be given to the series *Hidrogeologia* (1963-1988), in which diverse aspects of the Northeast's hydrogeology were analysed. This series represented a milestone in the knowledge of the region.

The coastal sedimentary basin Paraíba – Pernambuco was included in the SUDENE's studies through the *Inventários Hidrogeológicos Básicos – Folhas 11, 16, 21* (Manoel Filho, 1970; SUDENE, 1978a, 1978b). The following advance in the hydrogeological knowledge of the study area occurred with the work of Costa et al. (2007). In this study, it was analysed the groundwater availability of the sedimentary basin within the state of Paraíba. Afterwards, the ASUB project (ASUB, 2010) analysed the intersection of the sedimentary basin Paraíba – Pernambuco with the lower basin of the Paraíba River. This project resulted in considerable advances in hydrogeological knowledge and groundwater management approaches. As main results, it can be highlighted: i) the proposition of management zones based on several criteria; ii) the proposition of criteria for water rights and the charge of groundwater use (ASUB, 2010). Even though several projects were conducted in the study area, a regional groundwater numerical model has not been achieved, only a few local models exist (Batista et al., 2011; Braga et al., 2015). Finally, new advances were possible with the BRAMAR Project. Through this project, it was possible to define and assess climate change and socio-economic scenarios, refine the conceptual model for hydrogeological modelling and identify strategies seeking the Integrated Water Resources Management (Schimmelpfennig et al., 2018; Walter et al., 2018).

## 3.3 Climate

According to the Köppen classification, the climate of the study area can be classified as a tropical climate (As'). This means hot and dry summer with the wet season in autumn and winter with a marked transition from the dry to the wet season and low variation of the average temperature throughout the year (Francisco and Santos, 2017; SEMARH, 2000). Historical climatological data was available only at one location in the study area. This station was located in the city of João Pessoa, and the data comprised the period of 1961 – 2010. The analysis included two periods: from 1961 – 1990, and from 1981 – 2010. These periods were determined following the climatological normal determined by the World Meteorological Organization. A climatological normal can be defined as a 30-year period in which average climatological data are computed (WMO, 2017). The most recent climatological normal was utilized to describe the climatic characteristics of this cultural landscape.

### 3.3.1 Precipitation

For the climatological normal between 1981 – 2010, the wet season spans from March to August and includes more than 80% of the annual precipitation. Consequently, the dry season spans from September to February. The monthly distribution of the precipitation is depicted in Figure 3.2. For the climatologic normal the average accumulated precipitation was 1,914 mm/year (Diniz et al., 2018), and the month with the highest amount of rainfall was June.

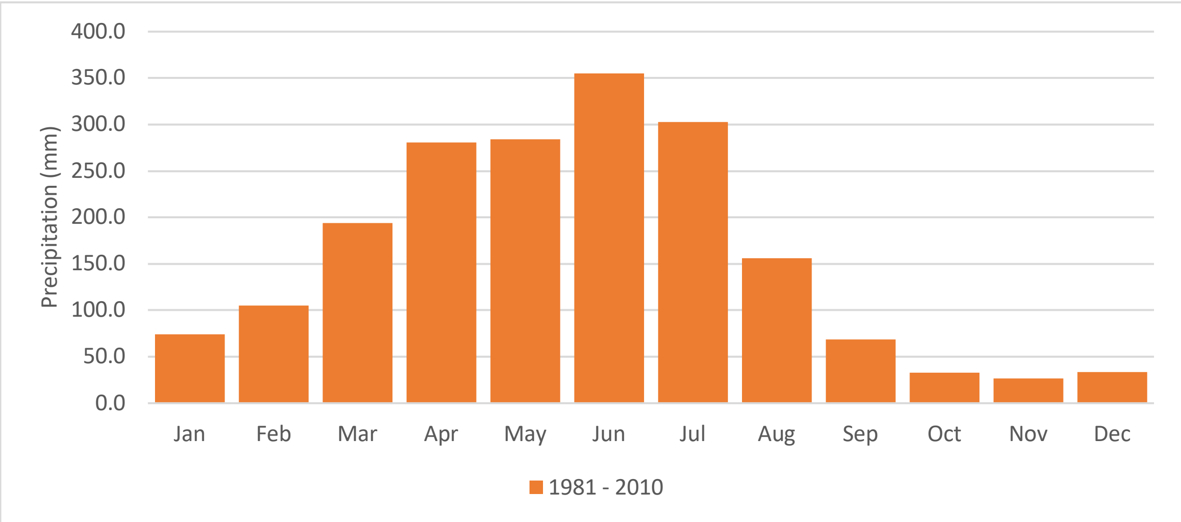


Figure 3.2 - Average Monthly Precipitation

### 3.3.2 Temperature

In the study area, there is little variation in the temperature across the year. A chart showing the average minimum, maximum and medium temperature for the climatological normal is presented in Figure 3.3. The mean temperature ranged between 25.1 C (July) to 27.9 C (February). The monthly maximum mean temperature was highest in March (30.9 C) but closely followed by February (30.8 C) and lowest in July (28.4 C), while the monthly minimum mean temperature had it highest in December (25 C) and lowest in August (21.8 C) (Diniz et al., 2018). At general, February can be assumed as the hottest month while the coldest is July.

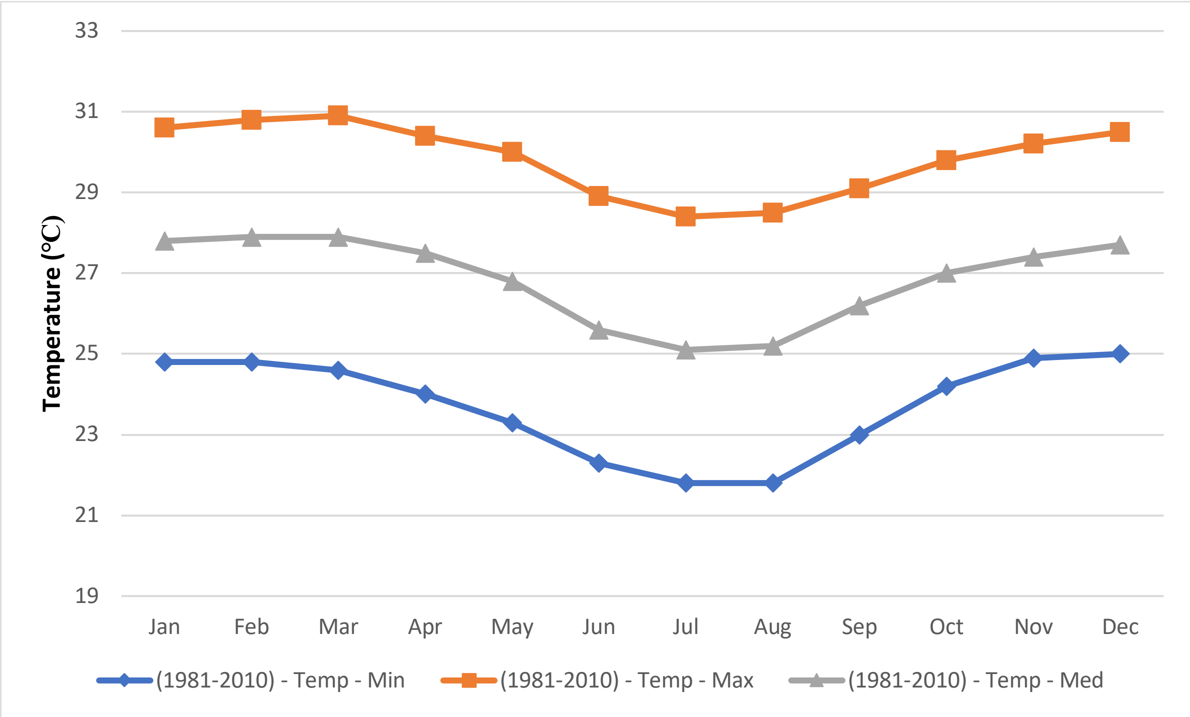


Figure 3.3 - Average Monthly Temperature (Maximum, Minimum, Mean)

### 3.3.3 Potential Evapotranspiration

As expected, the potential evapotranspiration is higher during the summer. The annual ETP is 1,976.22mm with the highest in March (193.4mm) and the lowest in July (131.2mm). The annual variation of ETP is depicted in Figure 3.4. Even though the relative humidity and insolation also affect the evapotranspiration; in the study area, the ETP follows the temperature. The driest month is October with a relative humidity of 72%, and the most humid is in June with 81.9%. Regarding the insolation, the annual number of hours in which the sunlight reaches the surface is 2,731 hours. October is the month with the maximum insolation (274.1 h) and June the lowest (173.2 h) (Diniz et al., 2018).

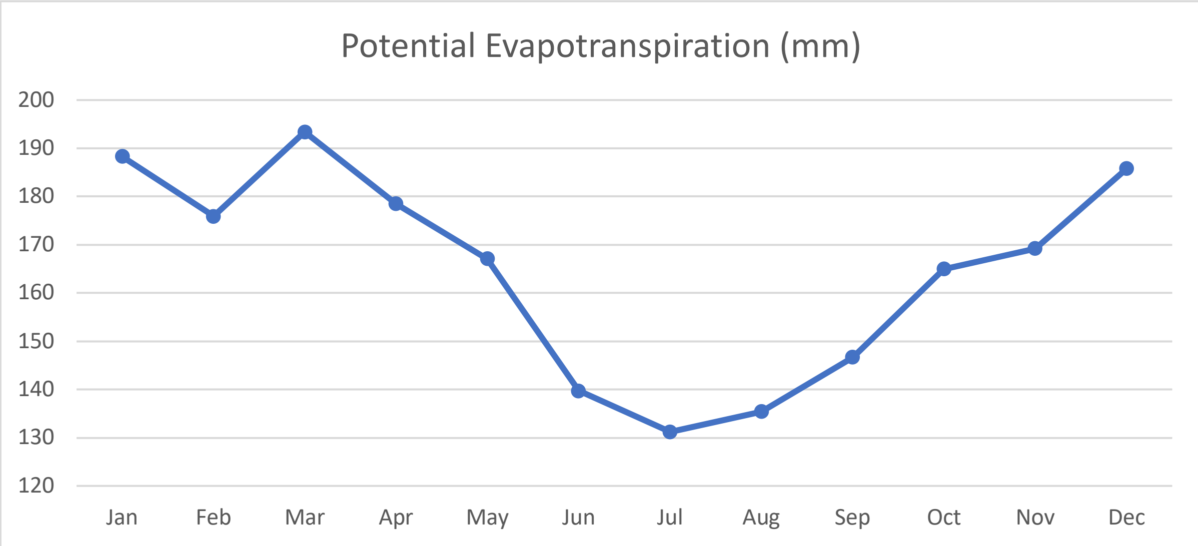


Figure 3.4 - Potential Evapotranspiration 1981 - 2010

### 3.3.4 Climatic Water Balance

The Climatic Water Balance (CWB) refers to the difference between precipitation and potential evapotranspiration during a specified period. The chart showing the monthly water balance is presented in Figure 3.5. The water surplus is apparent during April and July, while a more evident deficit can be identified from September to February. At total, a deficit of 62mm was identified for the period (Diniz et al., 2018).

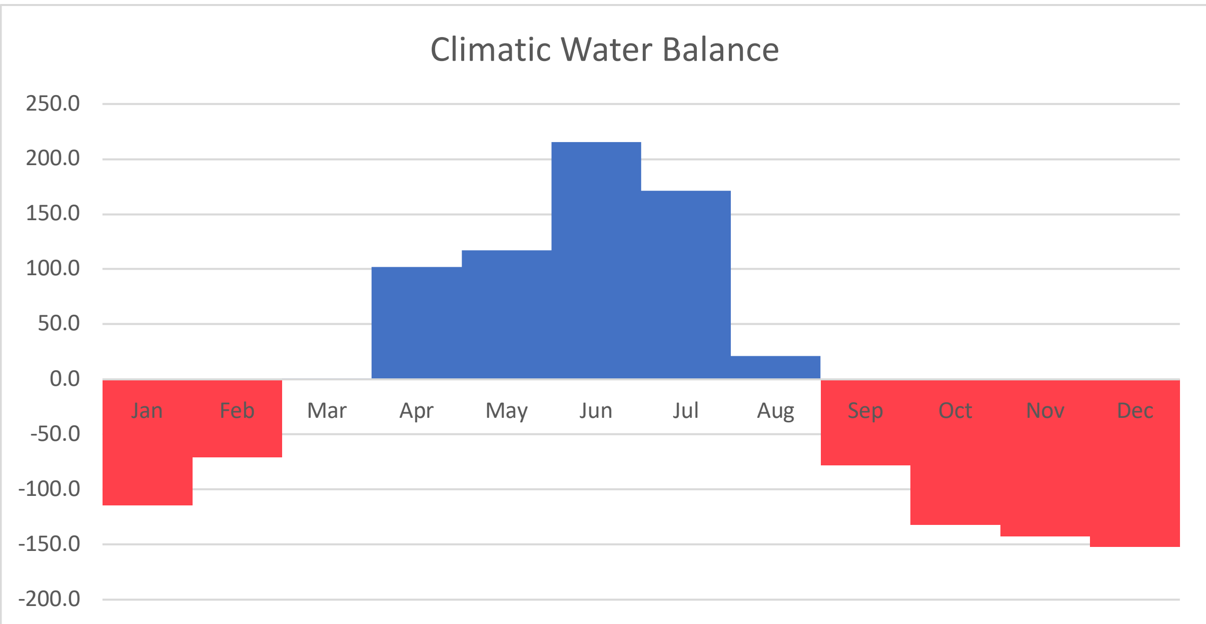


Figure 3.5 - Climatic Water Balance - 1981 - 2010

## 3.4 Geomorphology

The study area is situated on three different geomorphological units: the coastal plateau, coastal plains, and coastal lowlands (AESA, 2006). The coastal plateau is a plateau that reaches a minimum of 30m close to the shoreline; this plateau dominates most of the Lower Course Paraiba River Basin and Gramame River Basin. The coastal plains are morphological units that occur near the shoreline with an average altitude of 3m above sea level. These plains are constituted by sea and river sediments with the presence of clay and sand. In these plains, there are the presence of mangroves and some alluvial formations. Finally, the coastal lowlands are formed by narrow beaches reaching the sea and river estuaries. These lowlands have the presence of sand deposited from maritime and wind processes (Costa et al., 2007). A map with these units is presented in Figure 3.6.

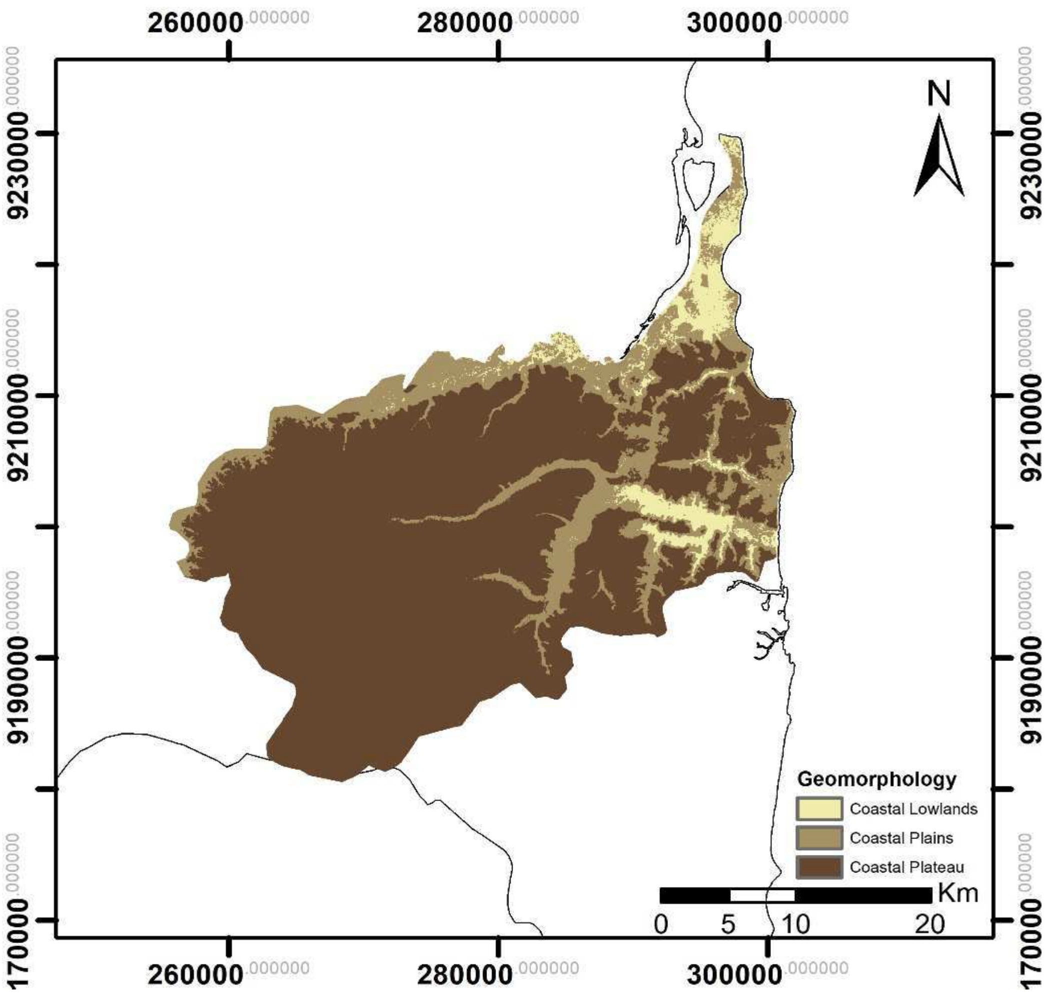


Figure 3.6 - Geomorphology of the Study Area (modified from Costa et al., 2007)

## 3.5 Hydrology

The study area presents two river basins: the Gramame River Basin and the Lower Course Paraíba River Basin. While the former is inserted in the study area, the latter has only a portion of its territory included (Figure 3.7). The sedimentary basin Paraíba-Pernambuco is underlying both basins.

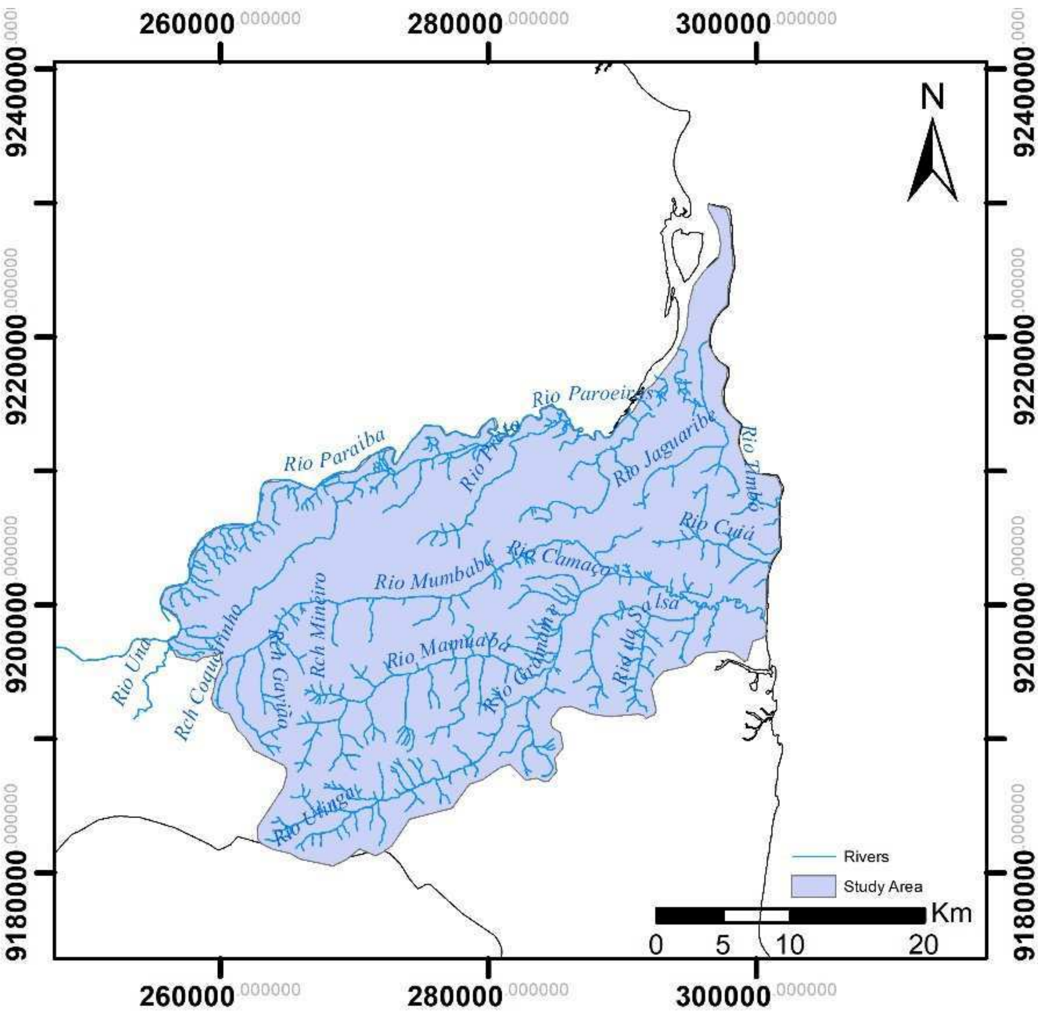


Figure 3.7 - Hydrography of study area (modified from AESA, 2006)

The Gramame River Basin has an area of 590 km$^2$. This basin has major importance to the region, given that it supplies approximately 25% of the population of the state of Paraíba, including the capital João Pessoa and other surrounding municipalities such as Cabedelo, Santa Rita and Bayeux. The withdrawal for the water supply of this region happens within the Gramame-Mamuaba System, that has a capacity of 56 Hm$^3$, at a nearby elevation dam. The Gramame River Basin follows the main river – Gramame – and its main sub-basin: the

Mamuaba, the Mumbaba, and the Água Boa. Besides that, the drainage network of the river is composed of another 11 main tributary rivers. The drainage density (Dd) of the basin is 1.23 km/km$^2$; therefore, it can be classified as low to medium drainage (SEMARH, 2000).

The Lower Course Paraiba River Basin has, at total, an area of 3,925.40 km$^2$. However, only a portion of this area was selected to compose the study area: with 442km$^2$. The area selected is located at the south of the Paraíba River. The reason was to use the Paraíba river as a boundary condition. Besides, north of the river, the data availability was highly scarce. This portion of the basin also presents significant importance to the region. In this area, the municipalities of João Pessoa, Cabedelo, and Bayeux are included. Therefore, this basin has a large urbanised area that resulted in many alterations of the surface flow. The Lower Course of Paraiba River Basin follows the main river – Paraíba. The selected subregion has four tributary and other rivers. Among them, the Jaguaribe, Cuiá, and Mandacaru rivers, that does not converge to the main Paraíba river, but crosses urban areas with high density and presents a high level of contamination (AESA, 2006).

## 3.6 Hydrogeology

The chosen study area has three hydrogeological formations: Barreiras formation, Gramame formation and Beberibe formation. The Barreiras formation is a phreatic aquifer with an average thickness of 42 meters. This aquifer is formed mainly by sandy-clay sediments. In some regions of this formation, marine-fluvial sediments and alluvial deposits can be found. These packages have a thickness between 10 and 60 meters, constituted by fine and medium-grained sand. The Gramame formation is located east to the geological fault and over the Beberibe formation. This formation is constituted by limestone with an average thickness of 50 meters and acts as the superior confining layer to the Beberibe formation. The Beberibe formation is situated on top of the bedrock, and it exists in the whole study area with an average of 200 meters of sandstone. East of the geological fault (green line – Figure 3.8) this formation is classified as a confined subsystem and west of the fault as a phreatic aquifer.

In the western region of the study area, there is a presence of crystalline outcrop. There is granitoid outcrop in upstream of Gramame River and granitic outcrop south of the Paraiba River. In the upstream region of Mumbaba River, there is the Sertânia complex, characterized by gneisses and limestone rocks.

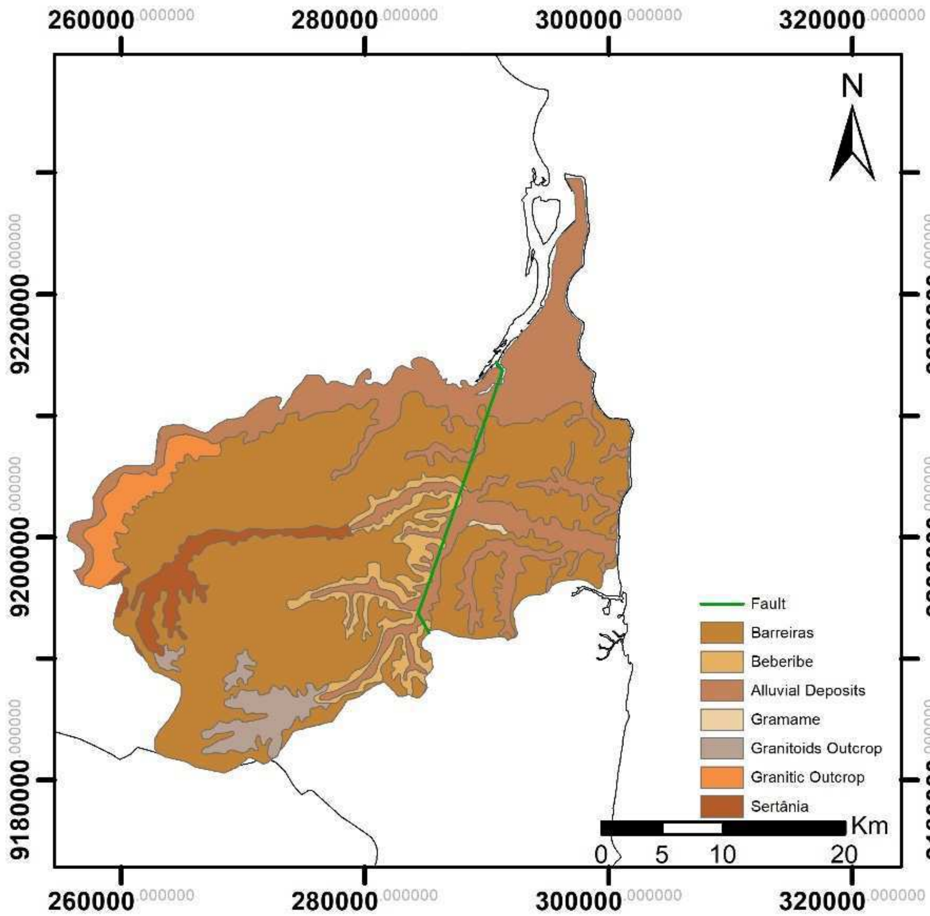


Figure 3.8 - Geology of the study area (modified from Costa et al., 2007)

Climatic characteristics offer an excellent condition to the recharge of the phreatic aquifer. The recharge occurs mainly during the wet season due to the infiltration of precipitation and, in a smaller amount, from the rivers. Estimates of recharge rates due to rainfall in the area varies between 5% to 30% of rainfall (Fernandes, 2017). However, other studies have shown lower recharge rates, between 10% and 15% (Costa et al., 2007; SEMARH, 2000). To the confined subsystem, the recharge occurs due to subterranean flow from the phreatic aquifer. There is no outcrop on the confined region.

The flow circulation conditions to the phreatic subsystem in the study area can be conceptualised from the potentiometric map (Figure 3.9) (Costa et al., 2007) derived from the static water level. Several hydrogeological behaviours are possible to identify:

i) In general, the flow occurs from southwest to northeast, according to the topographic slope discharging into the sea;

ii) The flow towards leans toward the northern boundary of the study area – this boundary is defined by the Paraiba River, therefore representing the baseflow to such river;

iii) Negative water levels (relative to sea level) were found close to the geological fault – this situation occurs due to the high localised exploitation – it must be highlighted that this region is the recharge area to the confined subsystem.

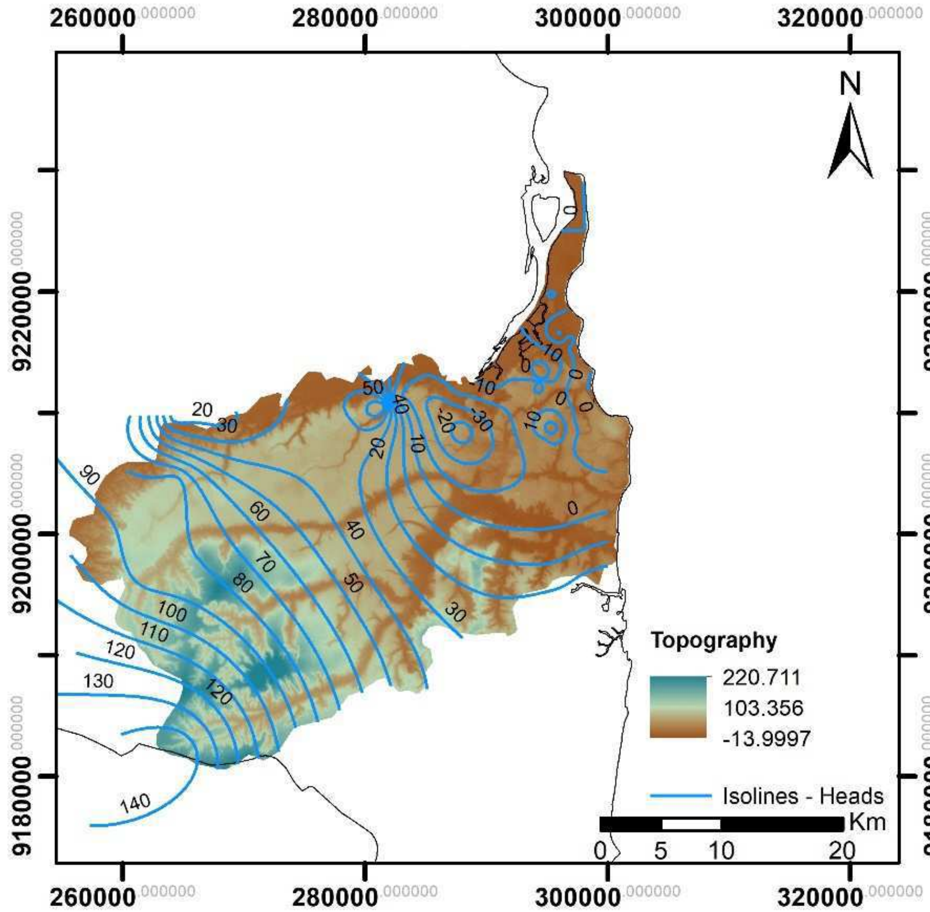


Figure 3.9 - Isolines of the hydraulic head (Costa, 2007)

According to Batista et al. (2011), negative water levels are also present in the confined subsystem. These alterations might be altering the baseflow to the Gramame river, given that the potentiometric lines are perpendicular to the river direction.

Potentiometric head data collected during the Bramar Project allowed an update of the isolines for the study area. Data from 39 wells located in the phreatic subsystem were used for this update. The isolines are presented in Figure 3.10. The isolines presented in Figure 3.9 were built by Costa (2007) using a wider dataset that comprised the whole coast of the Paraiba State, however, within the present study area, there were lower data available. Hence, the update of this isoline (Figure 3.10) are more detailed. Comparing the maps, it was identified the the potentiometric heads in the southwest region has been preserved, as well as the general flow direction towards the Paraíba River. The update in the isolines also improved the representation of the flow along the rivers Gramame, Mumbaba and Mamuaba.

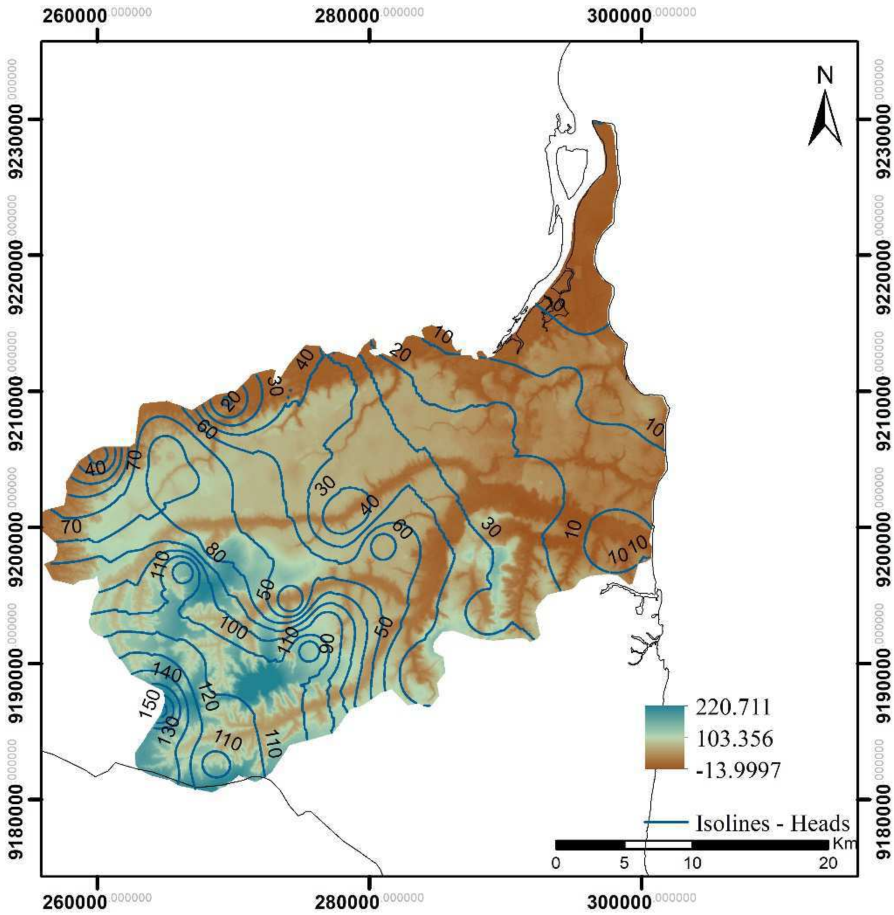


Figure 3.10 - Isolines of hydraulic head

The hydrodynamic parameters of the groundwater system were evaluated by Costa et al. (2007) through pumping test and by Braga et al. (2015) and Batista et al. (2011) through computational modelling. For the phreatic subsystem, the results were obtained in two locations through pumping tests. The first site, located in the southeast region of the study area, had a hydraulic conductivity of 2.62m/d; the second site, located in the municipality of Alhandra, had a hydraulic conductivity of 1.21m/d. Braga et al. (2015) found values between 3.35m/d and 8.13m/d in a region near the Paraiba river and the geological fault using groundwater modelling. To the confined subsystem, the pumping test resulted in values in the range of 0.24m/d to 5.94m/d; through the calibration of a groundwater model, the hydraulic conductivity values varied between 1.5m/d to 13.50m/d (Batista et al., 2011).

## 3.7 Land Use

A map with the existing land cover in the study area is presented in Figure 3.11. This map was obtained as part of the BRAMAR Project, using satellite images from U.S. Geological Survey, ASTRIUM and SPOT 5, and validated in the field with GPS, for the year of 2017 (Fernandes, 2017).

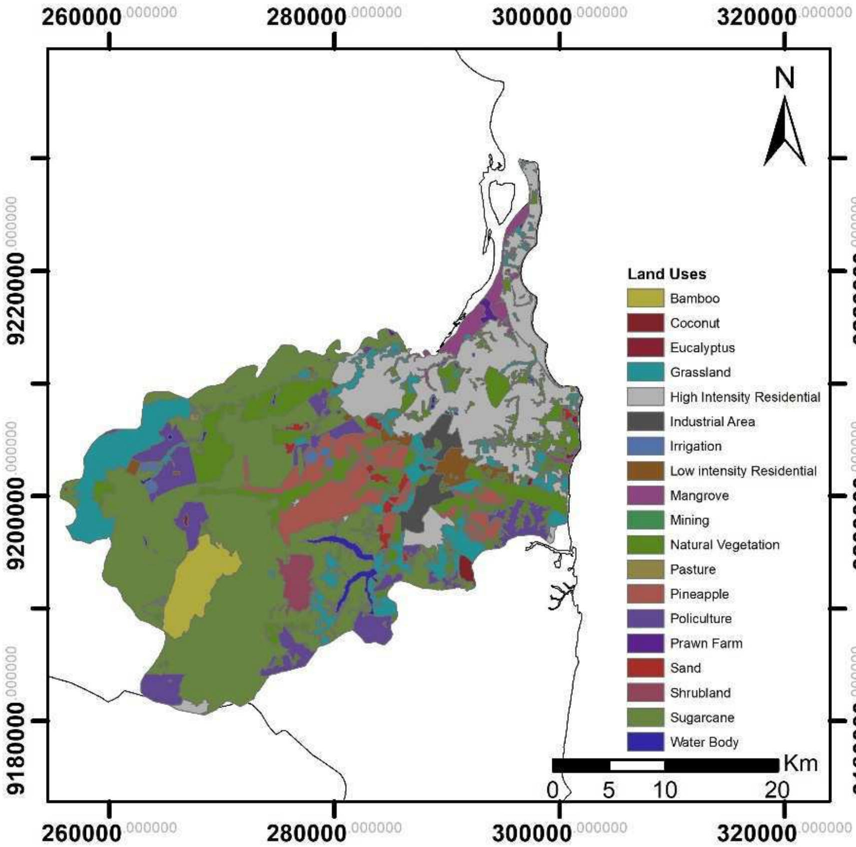


Figure 3.11 - Land Cover (Fernandes, 2017).

Among the different land uses in the region, it is possible to observe the predominance of agricultural, urban and native vegetation. Agriculture is the most common land use embracing almost 50% of the study area. Sugar cane and pineapple are the primary cultures existing. In the urban land-uses are included the residential, commercial and industrial areas. This type of land use represents approximately 18% of the total. The vast majority of the urban land uses are occupied by a high-density urban area, such as, the metropolitan area of João Pessoa, that also includes the conurbation with the municipalities of Cabedelo, Bayeux and Santa Rita.

Water bodies represent approximately 2% of the area. The Gramame-Mamuaba Dam located in the Gramame River Basin is utilized as the water supply for a large part of the João

Pessoa metropolitan region. This dam has a capacity of 56hm$^3$ and represents a large part of the water bodies in the study area with a water mirror that can reach 9.36km$^2$. Besides this reservoir, there is also the Mares dam, Reis dam, and several small dams.

The study area has been through an intense process of changes in land use. This fact is supported by the data obtained from the MapBiomas project. The land use for the year 1987 (Figure 3.12) and the year 2017 (Figure 3.13) are presented. Three significant changes in the study area can be seen along this period of 30 years. First, there has been a considerable advance in the urban area, resulting from the process of urbanisation and economic development of the region. Second, the natural cover (forest and mangrove) spatial distribution changed. There was an increase in natural cover near rivers, however, a decrease in this type of cover elsewhere. A possible explanation is the alterations in the legislation during this period, such as the forest code. Third, the construction of the Gramame-Mumbaba dam. This dam brought many benefits to the region, e.g. increase in water security for the João Pessoa metropolitan region, but also resulted in an inevitable change in the land uses of the area.

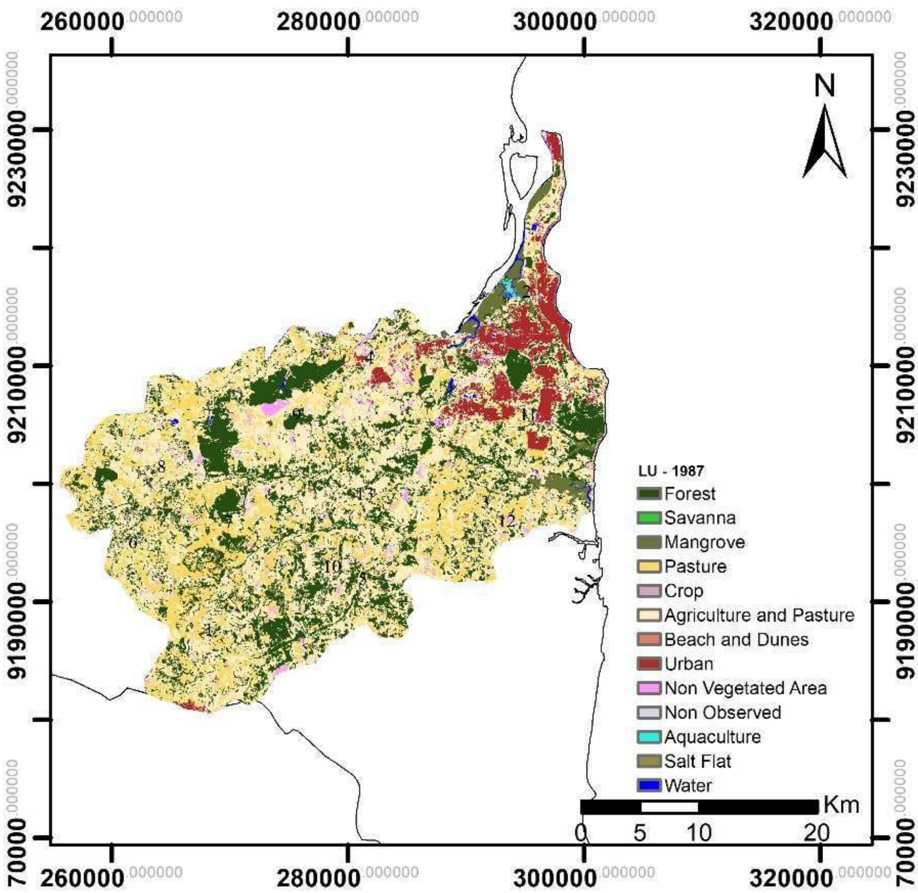


Figure 3.12 - Land Use – 1987 (MapBiomas, 2020)

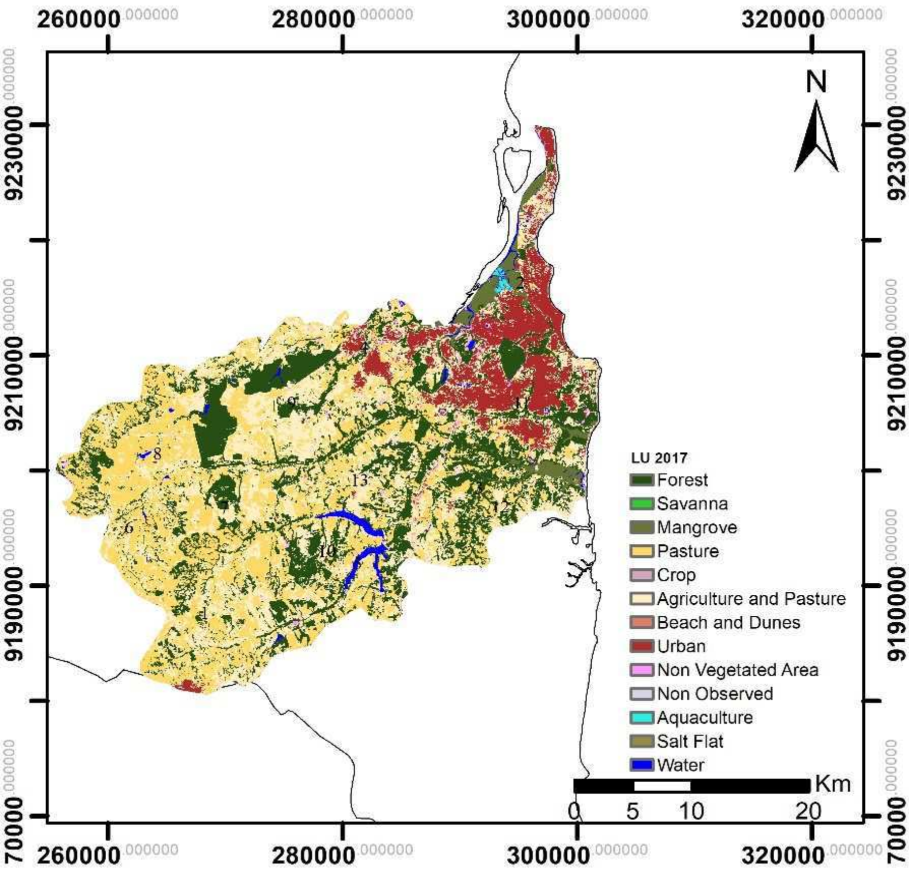


Figure 3.13 - Land use – 2017 (MapBiomas, 2020)

## 3.8 Perspectives for the integration of Land Use and Groundwater Management

The spatial distribution of wells in the study area presents a large concentration at the João Pessoa metropolitan area. This region has presented in the last years an urban sprawl mainly towards two directions. First, in the south direction, towards the municipality of Conde, that is compressing the margins of the Gramame River. Second, in the east direction, increasing the conurbation with the municipalities of Bayeux and Santa Rita, towards the margins of the Paraíba River (Rufino et al., 2009). This growth can provoke several changes in the land use; hence it is necessary to include the planning of land use looking to how it can be applied to manage groundwater.

The land use and land cover policies in Brazil follow the juridical hierarchy, starting at the federal level with the Law 6.766/1979 (Brasil, 1979), that concerns about the division of urban land. This law allows the State, Federal District, and Municipalities to establish complementary norms relative to this subject. In this law, and later alterations, there are present two features that can be applied as instruments to manage groundwater. One is the basic urban infrastructure that includes urban equipment of stormwater drainage, sewage systems, water supply and access ways. Another is the obligatory requirement to public open spaces according to the population density of the area.

In the Law 10.257/2001 (Brasil, 2001), that establishes general guidelines to urban policies, another feature is addressed that can be utilised to manage groundwater. This law concedes to the public administration the right to the pre-emption when it is needed to create open public space, green areas, conservation units, or the protection of areas of interest, such as environmental, historical, cultural or scenic protection area. Beyond that, it creates the requirement to include in the director plan of the municipalities the identification and guidelines to the preservation and occupation of green areas, seeking the increase of permeability of the cities (included by the Law 12.983/2014 - Brasil 2014). Other instruments to the creation of urban green areas are included by the new Forest Code – Law 12.651/2012 (Brasil, 2012).

Analysing the new Forest Code, it is noted the existence of some features that can be applied with a multifunctional aspect. While these features can preserve the forest and native vegetation, they are also crucial for the urbanisation process and to manage groundwater. Among these features, it can be highlighted the delimitation of Permanent Preservation Areas and Natural Landscape Protection Areas. Mainly when this includes buffer zones surrounding river courses, reservoir, springs, and mangroves.

In the city of João Pessoa, the urban code was established with the Law 2.102/75 (João Pessoa, 1975). In this code, it is assured the municipal government the domain of green areas and the warranty of the adequate landscape aspect. This assurance can be enforced through public areas of landscape and areas of preservation of the natural landscape. The Law 2.699/79 (João Pessoa, 1979) established the creation of four zones of special preservation, and the decree 5.285/2005 (João Pessoa, 2005) established an amount of permeable land in the public edification and pathways. Finally, in the last review of 2009, the director plan of the City of João Pessoa included the creation of Environmental Mitigation Sectors – destined to low-density occupation and high permeability; and the Landscape Protection Sectors that prioritise the characteristical view of rural zones and slow down the urban expansion.

Among the current land use and land cover policies, some instruments can be applied to manage groundwater: i) basic urban infrastructure; ii) green areas and public open spaces; and iii) permanent preservation areas and natural landscape protection areas.

Basic urban infrastructure includes the sewage, stormwater drainage, access ways and water supply. Currently, there is an unmanaged recharge from this equipment — the sewage system deserving careful attention due to the negative effects it has on groundwater quality. Drainage systems, along with access ways, should be used to mitigate the impacts of urbanisation on the water cycle. Applying urban sustainable drainage systems (such as seepage trenches or detention basins) and increasing permeability on public roads or parking lots to a level of infiltration close to the natural conditions will help the groundwater recharge process and have other effects beneficial to the urban system. The expansion of the city with its new housing areas need this equipment for its operation; if corrected planned, the expansion of the urban area can bring minimal or even beneficial effects to the recharge of aquifers (Walter et al., 2018).

Urban green areas are defined in the Forest Code as spaces, public or private, with a predominance of vegetation, preferably native, natural or recovered, unavailable for housing construction, intended for recreation, leisure, improvement of urban environmental quality, protection of water resources, and maintenance or improvement of the landscape. There is an intersection between urban green areas and free spaces for public use, as the latter can be understood as gardens, parks, courtyards, squares and access ways. As the access ways are already considered as part of the urban infrastructure, we can recognise the set of urban green areas and public free spaces simply as green spaces. The use of decentralised urban elements, such as parks or floodplains, or sustainable drainage devices, such as wells or infiltration ditches, in the green spaces, should be considered seeking to increase the permeability serving as a recharge area due to the possibility of its use for the accumulation and/or infiltration of precipitation. It is important to note that, as identified by Rufino et al. (2009), some locations in the study area have high groundwater levels and low topographical altitudes, such as the Cabedelo region and some of the João Pessoa neighbourhoods. As a result, frequent flooding happens in these regions. In such places, other instruments could be applied, taking into account their hydrogeological characteristics.

On the one hand, the delimitation of permanent preservation areas along the water bodies favours the reduction of intensity and duration of the flow, consequently allowing a more significant infiltration. As a result, there is a better balance between the hydraulic gradient of the aquifers and the rivers. A rapid flow situation is avoided, which could result in higher demand for base flow after it occurs. On the other hand, natural landscape protection areas favour the establishment of regions with high permeability, thus reducing the impact on the natural condition of the hydrological cycle. Permanent preservation areas and natural landscape preservation areas are essential in the urban and, especially, rural context (Walter et al., 2018).

## 3.9 Socioeconomics

As mentioned above, nine municipalities are partial or completely embraced in the study area. Among them, are included the state capital João Pessoa and several surroundings municipalities (Alhandra, Bayeux, Cabedelo, Conde, Cruz do Espírito Santo, Pedras de Fogo, Santa Rita e São Miguel de Taipu). The estimated population of these municipalities for the year 2019, according to the Brazilian Institute of Geography and Statistics (IBGE, 2019), is 1,197,828 inhabitants.

The chart in Figure 3.14 depicts the population growth according to the census conducted until the year of 2010 (IBGE, 2010). The population has constantly been growing, with almost 95% of the inhabitants situated in urban areas. This behaviour follows a trend expected in large economic centres. Even though the rate of growth has reduced in the last census data, there is still a considerable increase in the population rate. The census of 2010 pointed to a total population of 1,090,877 inhabitants. Comparing this data with the estimates from the year 2019, the increase in population is more than 9%.

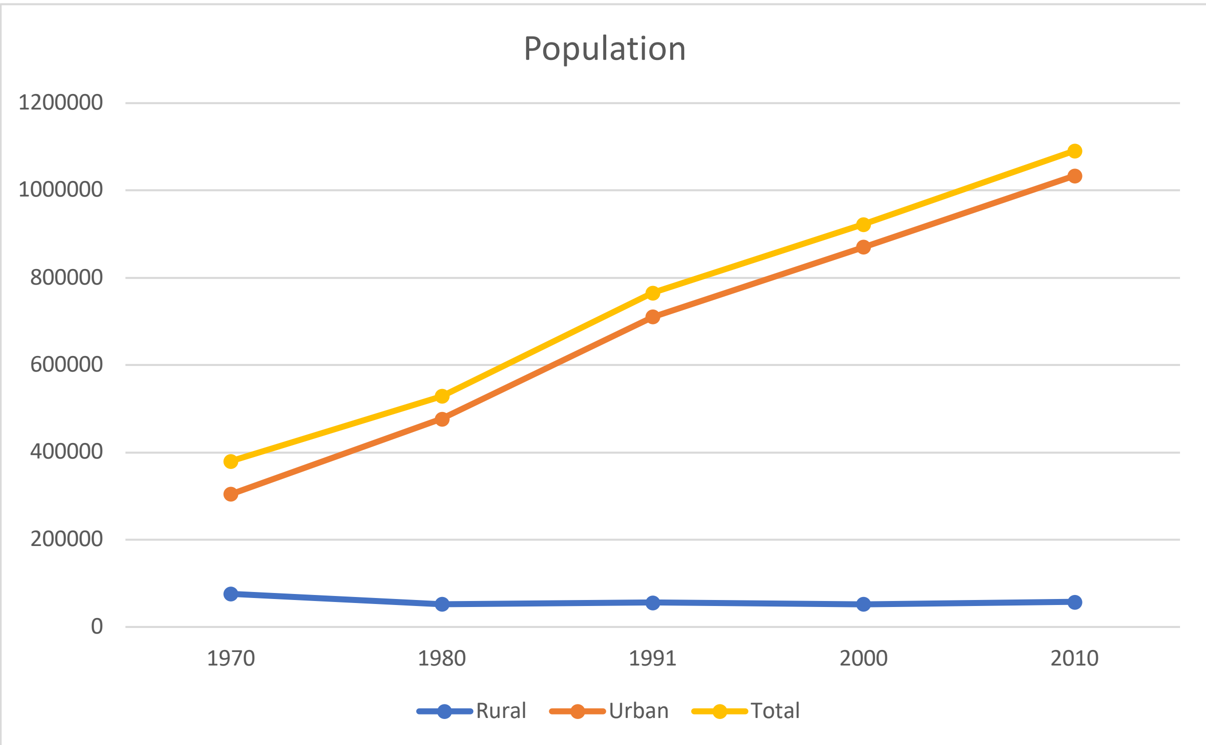


Figure 3.14 - Population

The low amount of population in the rural zone does not correspond to a reduction in agriculture. Farms in the region are concentrated in increasing production. The area of temporary crops on the coast of the Paraíba state has been kept steady since the year 2005. Besides, the area with permanent crops has increased in the past four years (MapBiomas, 2019). Both growths, in population and agriculture, results in an increasing demand for water resources.

## 3.10 Water Demand

The state capital – João Pessoa – and several other municipalities are situated in the study area, these cities compose the central metropolitan region of the state, with more than 1 million inhabitants. As expected, the population is concentrated in urban areas (94.75%) (IBGE, 2019). The water demand in the study area is destined for domestic, commercial, industrial, pasture and agricultural consumption.

According to data from the Water Resources Management Plan for the Gramame River Basin (PDRH-Gramame), the total water demand for urban supply for the year 2000 was 86.79hm$^3$/year and forecasted demand of 106.40hm$^3$/year (SEMARH, 2000). Such value presents a good agreement with the estimates from the BRAMAR Project (BRAMAR, 2017), 98.23hm$^3$/year. The estimates for the year 2020 are 124.09 hm$^3$/year. Close values are present in the Water Resources State Plan (PERH) where the demand for the year 2008 would be of 95.69 hm$^3$/year, and it is expected that the water demand for domestic supply, both urban and rural, reaches the value of 115.20hm$^3$/year in the planning horizon of 2023 (AESA, 2006). These values do not include industrial water demand, which for the year of 2000 and 2010 would be of 7.07hm$^3$/year and 17.54hm$^3$/year, respectively, with a projected demand of 28.01hm$^3$/year for the year 2020, according to the PDRH-Gramame.

The values for agricultural water demand obtained from the PDRH-Gramame and the PERH present considerable disagreement between them. In the former, for the year 2000, it was forecasted a water demand of 29.21 hm$^3$/year (SEMARH, 2000), while for the latter, in the year 2003, had an estimate of 107.79 hm$^3$/year (AESA, 2006). However, the methodology applied to estimate the Gramame River Basin demand for agricultural uses are more reliable, given that, in this project, a user registration was conducted while also identified the irrigation periods. Furthermore, the study area chosen includes a reasonable part of the Lower Course of the Paraiba River Basin. Due to such difficulties, the water demand for irrigation was calculated by the BRAMAR Project using the land use present in the study area, meteorological data from 2000 to 2014 and disregarding possible losses. The value calculated was 227.70 $hm^3$/year. If inefficiencies from the irrigation system and waste are considered, the value increases to 379.60 $hm^3$/year (BRAMAR, 2017).

Finally, when comparing the water demand values for pasture between the PDRH-Gramame and the PERH, considerable disagreement between them is, again, observed. Here is worth note that different methodologies were applied to calculate such values. The BRAMAR Project also calculated an estimate for pasture water demand. The results obtained was in line with the one presented by the PERH. Therefore, the water demand for the sector can be estimated at 10.8 $hm^3$/year (BRAMAR, 2017). This value does not include future estimates, given that it was assumed that such estimates would not be representative of possible fluctuation in the cattle due to climate variations.

In summary, the water demand for the study area to the year 2010 had values close to 514.34 $hm^3$/year, with a forecast to the year 2020 of 542.50 $hm^3$/year, according to the water plan currently in force. The values estimated during the BRAMAR project were close related: a mean annual value of 498.4$hm^3$ – 390.40$hm^3$ for agricultural and livestock uses, 106.8$hm^3$ for domestic uses, and 1.2$hm^3$ for industrial uses.

## 3.11 Groundwater management: Institutional framework

The management of water resources in Brazil follows the National Water Resources Policy (NWRP) established by the Federal Law 9.433/97 (Brasil, 1997). This law created the National Water Resources Management System (NWRMS) and regulated the item XIX of article 21 of the Brazilian Federal Constitution (Brasil, 1988). This law also determined the competence for the management of national (federal) and state water.

The NWRP has in its foundation six principles: i) water is a public good; ii) water is a limited natural resource, having an economic value; iii) in scarcity situations, the priority of use must be directed to human consumption and animal watering; iv) the management of water resources must provide multiple uses; v) the river basin is the territorial unit for the implementation of NWRP and the NWRMS; and vi) the management of water resources must be decentralised and participatory.

As objectives, the NWRP presents four points: i) to ensure the current and future generations the necessary availability of water resources with quality compatible to the intended use; ii) the rational and integrated use of water resources seeking the sustainable development; iii) the prevention and protection to critical hydrological events resulting from natural or inadequate use of natural resources; and iv) the incentive and promotion of the harvesting and reuse of stormwater.

Sustained by these foundations and objectives, five instruments are defined in the NWRP to be implemented by the NWRMS: i) water resources plans; ii) Classification of water bodies according to their uses; iii) water rights for the use; iv) raw water charges; and v) the Water Resources Information System.

The National Water Resources Management System has the objective to coordinate the integrated management of water resources, implement the NWRP, arbitrate conflicts related to water resources, promote the raw water charge, and plan, regulate, and control the use, preservation and restoration of water resources. In this system, two spheres of actions are established: one at the national level and another at the state level. River basins that are contained in more than one state are managed at the national level; river basins that are fully contained in one state and the groundwater resources that occurs at state territory are managed at the state level.

Two types of entities compose the NWRMS. One type is responsible for the formulation of policies, and the second type is responsible for the implementation of policies. Among those responsible for policy formulation, the National Water Resources Council (NWRC) and the State Water Resources Council act as normative and deliberative entities for groundwater management. Regarding the implementation of policies for groundwater management, the National Water Agency and the State Water Agency are the responsible ones.

The NWRC has been aiding the management of groundwater resources at the national level since 2001, with the resolution N°15/2001 (CNRH, 2001a), that established general guidelines for groundwater management under the Law 9.433/97, seeking the rational use, the application of the NRWP instruments and the integration between surface and groundwater resources. There are resolutions specific for regulating instruments of the NWRP, such as the CNRH N°48/2005 (CNRH, 2005) that regards the charging for water use, and the CNRH N°16/2001 (CNRH, 2001b), that regulates the principal for concession of water rights. Other resolution concerned about the classification of groundwater according to the quality - N91/2008 (CNRH, 2008a), criteria for preservation and conservation of groundwater -

N°92/2008 (CNRH, 2008b), guidelines for the implementation and operation of a national monitoring network of groundwater - N°107/2010 (CNRH, 2010), and more recently, guidelines for the conjunctive management of surface and groundwater resources, more specifically phreatic aquifers, taking into account the coordination between federal and state governments - N°202/2018 (CNRH, 2018).

In the case of the state of Paraíba, even though the State Water Resources Policy (SWRP) precedes the NWRP, in general, these policies are aligned in their principle and guidelines (Araújo et al., 2012). However, some instruments of the NWRP for groundwater management must be defined at the state level. An example of this refers to the water rights for groundwater. Each state regulates this instrument; therefore, the criteria depends on the local. In the Paraíba state, where the study area is located, the water rights are regulated by the state decree 19.260/1997 (Paraíba, 1997).

The state decree 19.260/1997 guides groundwater management under three aspects. First, the necessity of water rights: according to the decree, it is only required the water right for exploitation of groundwater if the amount exploited exceeds the value of 2000 litres per hour. Second, the impossibility of concession of water rights: the decree prohibits the concession of water rights if the intention is the deposition of pollutants on groundwater bodies. Finally, the criteria for the concession of water rights: this decree establishes as criteria the capacity of recharge of the aquifer and the well's' yield obtained from the production test. In practice, due to the low availability of data, only the latter criterion has been applied for the concession of water rights.

## 3.12 Groundwater management: Issues and alternatives

The current Brazilian institutional framework for both surface water and groundwater management encompasses several aspects necessary to good governance and integrated water resources management. However, many issues and challenges remain. While some of them are fundamental challenges from the management of hidden resources, such as groundwater, others exist due to flaws in the institutional arrangement.

Braga et al. (2015) analysed the criteria for the concession of water rights established by the Paraíba state decree 19260/1997 (Paraíba, 1997) for groundwater. The criteria were: capacity of recharge of the aquifer and the wells' yield obtained from the production test. These criteria appear to be ineffective to sustainably guide the concession of water rights, especially if not applied together. According to numerical modelling conducted by Braga et al. (2015), other aspects, such as the seasonal variation in the recharge, need to be taken into account to achieve sustainable value for a water right.

Other limitations from the criteria established by the state decree are its restriction to the quantitative aspect, the disregard to the hydrogeological behaviour of the system and their spatial constraint to the well itself. Building on these limitations, the ASUB project (ASUB, 2010; Costa, 2009) proposed a new framework for the concession of water rights for groundwater.

A significant advance proposed by the ASUB Project (ASUB, 2010; Costa, 2009) was the conceptualisation of management coverage levels. Three levels were defined: local, regional and basin. The basin level refers to the whole hydrologic basin using the hydrologic cycle as the source of information. In this level, the aquifer's limits of withdrawal and the ecological demand are taken into account to guide the definition of criteria. The regional level is intermediate: to be applied, management zones should be defined specifically for each basin. Aspects such as the recharge, discharge, and hydrogeological behaviour are applied to define the zones and validate the criteria. Finally, the local level refers to the scale of the well to be granted the water right, taking into account its interference in nearby systems and the consequences of its exploitation.

The ASUB Project (ASUB, 2010; Costa, 2009) also proposed new criteria to be applied for the concession of groundwater rights according to each coverage level:

At the basin level, two criteria were proposed. The first is the priority of use of surface water, given that groundwater should be a strategic resource destined for the maintenance of water security. The second is the availability of the groundwater potential after meeting the ecological demands of the river and other water bodies.

At the regional level, three criteria were proposed: i) priority of water use; ii) risk of saline intrusion; and iii) vulnerability of aquifers. First, the priority of water use is considered by the Law 9433/97 (Brasil, 1997), but only in scarcity situations, while the resolution N°16/2001 (CNRH, 2001b) from the NWRC establishes that the priority of use for the concession of water rights should be defined in the Water Resources Plan (an instrument of the Law 9433/97). However, this guidance of the resolution was not yet incorporated by the state decree. Second, the risk of saline intrusion is a constant in coastal aquifers. Extensive pumping can result in an inversion of the groundwater flow, bringing saltwater to the continental portion of the aquifer. Hence, a safe distance from the shoreline should be kept when drilling wells for exploitation. Third, the vulnerability of the aquifers refers to the sensitivity of the aquifer to contamination caused by natural or human-made impacts. Therefore, water rights should take into account the possibility of contamination of the groundwater bodies.

Finally, at the local level, four criteria for water rights were proposed. The first criterion regards groundwater quality. In the concession of water rights, the quality of the groundwater should be evaluated in the light of the existing regulations. Currently, this is not a criterion in the state decree for granting water rights. Second, the interference between wells. The well-pumping results in the occurrence of a radius of influence, meaning a certain radius where the potentiometric head is reduced as a consequence. When two or more wells are closely drilled, the radius of influences of these wells might interfere with each other; as a result, there can be a reduction in the well's yield and excessive drawdowns in the potentiometric surface. Third, the maximum drawdown. The reduction of water level through time is a sign that the withdrawal is not being replenished. When the exploitation happens without taking into account the drawdown, this problem might be exacerbated, leading to the necessity of drilling deeper wells and more energy consumption. The determination in the water right of a maximum depth of exploitation can avoid a constant withdrawal over the aquifer's productivity capacity. Fourth, water demand. As stated in the Law 9433/97 (Brasil, 1997), one of the objectives of the NWRP is the rational and integrated use of water resources seeking sustainable development. Therefore, the concession of water rights for groundwater could require, as a counterpart, the adoption of measures that would reduce the consumption of groundwater. These measures could include the adoption of equipment or mechanism designed to save water as well as the installation of water meters to verify the changes in consumption (ASUB, 2010; Costa, 2009).

Braga et al. (2017) assessed the sustainability of eight of the criteria proposed by the ASUB Project (ASUB, 2010; Costa, 2009) taking into account three aspects: economic impacts through the implementation/maintenance cost, environmental impacts, and social impacts using as a proxy the potential of second-order conflicts. This latter aspect refers to the scarcity of social resources and tools to deal with the societal consequences of the application of a management measure. The criteria assessed were: priority of use of surface water; availability of the groundwater potential; priority of water use; risk of saline intrusion; vulnerability of aquifers; groundwater quality; interference between wells; and water demand management. Braga et al. (2017) showed that the criteria at a basin level had the highest cost and the highest potential for second-order conflicts due to their high spatial reach and restrictive characteristics. Among the eight assessed criteria, the priority of use of surface water appears to be the least applicable to the study area, given that the surface water system is already stressed. The criteria of the vulnerability of the aquifer and demand management had the best potential for implementation. The former due to its conservational aspect and the latter to address the issue of groundwater management under the perspective of the demand, seeking rational use, and not through increasing the offer.

## 3.13 Conclusions

The study area selected for this thesis represents a complex groundwater system in a tropical coastal region exhibiting diverse anthropogenic pressures such as economic and populational growth, increasing water demands and rapid changes in the land use. This situation is not exclusive to this area and represents a typical setting found in many areas throughout the world.

This chapter also highlighted the possible paths to connect land use planning and groundwater management underpinned by the current legislation of the local institutional setting through the instruments of public policies. Basic urban infrastructure can be applied to provide a managed groundwater recharge, while green areas, public open spaces, permanent preservation areas and natural landscape protection areas can help to bring the water cycle of this modified system to more pristine conditions.

Afterwards, the current institutional framework for groundwater management was presented and analysed. The existing literature on the topic has shown that many aspects are not being taken into account by the current legislation. As a result, indications of groundwater exploitation have been detected throughout the years. However, alternatives had also been presented. A marked example is the subdivision of the groundwater system in different management levels with specific criteria for each level that considers both qualitative and quantitative status. These criteria have been assessed regarding their sustainability.

Sustainable groundwater management needs as more information as possible to support the application of its measures and instruments. An analysis of the current situation and institutional constraint is the first step. Furthermore, it is required the application of tools that can generate information about the groundwater system from a spatial-temporal perspective. One of such tools is the groundwater modelling. The preparation of a groundwater model as a tool to provide information about the groundwater system is presented in the next chapter.

# Chapter Four: Conceptual and Numerical Modelling of the Groundwater System

## 4.1 Introduction

Groundwater modelling seeks to represent the behaviour of a determined groundwater system with the objective to synthesise information and conceptualise hydrogeological processes through a quantitative framework (Anderson et al., 2015b). Therefore, groundwater modelling is defended as an initial step for any hydrogeological studies analysing nontrivial questions (Bredehoeft and Konikow, 1993). For the application of a groundwater model, a schematic representation of the system behaviour must be initially developed. This process is known as the conceptual model (Gondwe et al., 2011), and it is based on the current knowledge of the system. Subsequently, this conceptual model is translated into the numerical model through the groundwater flow equations and boundary conditions (Anderson et al., 2015b).

Due to the high nonlinearity of groundwater flow equations and, many times, the complex settings of the groundwater systems it is unpracticable to model a groundwater system without the application of numerical tools. The numerical models started to be more applied with the release of the USGS 3D Difference Model, USGS Modflow and the Finite Element Subsurface FLOW System – FEFLOW. The development of computation capacity led to the increasing ability of such models to simulate and process the information in even more complex settings, becoming the primary tool for analysis and management of groundwater resources.

The numerical models are applied for both quantitative and qualitative studies regarding groundwater resources. Among the many available computation codes, the FEFLOW has been applied. FEFLOW is a state-of-the-art computational code that uses the finite element method to solve the groundwater flow equations. This code allows the reconstruction of complex hydrogeological settings and features, and has been widely applied to groundwater simulations (Blackport and Dorfman, 2014; Lin and Lin, 2019; Viaroli et al., 2019).

Examples of applications of this tool are present throughout the literature. Stein et al. (2019) used the FEFLOW to analyse the effect of pumping saline groundwater in a phreatic coastal groundwater system on the dynamics of the fresh-saline water interface, comparing and validating the results with field-testing. Du et al. (2016) applied the FEFLOW to verified and diagnose how diverse hydraulic structures can affect an alluvial aquifer and constrain the groundwater management measures. Sklorz et al. (2017) applied the FEFLOW to analyse the artificial recharge from irrigation, the existing drawdowns due to overexploitation and its consequences. Niazi et al. (2017) estimated the spatial variability of the recharge through the combination of analysis using baseflow information and chloride mass balance method, verifying the results with the application of FEFLOW. This computational code has also been combined with a Geographic Information System (GIS) by Li et al. (2017) to analyse the impact of water demand measures in agriculture in different scenarios and taking into account the water balance in the vadose zone and the regional groundwater flow. Furthermore, Xanke et al. (2017) simulated the managed aquifer recharge with rainwater harvesting in a karst aquifer and the potentiometric fluctuations due to the pumping in regions near a recharge area also using the FEFLOW. Therefore, the FEFLOW is well established as a modelling tool to aid groundwater management.

However, this tool needs to undergo a calibration process before it can be applied. This process is time-consuming and requires a substantial amount of data (Anderson et al., 2015b). Such data is often a constraint in groundwater studies, especially in developing countries (Klaas et al., 2017). Due to its hidden aspect, the groundwater system often lacks all the data needed to calibrate the model with reduced uncertainty regarding the parameters. As a consequence, the final model tends to be oversimplified, characterised by less parameter, and have its spatial-temporal discretisation compromised (Klaas et al., 2017).

This chapter addresses the calibration and parametrisation of a numerical groundwater model built using the software FEFLOW (Diersch, 2014) for the groundwater system of the cultural landscape presented in the previous chapter. This study area was also included in the BRAMAR Project - Strategies and Technologies for Water Scarcity Mitigation in Northeast of Brazil. The BRAMAR project was a binational partnership between the Brazilian and the German Government, including diverse universities and management agencies. During this project, an extensive network of research and monitoring was established dedicated to the collection of climatic, physical, socioeconomic, hydrological and hydrogeological data. The present research contributed to the project providing the calibrated hydrogeological model to the study area as well as the simulations for water budget and new insight relative to management strategies that could be applied in the study area.

Throughout this chapter, both conceptual and numerical models for the study area is explained. Furthermore, the parameters' values obtained after the calibration are analysed taking into account existing geological materials and by using statistical indexes for the evaluation. Finally, the parametrisation of the model achieved is presented.

## 4.2 Materials and Methods

### 4.2.1 Data

In total, 132 lithological profiles and two geological cross-sections available were analysed with the aid of Geographical Information Systems (GIS) to identify the geometric characteristics of the hydrogeological formations in the study area. This information was obtained though the analysis the existing database, such as SIAGAS, AESA and Costa (2007). Topography was obtained from SRTM images with a spatial resolution of 30x30m to construct a digital elevation model – DEM of the area. The altitude varies between 220 and -5 above sea level. Through the DEM, it was also possible to delimitate the main river courses.

Groundwater level from the phreatic aquifer was measured monthly from February 2016 to March 2018. The depth from the surface to the water level ranged from 2 meters to 31 meters in the driest month. Of the 39 wells measured, 14 had a water level depth less than 5 meters. In these wells, losses due to evapotranspiration are expected. The water level from 13 wells in the confined aquifer was measured between August 2016 and March 2018. Out of 13 wells measured, seven presented negative values of potentiometric heads.

At total, it was used 52 groundwater level time-series data measured at wells without any exploitation. Of these, 39 wells were located in the phreatic aquifer, and 13 were in the confined aquifer. Eight wells were measured using automatic divers with different time-step, and the remaining were measured manually in monthly measurement campaigns. These data were collected between February of 2016 and February of 2018.

### 4.2.2 Groundwater Modelling

The software Finite Element Subsurface Flow System – FEFLOW was applied to construct the numerical model. Through FEFLOW, a hydrogeological model of the study area was built, and simulations were conducted. The FEFLOW simulates the groundwater flow solving the general flow equation (Eq. 4.1), according to the defined boundary and initial conditions. This equation mathematically represents the spatial-temporal variation of the hydraulic head according to the hydraulic parameters of the aquifer (hydraulic conductivity and specific storage).

$$\frac{\partial}{\partial x}\left(K_x \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_y \frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_z \frac{\partial h}{\partial z}\right) = \left(S_s \frac{\partial h}{\partial t}\right) + W \qquad \text{Eq. 4.1}$$

Where,

$K$x, $K$y, $K$z are the hydraulic conductivity in each direction,
$\partial h/\partial x$, $\partial h/\partial y$, $\partial h/\partial z$ are the components of the hydraulic gradient;
SS is the specific storage; and,
W are the variations in storage due to inflow or outflow.

### 4.2.3 Calibration

The calibration process seeks to determine the equivalent values for hydraulic parameters of the groundwater system by adjusting these values and comparing the calculated hydraulic head with the observed one. This is an iterative process, consisting of sequential steps. First, a variable was determined to calculate the objective-function for the calibration. The objective function can be defined as the sum of the weighted square residues (the difference between the calculated head and observed head). This variable is utilised to compare the calculated and observed data. In this case, the variable chosen was the hydraulic head. Afterwards, the hydraulic parameters were adjusted, and a new simulation was run to compare the resulting value of the objective function with the previous simulation. The purpose of this process is to optimise the resulting objective function to the smallest value.

For the calibration of the model to the steady-state, it was applied the tool Finite Element Parameter Estimation – FePEST. This tool is an adaptation of the well-known PEST for the finite element method; it automates the process of adjusting the parameters for calibration, substantially reducing the time necessary in the calibration step.

Statistical indexes are used to evaluate the calibration process (Anderson et al., 2015b). Among the commonly applied are the Mean Error (ME), Mean Absolute Error (MAE), Root Mean Square Error (RMSE), Normalised Root Mean Square Error (NRMSE), and Determination Coefficient:

$$ME = \frac{1}{n}\sum_{i=1}^{n}(h_o - h_s)_i \qquad \text{Eq. 4.2}$$

$$MAE = \frac{1}{n}\sum_{i=1}^{n}(h_o - h_s)_i \quad \text{Eq. 4.3}$$

$$RMSE = \left[\frac{1}{n}\sum_{i=1}^{n}(h_o - h_s)_i^2\right]^{0.5} \quad \text{Eq. 4.4}$$

$$NRMSE = \frac{\left[\frac{1}{n}\sum_{i=1}^{n}(h_o - h_s)_i^2\right]^{0.5}}{h_o^{max} - h_o^{min}} \quad \text{Eq. 4.5}$$

Where, $h_o$ is the observed hydraulic head, $h_s$ is the simulated head, $h_o^{max}$ is the maximum observed head and $h_o^{min}$ is the minimum observed head.

## 4.3 Conceptual model

Through an extensive analysis of the geological, morphological, hydrological, and hydrogeological available data, the conceptual model for the study area was built. As a result, a narrative description of the groundwater system behaviour was inferred. This description includes the directions of groundwater flow, possible recharge and discharge areas, boundary conditions, and connection with surface water systems.

The conceptual model for the study area comprises two different aquifer formations (one phreatic and other confined) underneath two different catchments. In these catchments, there are three main rivers Paraíba, Gramame, Mumbaba, and Mamuaba. The first acts as one of the limits for the delimitation of the study area. In the Gramame, Mumbaba and Mamuaba rivers, there are two major reservoirs connected, and it is the primary water supply source for the city of João Pessoa. In the study area, there is another reservoir – Marés – in the Paraíba River Basin. In this way, there are several exchanges of flow among the rivers, the reservoir and the groundwater system. Therefore, the water movement in the study area has been altered from predevelopment conditions.

As previously mentioned, the system has three different hydrogeological formations. The topographical surface is considered the top boundary of the groundwater system. The superficial hydrogeological formation is the Barreiras formation. According to the data gathered, the Barreiras formation in the study area has an average thickness of 28m. The lithological profiles indicate a thickness varying between 3m and 174m. This wide range happens because this formation has a gradual increase in its thickness from the West-East direction towards the sea (Costa et al., 2007). This increase occurs due to the geological process of sedimentation in the aquifer. The second hydrogeological formation is the Gramame formation. This formation is not capable of storing nor of allowing the flow of water, acting as a confining layer. The profiles and geological cross-sections indicate that in the study area this formation as a thickness of 45m and it only exist at West of the geological fault. The last formation is Beberibe formation. This formation has an average thickness in the study area of 260m. In the confined region, the total depth of the formation can reach more than 400m below sea level (b.s.l.) close to the shore, whereas near the geological fault (that defines the beginning of the confined system) the total depth is 200m (b.s.l.). A schematic representation of the conceptual model is presented in Figure 4.1.

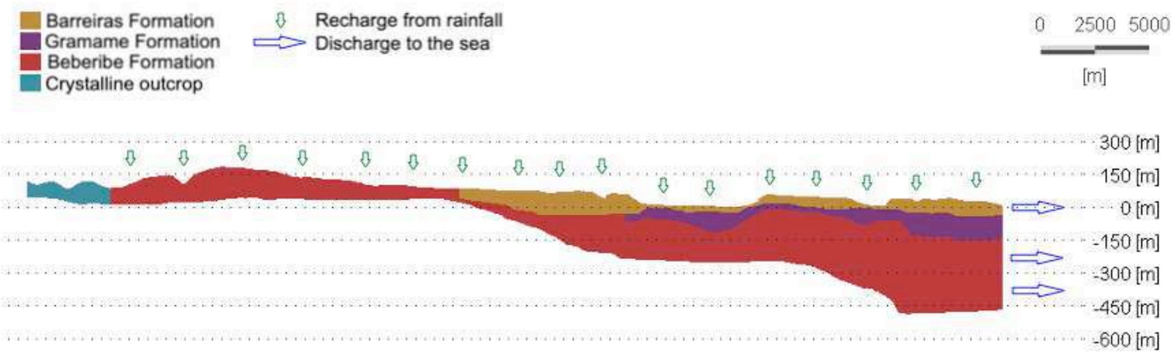


Figure 4.1 - Schematic representation of the conceptual model

Based in the lithological profiles, other studies conducted in the study area (e.g. Costa 2009; ASUB 2010), and from interviews with experts, it was possible to improve the knowledge regarding the initial geological map. The Barreiras formation was initially expected to comprise the whole study area as the first existing formation. However, after 10km from the geological fault, sandy-clay, marine-fluvial sediments and alluvial deposits were not found. Instead of, the presence of diverse sandstones indicates that what has been previously considered as Barreiras formation might be the outcrop from the Beberibe formation.

The Paraíba river delimits the northern boundary of the groundwater system. This river is situated on top of an alluvial formation overlying the crystalline. The elevated level of the crystalline in this area is due to a geological fault. There is no subterranean connection between the Barreiras hydrogeological formation in the study area with the continuity of the Paraíba-Pernambuco sedimentary basin, at the north of this region. Therefore, the Paraíba river basin acts as a fixed head boundary condition, given that the water level in the river can be approximated to the water level in the aquifer. The sea delimitates the eastern boundary of the groundwater system. At the Barreiras formation, the sea level can be considered as a boundary;

therefore, a fixed head condition. The Beberibe formation has also a fixed head condition equal to the sea level when this formation contacts the sea. However, this situation only occurs at 50km from the coast. The head in this formation at the shoreline can be determined using an approximation similar to the one applied by Batista et al. (2011). The southern boundary of the study area is the Gramame river basin limit. The Gramame river basin delimits the western border of the groundwater system. However, in this border, several crystalline outcrops define the beginning of the Paraíba-Pernambuco sedimentary basin in the study area; hence there is no flow entering or leaving the groundwater system in this boundary.

The inflows to the groundwater system occur mainly from the precipitation, but also form the rivers or reservoirs. The precipitation is concentrated from March to August, when most of the annual rainfall happens. During April and July happens most of the recharge to the aquifer. This diffuse recharge is spread throughout the area of the aquifer and heavily influenced by the existing land use. During the dry season, some seepage happens from the rivers and the reservoirs to the aquifer. This is expected to occur more consistently downstream to the Gramame-Mamuaba dam, where the releases from the dam contribute to the perennial aspect of the river. However, the recharge from these sources is smaller than the rainfall recharge (Costa et al., 2007).

The outflows from the groundwater system consist of stream base flow, submarine discharge, evapotranspiration and withdrawals. The stream base flow sustains the main rivers and reservoirs during the dry season. Discharge to sea occurs from both formations (Barreiras and Beberibe). The discharge from the Barreiras formation is at the shoreline. The discharge from the Beberibe happens at 50km distance from the coast into the sea, where the Gramame formation ceases to exist. Some regions of the study area, such as near rivers and reservoirs, have a shallow water level and evapotranspiration contributes as an outflow. Finally, withdrawals to agriculture, industrial or urban water supply are one of the primary sources of outflow. On top of that, it is essential to note that a considerable number of clandestine and unregulated wells exist.

The flow in the groundwater system occurs, in general, from the regions with high altitude to low altitude towards the sea in West-East direction. However, based on groundwater flow, two specific behaviours can be identified. First, between the Gramame and Mumbaba, and Mumbaba and Mamuaba rivers there is a topographic limit with high altitude, in these regions, a considerable of amount of the recharge is discharged as baseflow to these rivers. Second, rainfall recharges the phreatic aquifer and part of the amount recharged goes to the confined aquifer to be discharged at the east system’s boundary; however, there has been considerable exploitation near the recharge area of the confined aquifer; hence the amount recharged to the confined aquifer could have been reduced. Furthermore, this might be even changing the flow direction at the beginning of the confined aquifer.

As mentioned before, the recharge area for the confined aquifer is located at the geological fault. This fault is located at an approximated distance of 12 km from the coast with positive values of altitude (a.s.l). Hence, the groundwater level at this region has positive values relative to the sea level. As the confined formation starts to occur, and its recharged, in this region, these groundwater levels become the potentiometric heads at the beginning of the confined aquifer. Therefore, in pristine conditions, the potentiometric heads for the confined aquifer should also be positive. However, part of the measured values of the potentiometric head was negative. This situation indicates that the confined aquifer has not been recharged at the same level as in the pristine conditions and that the withdrawal has been consistently higher than the amount recharged, resulting in overexploitation.

## 4.4 Numerical Model

Spatial discretisation of the study area is necessary to build the groundwater model in the FEFLOW. This discretisation represents the broken down of the groundwater system into triangular elements and diverse layers. Elements are three-dimensional subdivision; layers are a vertical subdivision of the system; the face of each layer is denominated slice.

Initially, an iterative process was conducted seeking to determine the most feasible spatial resolution of the groundwater model, taking into account the resolution spatial-temporal resolution of the data available and the computational processing required. The implementation of the hydrogeological spatial model resulted in four layers and more than 200,000 elements. It was required four layers because a structured mesh was implemented. Certain formations, such as the Gramame, does not cover all the study area, to implement this, it would be necessary the use of an unstructured mesh. However, the data available limited the construction to a structured mesh, meaning that each layer had to cover all study area. To correctly represent the geometry of the system, part of the elements was configured as inactive; therefore, these elements do not allow any inflow/outflow nor is considered in the model calculation. The representation of active and inactive elements for the numerical model is shown in Figure 4.2 for the slices 2 (a), 3(b), and 4 (c).

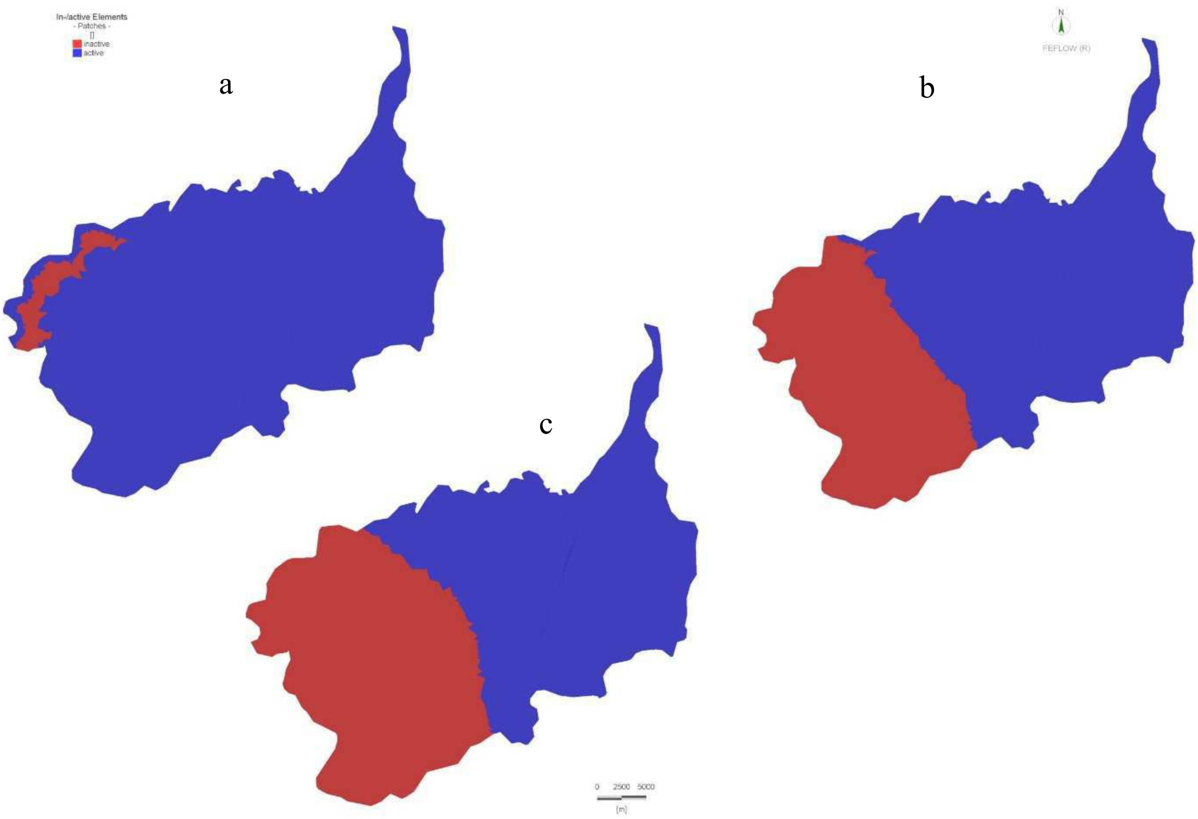


Figure 4.2 - Active (blue) and inactive (red) elements of the numerical model for the slices 2 (a), 3 (b), and 4 (c)

The first layer is limited on the top by the topography of the terrain and has a fixed thickness of 2m, delimiting the bottom. The final implementation of topography in the model and the location of the wells in the phreatic aquifer is presented in **Error! Reference source not found.**. This layer was added to overcome a limitation of the FEFLOW regarding the computation of evapotranspiration in the saturated zone for the transient simulation. The withdrawal due to the evapotranspiration until a certain depth can only be calculated in the FEFLOW through a plug-in to be developed using a programming interface. A more straightforward solution adopted was to create this top layer of fixed thickness – this thickness represents the extinction depth of the evapotranspiration. Given that, the FEFLOW allows to specify the inflow/outflow on the top layer, these values were included in the model, and an equation based on the depth from surface, element position and value of evapotranspiration at the surface was added. Using this technique, the amount of withdrawal due to the evapotranspiration in the transient state was calculated. As the formations represented by this layer are the same as the second layer, the parameter values were also the same.

The second layer represents the Barreiras formation (on top of the confined aquifer), the Beberibe formation (after 10km from the geological fault) and alluvial deposits near the rivers.

Such differentiation was made by adopting hydraulic conductivity zones for each formation. In the confined region, the bottom of this layer was defined by the Gramame formation; in the remaining area, the bottom represents a subpart of the aquifer depth.

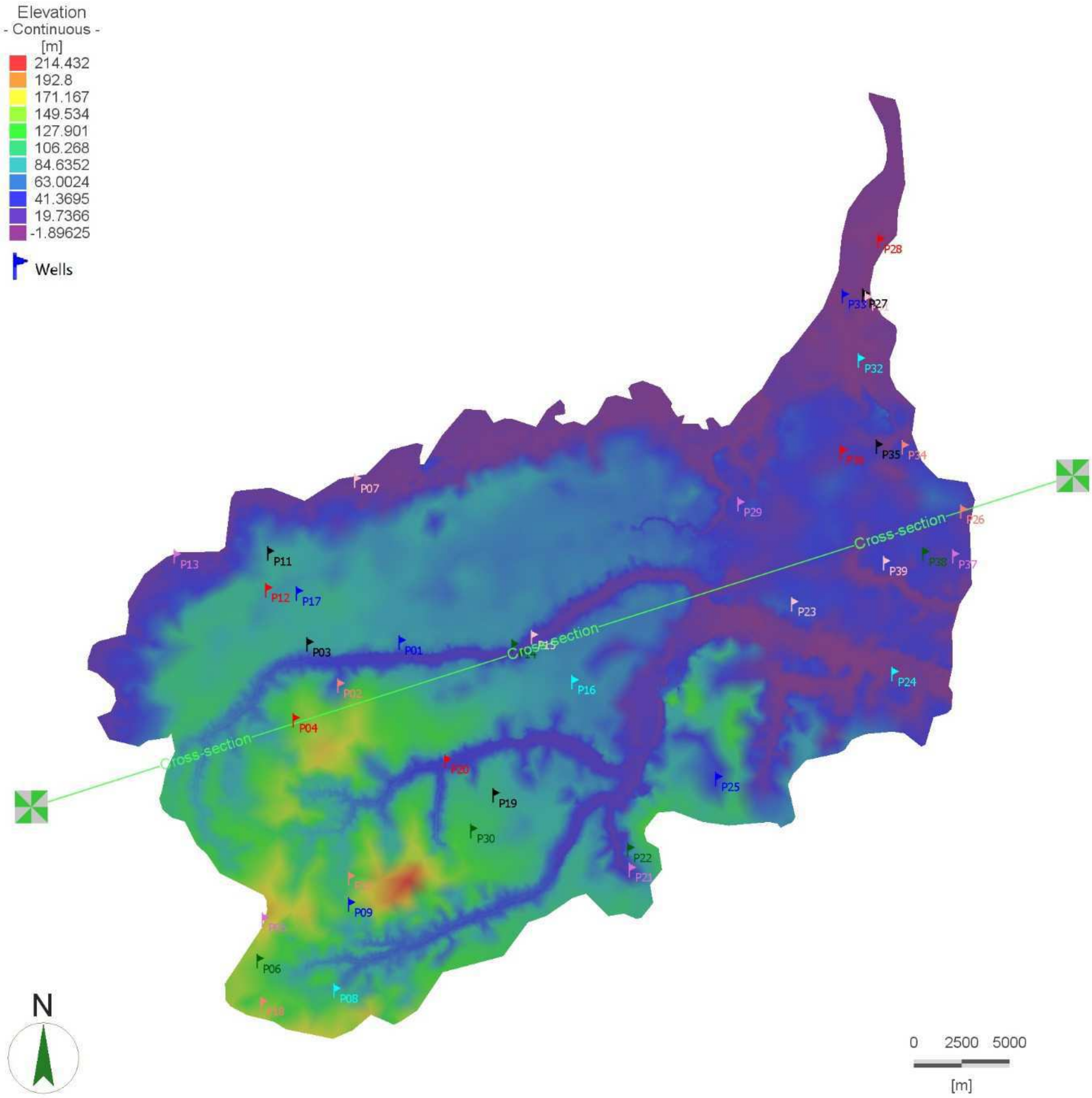


Figure 4.3 - Numerical model topography and location of the phreatic wells (colors are for differentiation)

The second layer represents the Barreiras formation (on top of the confined aquifer), the Beberibe formation (after 10km from the geological fault) and alluvial deposits near the rivers. Such differentiation was made by adopting hydraulic conductivity zones for each formation. In the confined region, the bottom of this layer was defined by the Gramame formation; in the remaining area, the bottom represents a part of the aquifer's depth.

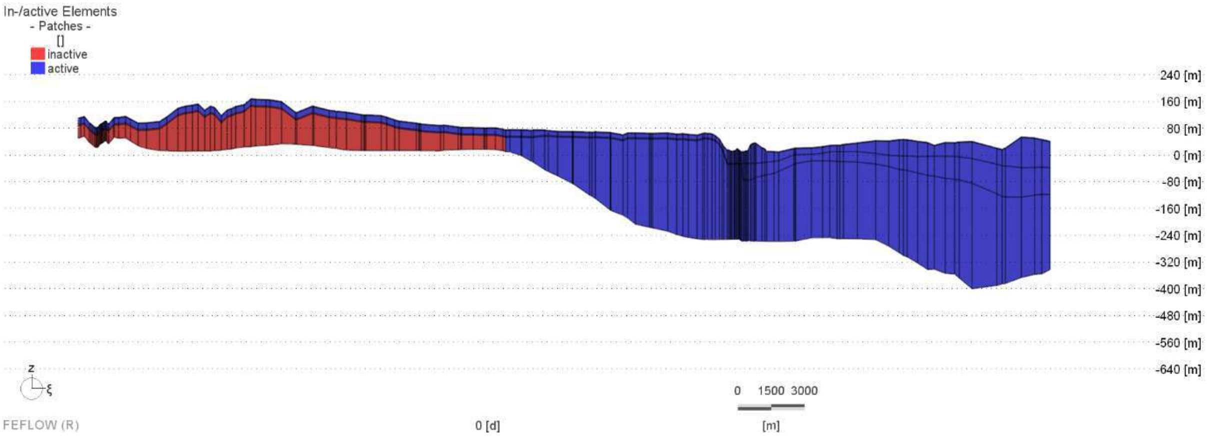

Figure 4.4 - Cross-section of the groundwater system

Finally, the fourth layer represents the Beberibe formation. The crystalline delimitates the bottom. At the west part of the system, this layer is delimitated by existing outcrops and has a small thickness, whereas, in the east part, it has a large thickness, this information was obtained from the lithological profile and geological cross-section. This variation in thickness of the formation occurs due to the geological process of sediment deposition. A cross-section depicting this variation is presented in Figure 4.4.

The temporal discretisation of the groundwater model was limited by the groundwater level data available. After compatibilization of the data available for both aquifers, the data used for the calibration/parametrisation of the groundwater model spanned from February 2017 to February 2018 with a monthly stress-period.

## 4.5 Water budget

In order to proceed with the calibration/parametrisation of the groundwater model, a water budget to estimate the recharge to the aquifer is necessary. Data from precipitation and evapotranspiration were obtained from several sources, and estimates of runoff were generated to the period of February 2017 to February 2018, given that, for this period, data from the variation in the potentiometric head was also available. The water budget for the study area was calculated on a monthly time-step following the approximation of the water budget equation:

$$R = P - E - Q \qquad \text{Eq. 4.6}$$

Where R is the infiltration towards the aquifer, P is the rainfall, E is the actual evapotranspiration and Q is the runoff. Furthermore, the water balance was calculated using a spatial resolution of 1km x 1km. Due to the temporal resolution of the model, the irrigation withdrawal was not taken into account given that such values are already included in the evapotranspiration. Similar procedure to calculate the water budget has been widely applied (Coelho et al., 2017; Crosbie et al., 2015; Szilagyi et al., 2011) in the literature. A more simplified application was conducted due to the lack of data, but still valid given the objective of the groundwater model.

### 4.5.1 Precipitation

Precipitation data were obtained from eleven rain gauges in the study area. The source of the rainfall data was the *Agência Estadual das Águas* (AESA) and *Centro Nacional de Monitoramento e Alerta de Desastres Naturais* (CEMADEN). A map with the spatial distribution of the rain gauges is presented in Figure 4.5, and the rainfall for all stations is presented in Figure 4.6. The data from this period was compared with the climatological normal from 1981 to 2010; the annual rainfall in eight stations was higher than the average. Besides, in six of the months, the monthly rainfall was higher than the climatological normal. Therefore, the period selected in this study can be considered as a wet year.

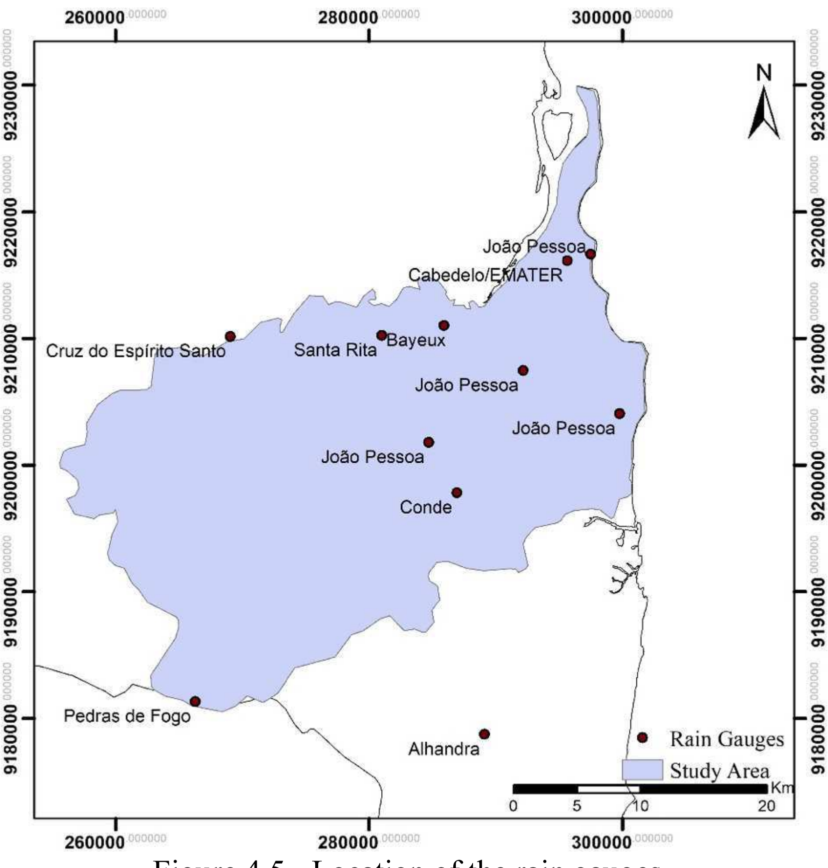


Figure 4.5 - Location of the rain gauges

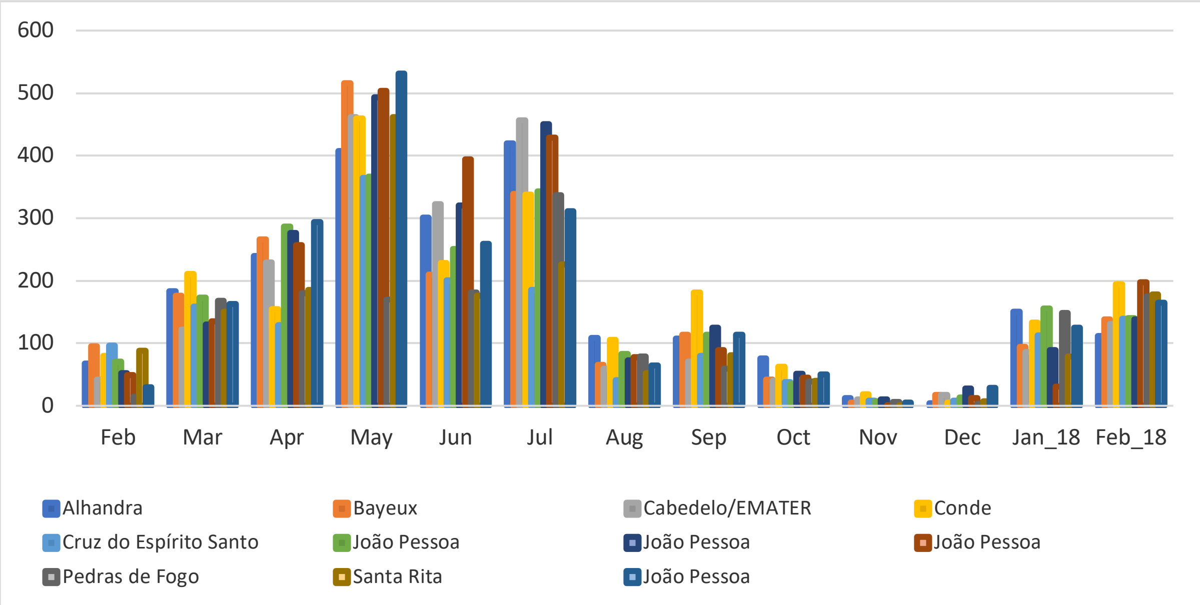


Figure 4.6 – Rainfall – Selected Gauges (Feb/17 to Feb/18)

### 4.5.2 Evapotranspiration

Evapotranspiration and temperature data for the study area was limited to the city João Pessoa. To improving the spatial coverage of this variable, the reference evapotranspiration for other locations was calculated using the method of Thornthwaite and Mather (1955). The temperature data was obtained using the ERA-Interim Reanalysis Dataset (Dee et al., 2011). Given that this data has a resolution of 0.5 x 0.5, a grid was implemented to calculate the values of ETo and afterwards, the ETo was interpolated using IDW for the study area. Figure 4.7 shows the ETo for the grid point (GR) with ERA data available, the dashed grey line shows the ETo from the climatological normal, and the dashed brown line shows the ETo for the INMET João Pessoa station. The ETo calculated using the Thornthwaite was underestimated when compared to the data of ETo from the INMET station. Such bias using ERA data have been previously reported (Paredes et al., 2018). When compared with the climatological normal, the selected period presented lower values of ETo.

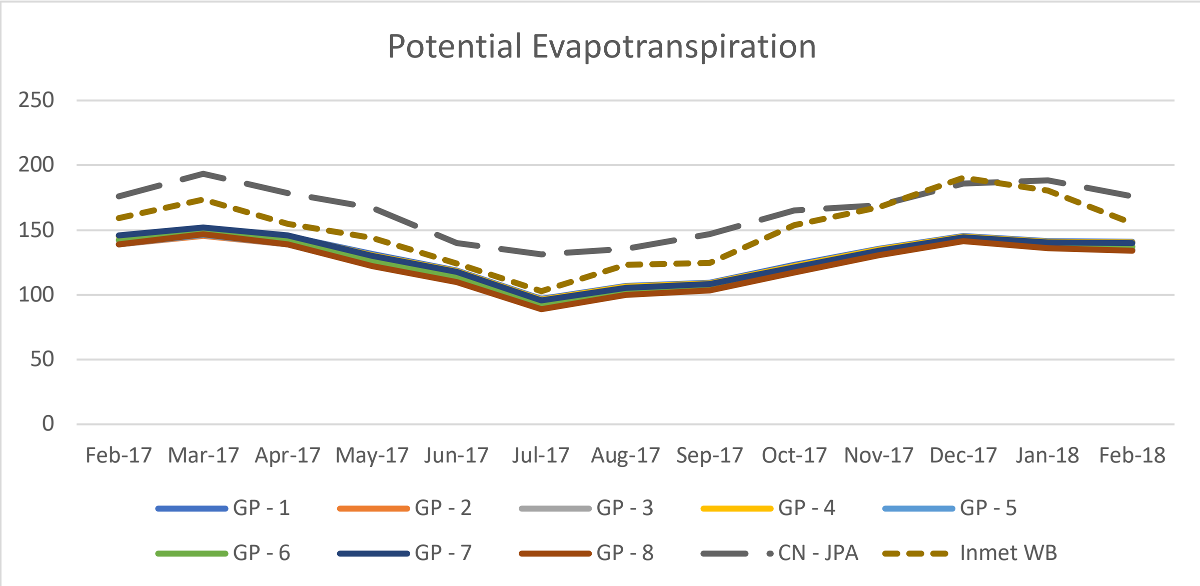


Figure 4.7 – Evapotranspiration between Feb/17 to Feb/18 (GP – Grid Points from the ERA-Interim)

### 4.5.3 Runoff

The estimation of the runoff followed the Curve Number Method proposed by the Soil Conservation Service (SCS) (USDA, 1986). This method estimates the runoff from known values of rainfall, based on a developed empirical equation for a large set of watersheds. The equation is expressed as:

$$\frac{F}{S} = \frac{Q}{P-Ia} \qquad \text{Eq. 4.7}$$

Where, F is the retention after the Q (runoff) begins, S is the storage in the watershed, equivalent to the potential infiltration, and Ia is the initial abstraction. The retention F can be expressed as:

$$F = (P - Ia) - Q, \qquad \text{Eq. 4.8}$$

The parameters Ia and S can be correlated using an estimation that Ia = 0.2S. This relation was verified by the USDA for many watersheds.

Hence, the equation can be simplified to:

$$Q = \frac{(P-0.2S)^2}{P+0.8S}$$ Eq. 4.9

To estimate the value for S, the SCS (USDA, 1986) obtained the following equation based on a large number of hydrographs for diverse watersheds:

$$S = \frac{25400}{CN} - 254$$ Eq. 4.10

To determine the curve number (CN) for the study area, the land use and the hydrologic soil group were necessary. The land use map was obtained from the MapBiomas database for the year 2017 (MapBiomas, 2019). The hydrologic soil group was determined from the existing soil classification for the study area using the classification proposed by Sartori et al. (2005). Finally, the extension ArcCN-runoff was used to generate the CN map, which is presented in Figure 4.8.

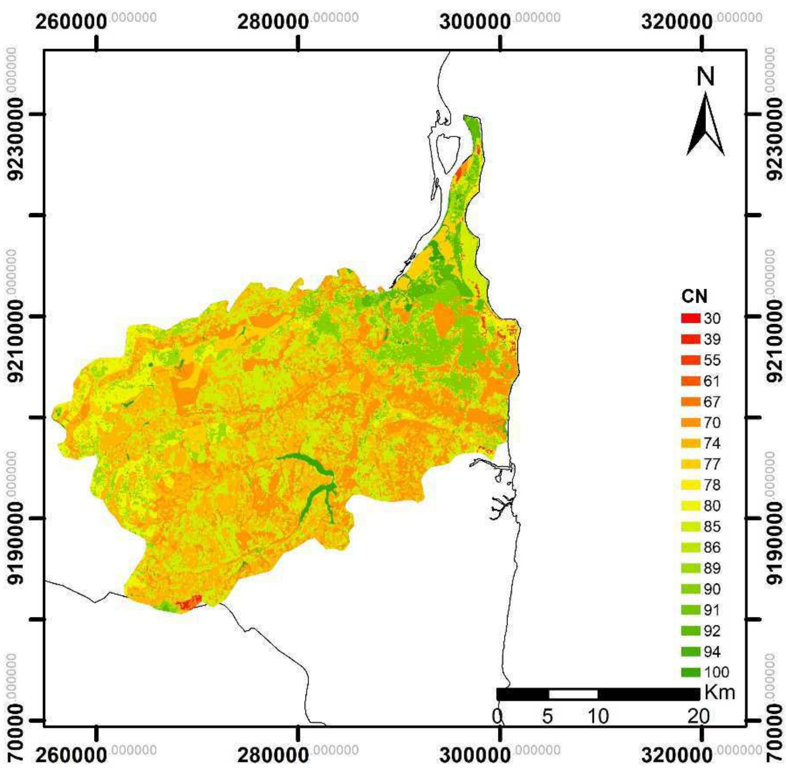

Figure 4.8 - CN value for the study area

### 4.5.4 Recharge

The groundwater recharge was calculated from the Eq. 4.6 using the software ArcGis, applying the raster layers of rainfall, actual evapotranspiration and runoff as input. For the calculation of the actual evapotranspiration, a monthly fixed rate of the potential evapotranspiration was assumed due to the lack of data. A positive value from the water balance means a recharge to the aquifer. The potentiometric heads data were not used to estimate the recharge, but to compare with the results from the water balance. As an example, Figure 4.10 and Figure 4.9 show the potentiometric head and the water balance for two locations. An analysis through visual comparison of the estimated recharge from the water balance and the recharge in the aquifer observed by the increase of the potentiometric head shows a fair agreement.

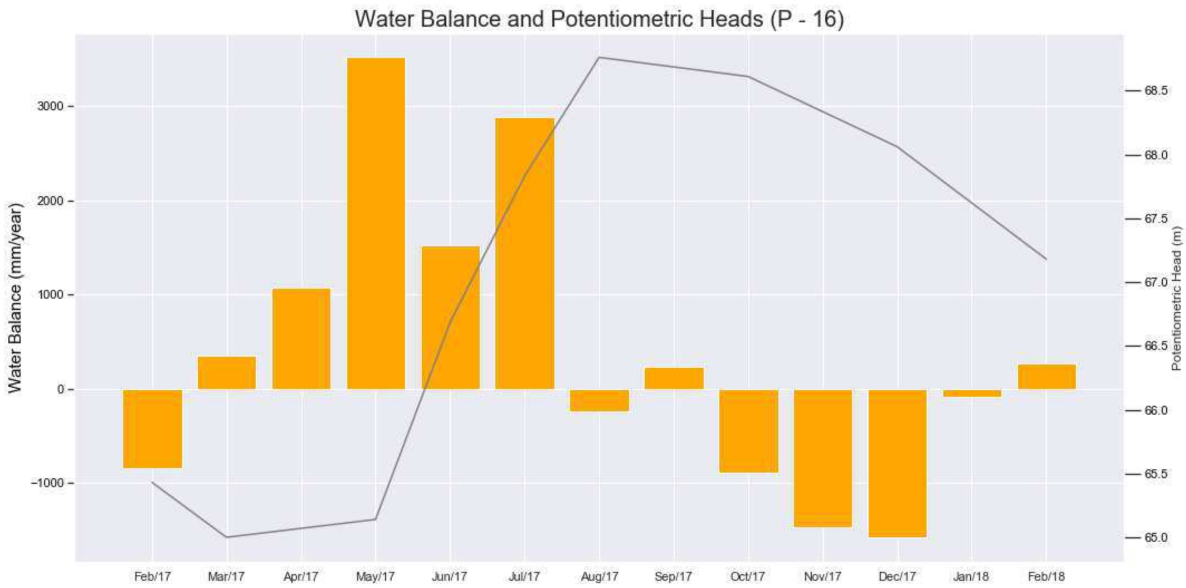


Figure 4.9 - Measured heads (grey) and water balance (orange) for well P16

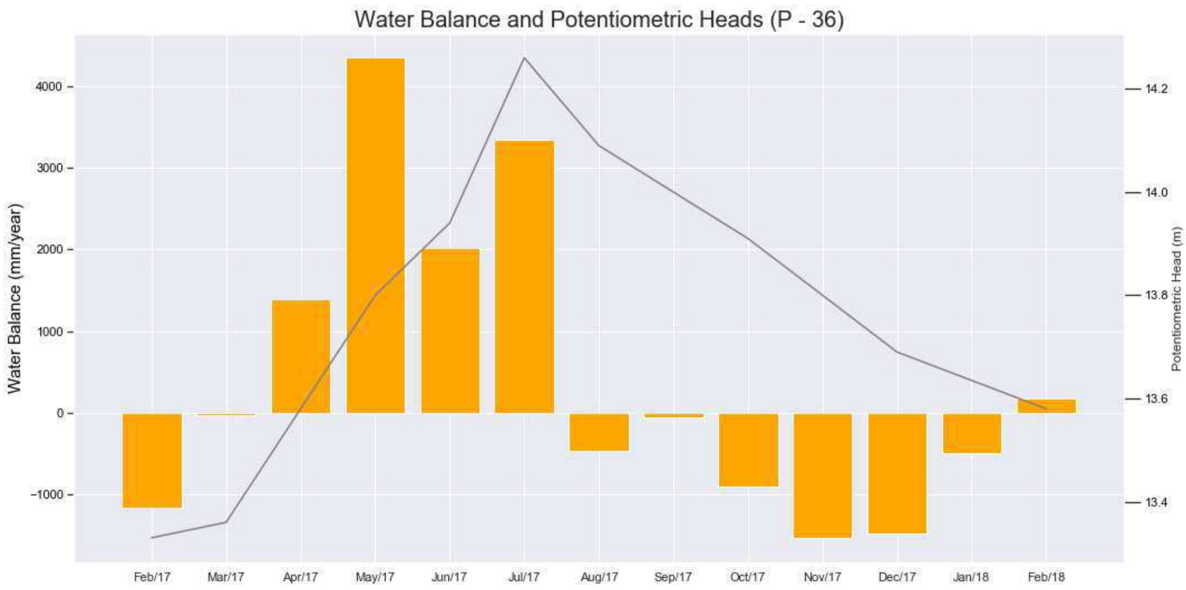


Figure 4.10 - Measured heads (grey) and water balance (orange) for well P36

## 4.6 Calibration for the steady-state

The calibration for the steady-state was performed for February 2017. Water level data for the Paraíba river was obtained from the National Water Agency (ANA), and linear interpolation was applied to estimate the values at the north boundary condition of the study area. The water level of the rivers within the study area was interpolated from the DEM. Both hydrogeological systems, phreatic and confined, were calibrated. For the phreatic subsystem, the study area was divided into zones according to the existing hydrogeological information seeking to represent the spatial distribution of the formations. For the confined aquifer, pilot points were used, given that this subsystem is composed of a single formation. Of the 39 wells measured in the phreatic aquifer, 11 were used as boundary conditions for this subsystem. In the confined aquifer, of the 13 wells, four were applied as boundary conditions. Due to the high number of interferences in the potentiometric heads caused by clandestine pumping in the confined aquifer, the calibration was conducted in a dynamic equilibrium. Therefore, 19 pumping wells were added. These wells are exploited by the State Water Supply Company (CAGEPA). The scatter plot between measured and simulated heads are shown in Figure 4.11 and Figure 4.12. Table 4.1 contains the statistics for the calibration process.

The model achieved a good correlation (Figure 4.11, Figure 4.12). Even though the system presents a high amplitude of potentiometric heads, varying between -3m to 108m for the phreatic aquifer, most of the heads were reasonably represented. A small bias underestimating the heads can be noted for the values of potentiometric heads higher than 80m. The confined subsystem presented a larger RMSE than the phreatic subsystem; however, the calibration of this subsystem will be discussed further (Chapter 5).

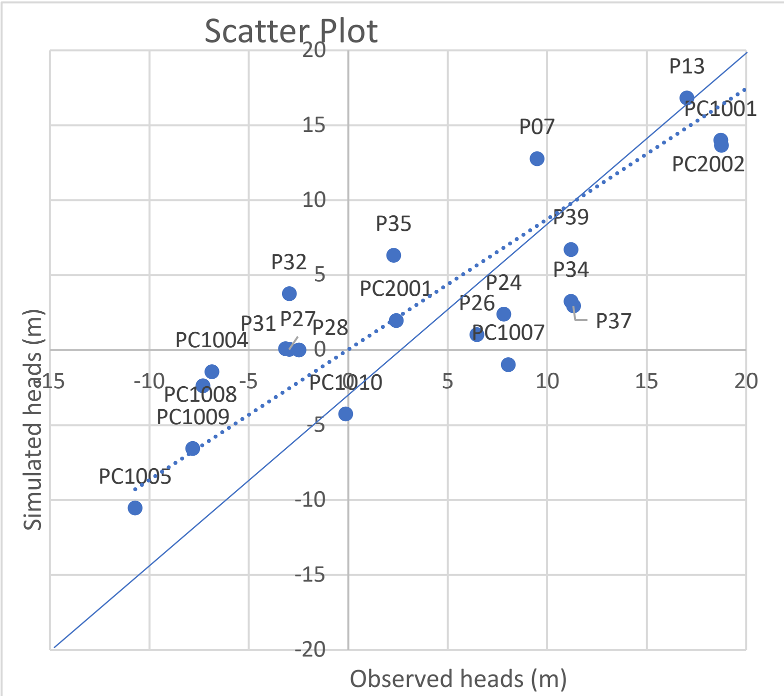


Figure 4.11- Scatter plot - Observed and simulated heads

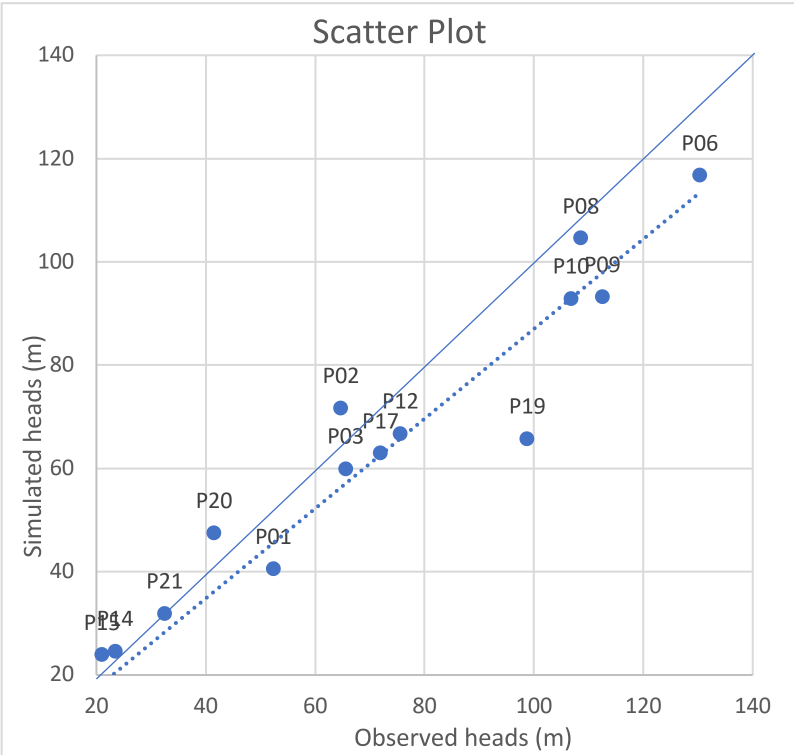


Figure 4.12 - Scatter plot - Observed and simulated heads

Table 4.1 - Statistics for the calibration (hydraulic head)

| | System | Confined | Phreatic |
|---|---|---|---|
| Mean absolute error (MAE) | 6.742 | 3.906 | 6.800 |
| Mean error (ME) | 3.960 | 1.311 | 4.626 |
| Normalized Root mean squared error (RMSE) | 6.446% | 16.073 | 5.864 |
| Root mean squared error (RMSE) | 9.090 | 4.738 | 9.870 |
| Correlation Coefficient | 0.982 | 0.921 | 0.985 |

Figure 4.13 presents the map of the modelled area with the relative error for each well applied as observation during the calibration procedure. The largest erros were concentrated close to the shoreline, most pronounced in the wells P35 and P32. Between these wells, there are several rivers such as the Mandacaru River and the Jaguaribe River which water level information were not available. Furthermore, the wells located in the northern area (P28, P27, P31, and P32) had negative heads, given that the boundary condition at east was the sea level (with hydraulic head equal to zero) and no withdrawal for the phreatic aquifer was present, the model could not simulated such condition.

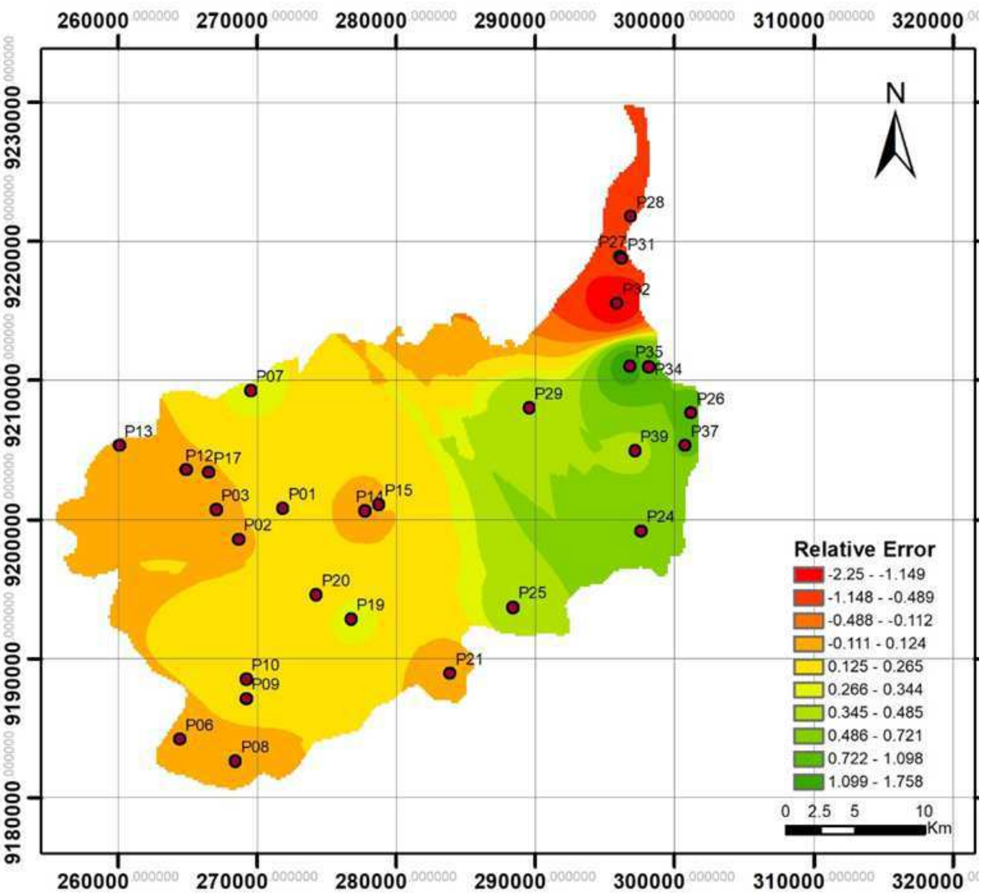


Figure 4.13 - Relative error for the hydraulic head

A map with the zones and calibrated values of hydraulic conductivity (K) (Figure 4.14) are presented. The values of K ranged between 1 m/d and 15.56 m/d. Table 4.2 summarises the K values and the hydrogeological formation in each zone. The alluvial formations, close to the rivers, resulted in the highest calibrated values for K. This is expected, given that, the alluvial deposits in the regions are majority formed by unconsolidated sands. The zone 11 comprises the Barreiras formation, but there is also the presence of sands of dunes. The calibrated values of K are equivalent values for the whole patch; therefore, the value of 12.65 m/d can be considered reasonable for this zone. In general, the calibrated values presented a good agreement compared with the geological characteristics.

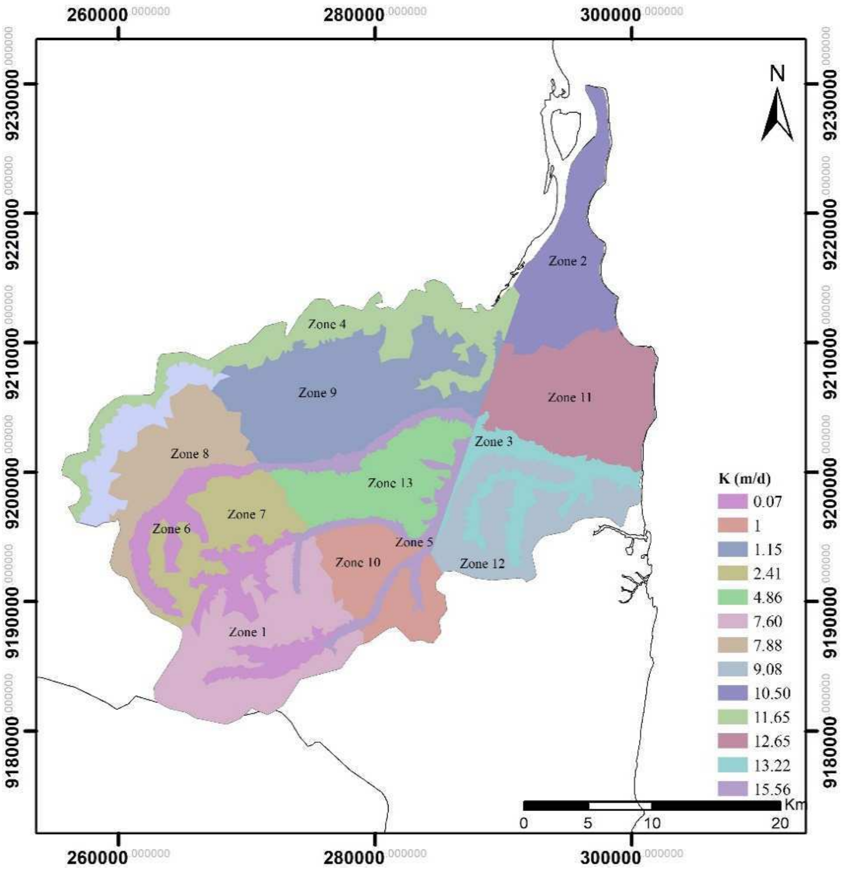


Figure 4.14 - Zones of hydraulic conductivity and calibrated values

Table 4.2 - Relationship between calibrated values of K and the geological formation

| Zone | K - Calibrated Value (m/d) | Description of the formation |
|---|---|---|
| 1 | 7.60 | Barreiras/Beberibre - The geological map shows this area as Barreiras formation; however, the local lithology presents diverse sandstones, suggesting an outcrop of the Beberibe formation |
| 2 | 10.50 | Alluvial formation of the Paraiba river on top of the Barreiras formation |
| 3 | 13.22 | Alluvial formation of the Gramame river on top of the Barreiras formation |
| 4 | 11.65 | Alluvial formation of the Paraiba river on top of the crystalline bedrock |
| 5 | 15.56 | Alluvial formation of the Gramame river on top of the Barreiras and Beberibe formation |
| 6 | 0.07 | Crystalline outcrops (Granitoids and Sertânia formation)<br>It was adopted a low value of K to represent the low conductivity in this region |
| 7 | 2.41 | Barreiras/Beberibre - The geological map shows this area as Barreiras formation; however, the local lithology presents diverse sandstones, suggesting an outcrop of the Beberibe formation |
| 8 | 7.88 | Barreiras/Beberibre - The geological map shows this area as Barreiras formation; however, the local lithology presents diverse sandstones, suggesting an outcrop of the Beberibe formation |
| 9 | 1.15 | Barreiras formation |
| 10 | 1.00 | Barreiras formation |
| 11 | 12.65 | Barreiras formation with the presence of unconsolidated sands from dunes, and sea/river sediments |
| 12 | 9.08 | Barreiras formation with the presence of unconsolidated sands from dunes, and sea/river sediments |
| 13 | 4.86 | Barreiras formation with the presence of outcrops from the Beberibe Formation |

A map with the potentiometric head for the phreatic system is shown in . The simulated values of hydraulic heads for the phreatic aquifer were able to represent the flow characteristics — e.g., the direction of the flow towards the shoreline and to the Paraíba River. Two boundary conditions. Internally, the direction of the flow to the rivers (Gramame, Mumbaba e Mamuaba) was also well represented by the model.

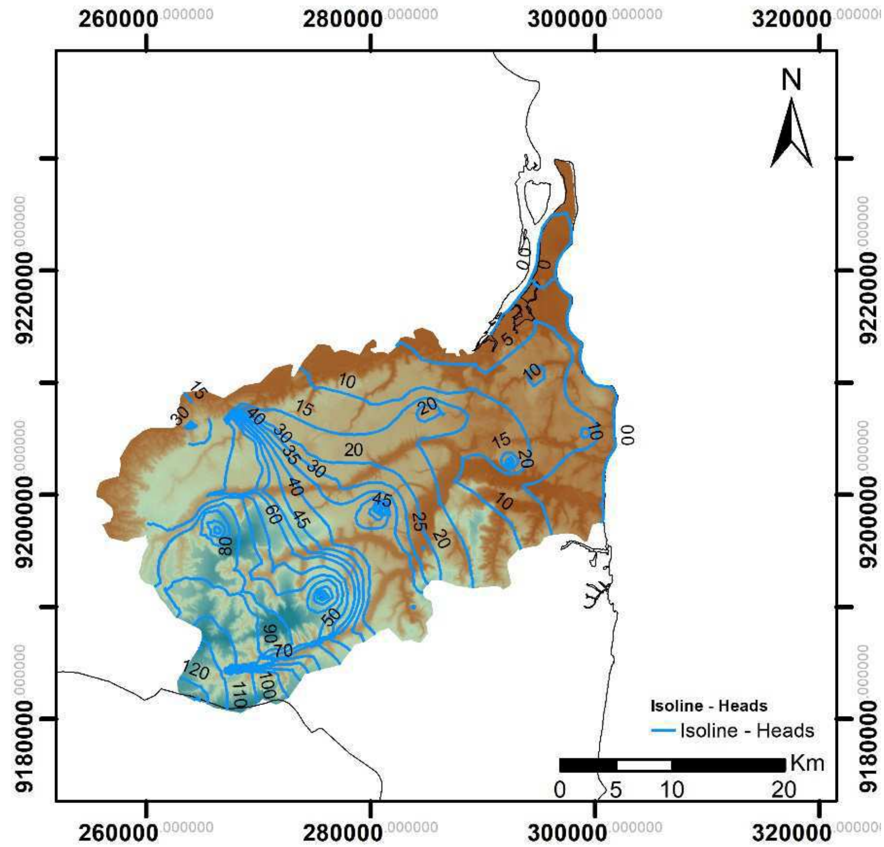

Figure 4.15 - Isolines: simulated potentiometric heads

## 4.7 The parameterisation for the transient state

Unfortunately, a calibration for the transient state was not possible to achieve. The data scarcity limited the possibility of matching the simulated and observed heads on a temporal scale. The system presents a complex flow configuration. While part of the flow is directed towards the sea, another part is discharged to the rivers as baseflow. Therefore, the calibration for the transient state requires the variation of the water level at the rivers throughout the period. Due to such a lack of data, the application of the FePest tool for the transient state presented unsatisfactory results. These data could be obtained through a hydrologic surface model; however, such modelling effort lies outside the scope of this thesis.

Given that the purpose of this groundwater model is for proof of concept, the hydrogeological parameters were estimated and not calibrated. Afterwards, a refinement of these parameters was manually conducted. The objective is that the groundwater model represents the general hydrogeological behaviour of the system.

The effective porosity ($n_e$) values for the phreatic aquifer were determined following the zoning obtained during the calibration for the steady-state. The Biecinski equation (Pazdro, 1977) was applied to calculate the first estimate of effective porosity:

$$n_e = 0.117\sqrt[7]{k} \qquad \text{Eq. 4.11}$$

Where k is the hydraulic conductivity in m/day.

Reference values of effective porosity from the literature were also used to guide the determination of the final parameter. Afterwards, a manual trial and error process was applied to adjust the values of the simulated heads with the observed ones. The final values of effective porosity are presented in Table 4.3. The $n_e$ values varied between 11.7% and 27.6%. The highest values were attributed to the alluvial deposits, given that these formations are composed, in their majority, by fine and medium sands. The lowest values were attributed to the regions of the Barreiras formations. In these regions, there is a presence of sand and clay. For zone 6, where crystalline outcrops are located, the value of 0.001 was adopted to represent the impermeable characteristics of rock formations.

Table 4.3 - Effective porosity values for zones of K

| Zone | K (m/d) - Calibration | Effective Porosity |
|---|---|---|
| 1 | 7.60 | 0.125 |
| 2 | 10.50 | 0.262 |
| 3 | 13.22 | 0.270 |
| 4 | 11.65 | 0.266 |
| 5 | 15.56 | 0.276 |
| 6 | 0.07 | 0.001 |
| 7 | 2.41 | 0.132 |
| 8 | 7.88 | 0.150 |
| 9 | 1.15 | 0.119 |
| 10 | 1.00 | 0.117 |
| 11 | 12.65 | 0.168 |
| 12 | 9.08 | 0.160 |
| 13 | 4.86 | 0.146 |

The model was able to represent some of the hydrogeological behaviours of the system, such as the rising of heads due to the recharge and the reduction due to evapotranspiration losses and baseflow. The groundwater budget for the period also presented a reasonable estimated of the behaviour. A reasonable parametrisation was achieved, taking into account the limited data. Following are the charts with the simulated and observed heads for selected wells and the water balance.

From the charts, it can be seen that the numerical model was able to simulate the aquifer's response to the recharge process. The positive part of the water balance (orange bars) was succeeded by an increase in potentiometric heads, as well as the opposite. Twenty-three wells could be used to compare the simulated and observed heads. In eight wells, a reasonable representation of the behaviour of the system was not achieved. In three well (P10, P13, P39), the model represented closely both the value of the head as well as the behaviour (rising and descend) of the heads. In the remaining wells, it was possible to represent the behaviour of the heads. Among them, the wells P24, P29, P35, and P39 depicted in Figure 4.16 to Figure 4.20.

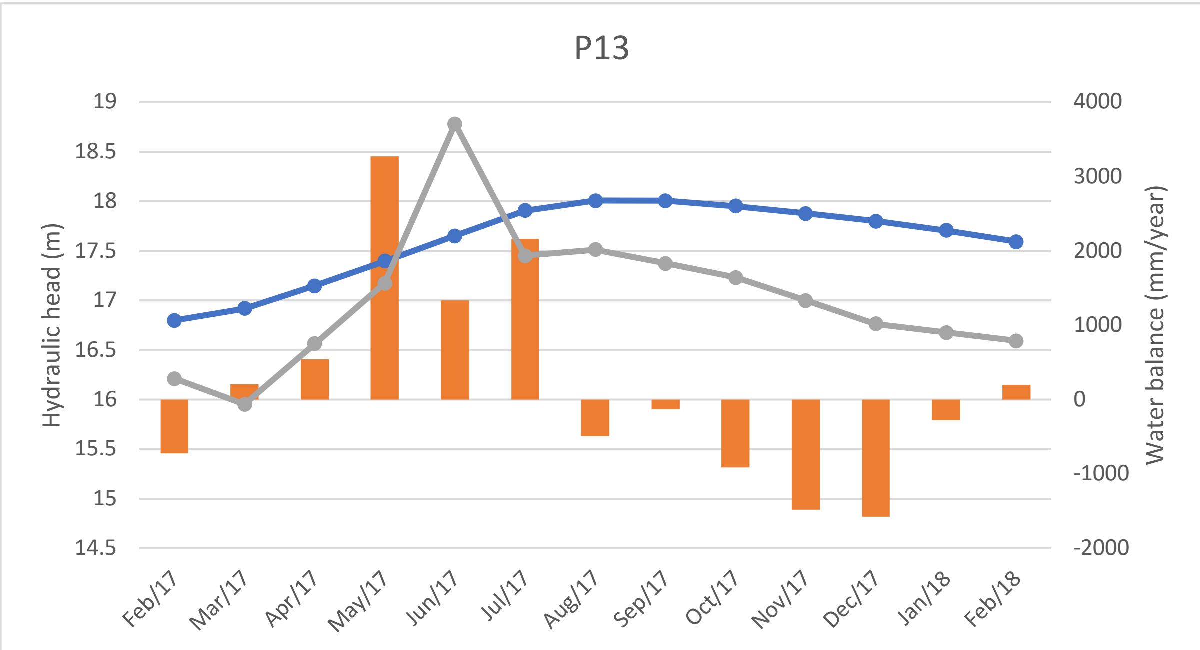


Figure 4.16 - Simulated (blue) and observed heads (grey), and water balance (orange) for the well P13

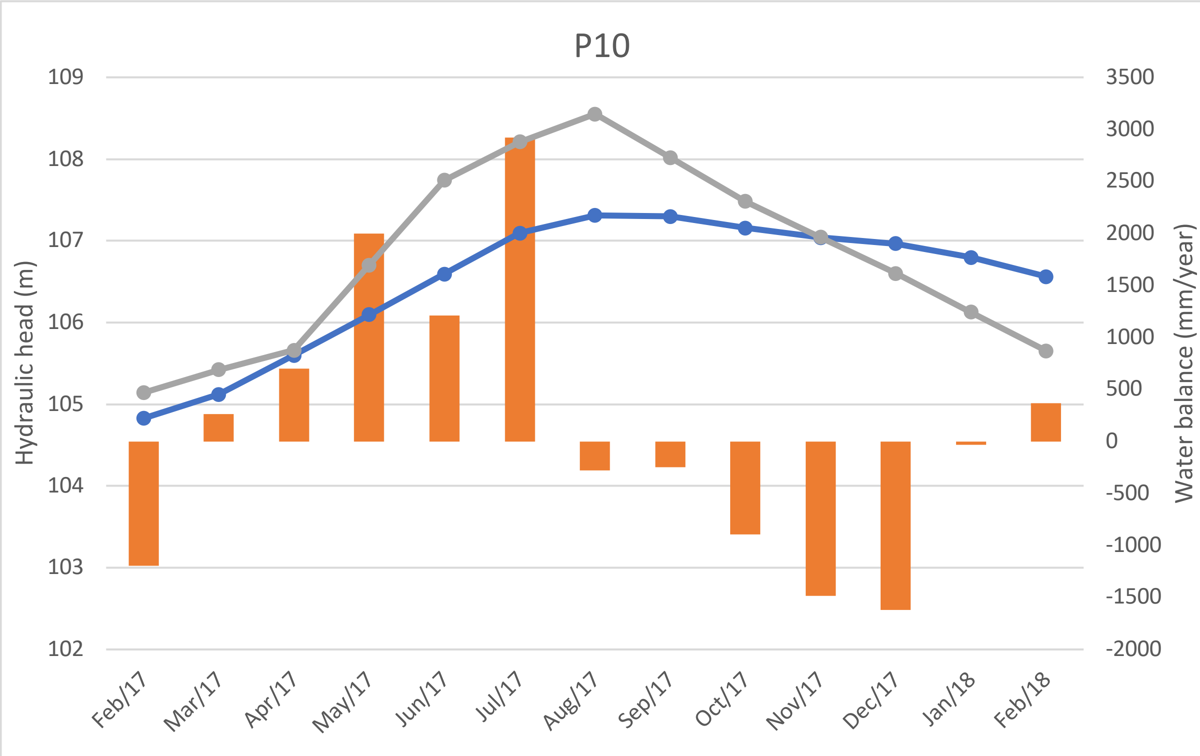


Figure 4.17 - Simulated (blue) and observed heads (grey), and water balance (orange) for the well P10

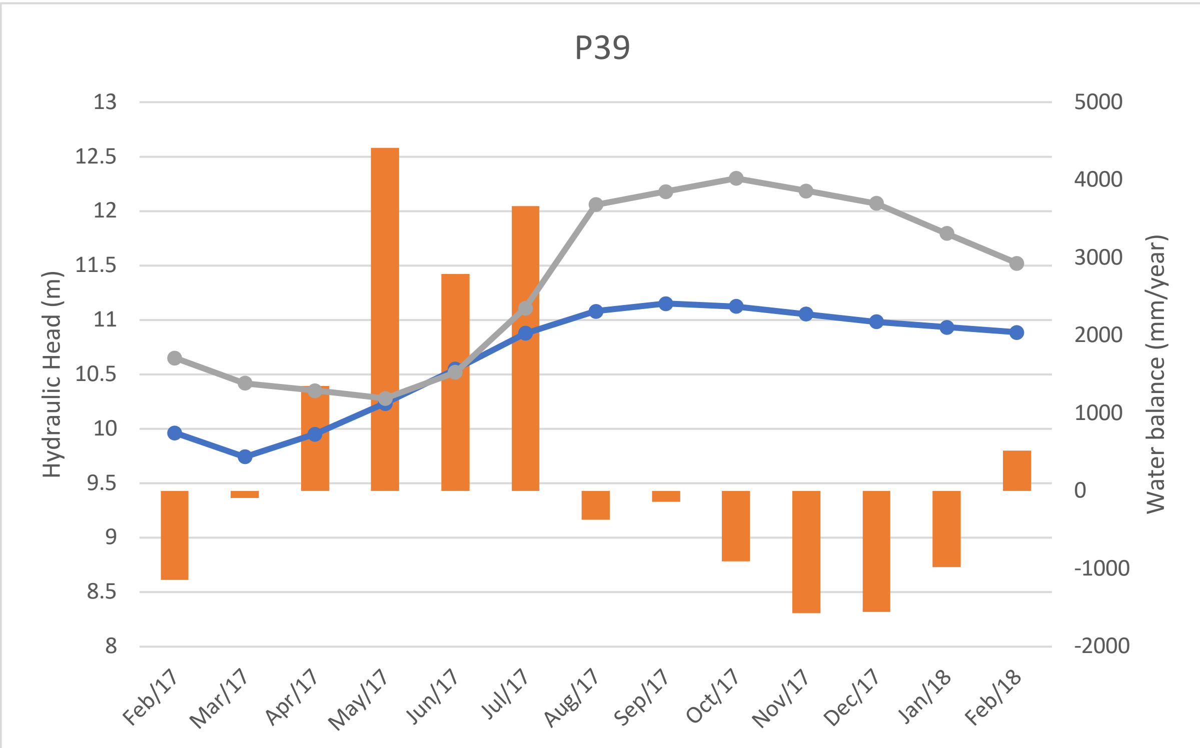


Figure 4.18 - Simulated (blue) and observed heads (grey), and water balance (orange) for the well P39

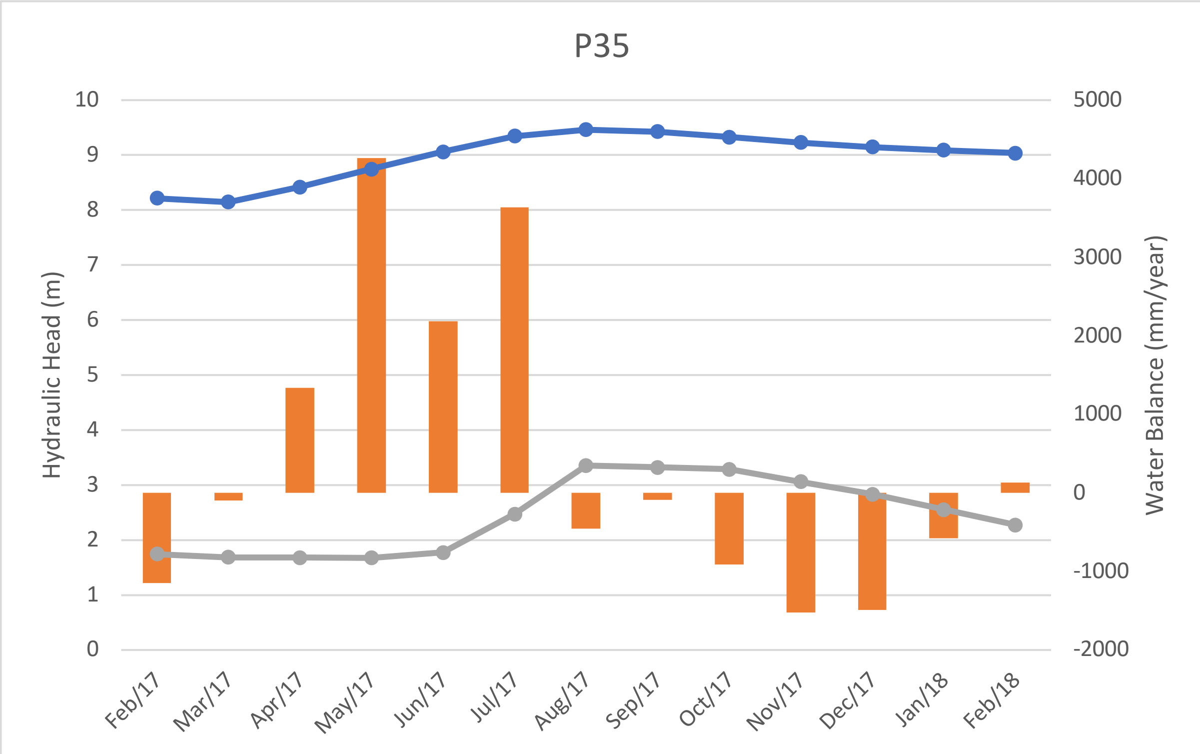


Figure 4.20 - Simulated (blue) and observed heads (grey), and water balance (orange) for the well P36

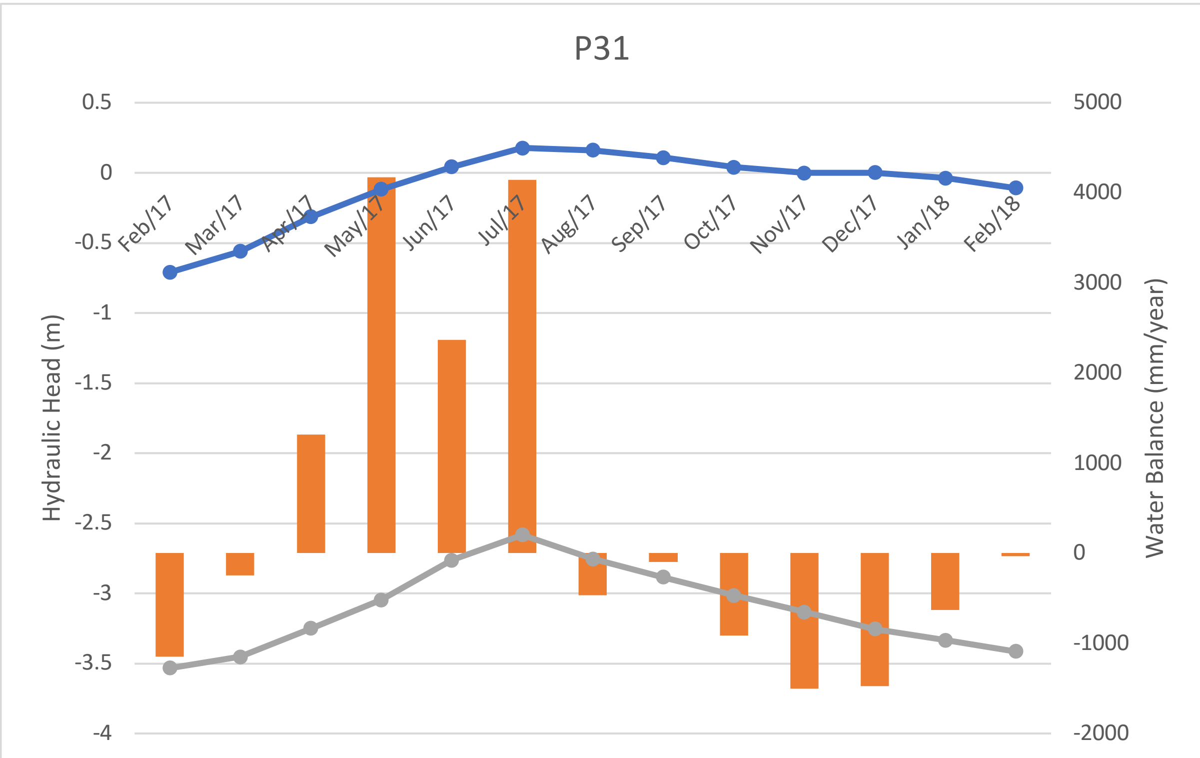


Figure 4.19 - Simulated (blue) and observed heads (grey), and water balance (orange) for the well P31

## 4.8 Conclusions

Based on the analysis of the existing data it was possible to determine a conceptual model for the study area, identifying the general flow behaviour, inflows, outflows and recharge setting for the groundwater system as well as for its confined and phreatic subsystems.

The numerical groundwater model for the study area was successfully calibrated for the steady-state, achieving satisfactory statistical indexes such as the normalised root mean error (6.44%) and mean absolute error (6.74m), considering the wide variety of hydraulic heads in the region. For the phreatic subsystem, it varied from 130m to -3m, while for the confined subsystem it varied from -10m to 18m. Through the analysis of the lithological profiles and geological maps, 13 zones of hydraulic conductivity were determined. The values of hydraulic conductivity obtained varied between 1 m/d and 15.56 m/d. These values represented from sandstone with the presence of clays to sandy alluvial aquifers. Furthermore, the isolines of simulated potentiometric heads presented a good agreement with the existing one determined in previous studies.

For the transient state, due to the limited existing data regarding the river's levels, it was not possible to proceed with automatic calibration. As an alternative, a parametrisation of the specific porosity and coefficient storage was conducted, followed by a manual trial and error adjustment of the parameter. The values of specific porosity were determined based on the hydraulic conductivity obtained for the 13 zones. These values spanned from 11.7% to 27.6%. Finally, model simulations for the transient period were able to represent the hydrogeological behaviour of the system with a reasonable matching, such as the rising of heads due to the recharge and the reduction due to evapotranspiration losses. In eight wells, a reasonable representation of the behaviour of the system was not achieved. In three, the model represented closely both the value of the head as well as the behaviour (rising and descend) of the heads. In the remaining wells, it was possible to represent the behaviour of the heads.

# Chapter Five - Increasing the knowledge of a groundwater system using estimates of hydraulic conductivity in data-scarce area

## 5.1 Introduction

Groundwater management seeks sustainability through a balance between withdrawn and recharge. Among the most common measures for groundwater management are the volumetric allocation, trigger-level management, and buffer zones (Noorduijn et al., 2018). To underpin the effective application of these measures, knowledge about the groundwater system is necessary (Eden et al., 2016). The building of knowledge lies in the provision of scientifically sound information. Thus, this is of paramount importance to the successful outcome of evidence-based groundwater management (Foster and Chilton, 2017; Pezij et al., 2019).

There is a relationship proposed by the Data-Information-Knowledge-Wisdom (DIKW) hierarchy, where knowledge is derived from information, and information comes from data; this hierarchy is not unidirectional, flowing both ways (Ackoff, 1989; Rowley, 2007). Data are records of a past or present situation (van der Gun, 2017). Information is generated by processing the relevant data, and information also carries a message (van der Gun, 2017). When the information is combined with previous experience or perception, knowledge is then produced (van der Gun, 2017). Among the different processes to produce knowledge, one can be described as a synthesis of multiple sources of information (Rowley, 2007). In groundwater management, limited data, information and, consequently, knowledge leads to a range of uncertainties and difficulties to manage the resource (Jakeman et al., 2016).

Groundwater is quintessentially a hidden resource (Albrecht et al., 2017) due to some aspects. First, the spatial delimitation of groundwater systems is difficult, given that the boundaries for the system depend on the geological characteristics (Gondwe et al., 2011). Second, most aquifers span through large areas where their hydrogeological parameters, such as hydraulic conductivity, are mostly unknown, and only localised data are available (Gondwe et al., 2011). Third, the definition of recharge and discharge areas within the aquifer is not trivial, requiring more careful analysis of the hydrogeological settings (Batelaan et al., 2003). Therefore, uncertainty is an inherent part of groundwater management. These aspects highlight the need to increase knowledge about the groundwater system to be managed. Numerical modelling and geostatistics are essential tools to understand the complexities of any groundwater system.

A model is a simplified representation of a physical process. In groundwater, numerical models have been applied for a long time to analyse a range of issues — both in quantity (Rotiroti et al., 2019; Wang et al., 2019) and quality aspects (Kalhor et al., 2019; Zhang et al., 2019). Two main steps are needed to build a groundwater model (Anderson et al., 2015b). Initially, a schematic representation of the system behaviour must be developed. This is known as the conceptual model (Gondwe et al., 2011), and it is based on the current knowledge of the system. Subsequently, this conceptual model is translated into the numerical model through the groundwater flow equations and boundary conditions (Anderson et al., 2015b). In both steps, a wide variety of data are necessary. Groundwater models can be used as a tool to provide information for groundwater management after calibration is achieved (Hutchins et al., 2018). The calibration presents a set of hydrogeological parameters obtained from comparing calculated values with the observed data. The evaluation of the calibration is done using traditional statistical indices, such as Mean Absolute Error (Silvestro et al., 2015) or Root Mean Square Error (Zipper et al., 2017). However, this set of parameters is not a unique solution and, if no change is made to the system, it can vary according to the bounds defined for the hydrogeological parameters. These bounds are generally defined using data and information available throughout literature or using data from field surveys (Anderson et al., 2015b).

Geostatistics has been traditionally applied in groundwater studies to map and characterise hydrogeological parameters (Fontenele et al., 2014; Rotzoll and El-Kadi, 2008). This method allows a spatial representation of a variable based on sample data (Razack and Lasm, 2006). Among the classical methods, Kriging is the most commonly applied in groundwater (Varouchakis et al., 2012). Kriging has been used to interpolate groundwater level (Varouchakis et al., 2012), hydraulic parameters (Fontenele et al., 2014), hydrogeochemical parameters (Karami et al., 2018) and geophysical characteristics of the system (Naranjo-Fernández et al., 2018), generating spatially distributed information. However, the efficacy of this method depends not only on the methods but also on the amount and quality of data available (Fontenele et al., 2014; Karami et al., 2018).

In many aquifers, the hydrogeological data are dispersed, scarce or can even present potential misinformation (van der Gun, 2017). Data scarcity hinders the use of numerical models or geostatistics because a satisfactory agreement between observed data and simulated values becomes tougher to achieve (Wu et al., 2011). As the monitoring of groundwater systems is usually costly and time-consuming (Alfaro et al., 2017), such challenge is hard to overcome. To utilise numerical modelling and geostatistic in these regions, certain approaches are applied. This includes using data from previous studies or reports (Candela et al., 2014), remote sensing to fill data gaps (Switzman et al., 2015) and refinement of the conceptual model (Wu et al., 2011) so that scarce data challenges can be surpassed and a reliable tool can be achieved to combine and provide information about the groundwater system.

Used to support groundwater management, numerical groundwater flow models need an essential parameter to its calibration: the spatial distribution of the hydraulic conductivity (K). Accurate measurements of K are rare in many regions of the world (Alfaro et al., 2017). However, when wells are drilled, two sets of data are readily available: the discharge rate (Q) and the drawdown (s) (Priebe et al., 2018). The ratio of these data determines the specific capacity, and this can be used to estimate an empirical relationship with transmissivity (T). Another approach is determining the specific capacity index ($S_i$), defined as the specific capacity divided by saturated aquifer thickness, and after that, estimating an empirical relationship with the hydraulic conductivity (K) (Mace, 2001; Verbovšek, 2008). The scientific literature presents a plethora of examples on the determination of such estimates for different types of geological formations (Mace, 2001; Priebe et al., 2018; Richard et al., 2016; Verbovšek, 2008). Nevertheless, approaches of how the information generated through these estimates can be effectively combined with other sources of information to improve the knowledge about a groundwater system are still vague, especially in data-scarce regions.

This paper seeks to evaluate whether these empirical estimates of hydraulic conductivity can be applied to increase the knowledge of a groundwater system aiding the provision of information for groundwater management. More specifically: i) if these empirical estimates of hydraulic conductivity can be used to improve the calibration of a groundwater model by reducing its uncertainty; ii) if a valid spatialization of the estimates of hydraulic conductivity can be determined by the application of geostatistics.

## 5.2 Material and Methods

### 5.2.1 Study Area

An example of a scarce data region with a complex hydrogeological setting can be found in the coastal area of Paraíba State, in the Northeast of Brazil (Figure 5.1). The capital of the state, João Pessoa, is located in this region and has a conurbation with two other cities

(Cabedelo and Bayuex) presenting more than 800,000 inhabitants. The precipitation in the area is concentrated during March and August; hence, groundwater is used as part of the water supply, especially during the dry season (Braga et al., 2018). In this area, there are two overlapped aquifer formations, Barreiras and Beberibe (from top-down), separated by a confining formation called Gramame. The confined aquifer in this portion of the study area comprehends the Beberibe formation and has an area of approximately 300 km$^2$. This formation consists of sandstones sitting on top of the crystalline bedrock. The confined aquifer has a better water quality, and as a consequence, it has been more exploited for water supply (ASUB, 2010).

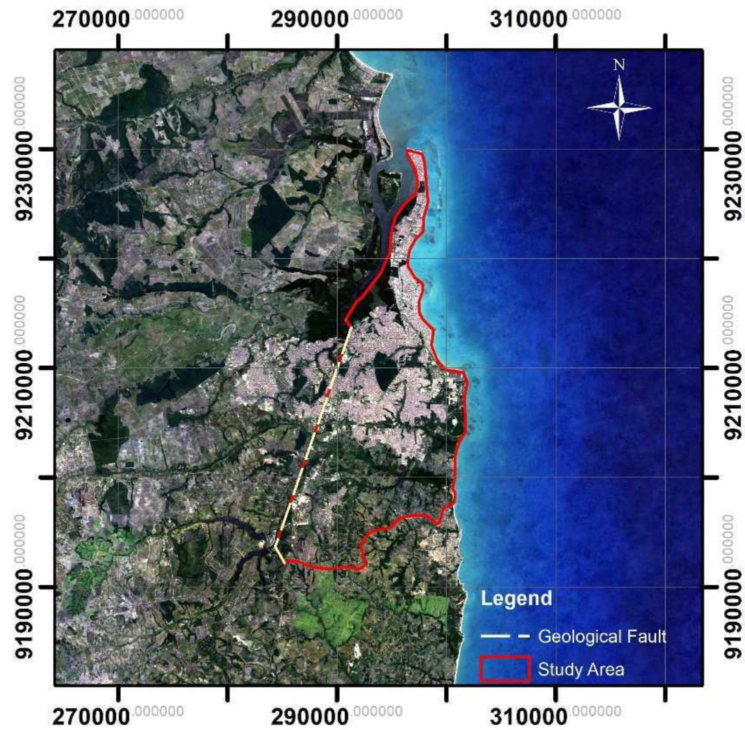


Figure 5.1 – Study Area

Batista et al. (2011) and Braga et al. (2015) have detected groundwater over-exploitation problems in this area, through the application of hydrogeological modelling. Furthermore, the geological characteristic of the confined aquifer implicates that this system presents a natural recharge area with positive (above sea level) heads; therefore, negative heads is an indication of overexploitation (Walter et al., 2018). Even though this region has a large number of exploitation wells, potentiometric data from monitoring network, pumping test with observation wells, and more detailed characterisation of the groundwater system are scarce.

### 5.2.2 Methodological approach

The general methodological approach for this paper followed these steps: i) an estimate of the empirical relationship between the specific capacity index and hydraulic conductivity was determined; ii) the groundwater model was calibrated for two different settings and bounds for the hydraulic parameters, one using literature values (C1) and the second using values obtained from the estimates (C2); iii) uncertainty analysis of the calibrated models was done applying Monte Carlo method; iv) results were compared by evaluating the changes in variance and mean absolute error among the simulated values of the two calibrated models; v) finally, the empirical relationship was spatialised applying the geostatistical method Cokriging, using the hydraulic conductivity and specific capacity index as variables.

### 5.2.3 Data

At total, 30 lithological profiles obtained from drillers and one geological cross-section available were analysed with the aid of Geographical Information Systems (GIS) to identify the geometrical characteristics of the hydrogeological formation in the study area. The water level from 12 wells in the confined aquifer was measured between August 2016 and March 2018. The month of February 2017 was chosen to calibrate the model for the steady-state, given that this was the driest month during the time-series available. The hydraulic head was measured using automatic divers with an accuracy of 0.05%/50m to reduce the uncertainty.

Furthermore, two different sets of data of specific capacity were used. The first set was applied to determine the values of hydraulic conductivity in the region. This set was obtained from a database of previous studies conducted in the area and consisted of data from short-duration pumping tests. The second set of data was applied to estimate the hydraulic conductivity from the specific capacity based on the empirical relationship. This set was obtained from the State Water Agency (AESA) database and Groundwater Information System (SIAGAS) and consisted of coordinates, discharge, dynamic and static level, the radius of well, and depth of the well.

### 5.2.4 Estimates of Hydraulic Conductivity

The method proposed by Cooper and Jacob (1946) was applied to the dataset containing the short-term pumping test to determine the transmissivity. For a confined aquifer, the transmissivity is the saturated aquifer thickness multiplied by the hydraulic conductivity. Hence, the values for hydraulic conductivity were calculated from these values of transmissivity. The drawdown was determined by calculating the difference between the initial head and the head at the end of the test when the equilibrium was achieved. The specific capacity was then calculated, dividing the discharge by the drawdown. The specific capacity index was calculated dividing the specific capacity by the saturated aquifer thickness to normalise the dimensions with the hydraulic conductivity, as suggested by Verbovšek, (2008). This process resulted in pairs of data containing the hydraulic conductivity and the specific capacity index for the selected wells.

For these pairs of data, a log transformation was applied, and then, the least-squares linear regression model was used to determine the empirical relationship of the K and $S_i$. The correlation coefficient of this relationship was calculated, and prediction intervals at a significance level of 0.05 were also calculated to determine an envelope around the best fine line, as suggested by several authors (Mace, 2001; Verbovšek, 2008).

### 5.2.5 Groundwater Modelling

The numerical model is the mathematical representation of the physical model. To this end, the software Finite Element subsurface Flow System – FEFLOW (Diersch, 2014) was applied. The FEFLOW has been widely applied to groundwater simulations (Lin and Lin, 2019; Usman et al., 2019; Viaroli et al., 2019). The FEFLOW simulates the groundwater flow solving the general flow equation, according to the defined boundary and initial conditions. This equation mathematically represents the spatial variation of the hydraulic head according to the hydraulic parameters of the aquifer (hydraulic conductivity and specific storage) for the steady-state.

$$\frac{\partial}{\partial x}\left(K_x \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial y}\left(K_y \frac{\partial h}{\partial y}\right) + \frac{\partial}{\partial z}\left(K_z \frac{\partial h}{\partial z}\right) = W \qquad (1)$$

Where,

$Kx$, $Ky$, $Kz$ are the hydraulic conductivity in each direction,

$\partial h/\partial x$, $\partial h/\partial y$, $\partial h/\partial z$ are the components of the hydraulic gradient.

SS is the specific storage; and,

W are the variations in storage due to inflow or outflow.

### 5.2.6 Calibration

The calibration process seeks to determine the equivalent values for hydrogeological parameters of the groundwater system by adjusting these values within determined bounds and comparing the calculated hydraulic head with the observed one. This is an iterative process consisting of sequential steps (Anderson et al., 2015b). First, a variable was defined to calculate the objective-function for the calibration. The objective function can be defined as the sum of the weighted square residues (the difference between the calculated head and observed head). This variable is utilised to compare the calculated and observed data. In this case, the variable chosen was the hydraulic head. Afterwards, the hydraulic parameters were adjusted, and a new simulation was run to compare the resulting value of the objective function with the previous simulation. The purpose of this process is to optimise the resulting objective function to the smallest value (Anderson et al., 2015b). To calibrate the model to the steady-state, it was applied the tool Finite Element Parameter Estimation – FePEST. This tool is an adaptation of the well-know PEST (Doherty and Hunt, 2010) for the finite element. The FePEST automates the process of adjusting the parameters for calibration, substantially reducing the time necessary in the calibration step.

Three of the twelve observations wells with hydraulic heads measured were applied as boundary conditions. The remaining nine wells were used as observation values for the calibration. The groundwater model was then calibrated for two settings: one using values obtained from previous studies as bounds for the hydrogeological parameters, and other using values from the prediction interval derived from the empirical estimates of K-$S_i$. The technique of calibration through pilot points was used. The pilot points were determined by selecting wells randomly from the second specific capacity dataset. This was performed so it could be possible to use an existing value of the specific capacity index to determine the bounds for the parameters to be calibrated. Therefore, eight locations were selected while seeking to maintain a uniformly spatial distribution of the pilot points in the study area (Figure 5.2).

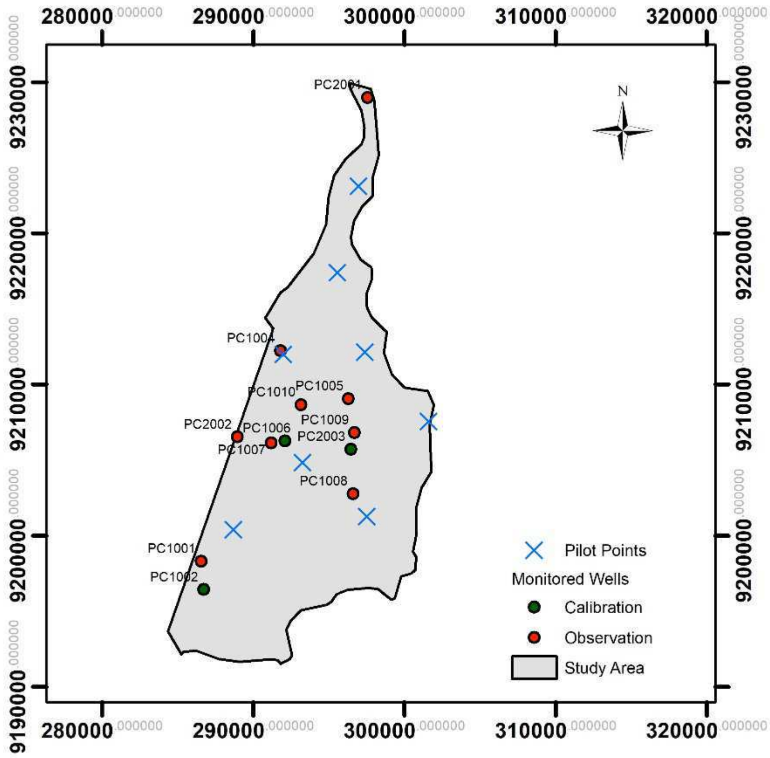


Figure 5.2 – Wells measured and pilot points in the study area

### 5.2.7 Assessment of the predictive uncertainty

The Monte Carlo Method has been widely applied for assessing the predictive uncertainty of the parameters of a groundwater model (Højberg and Refsgaard, 2005; Sepúlveda and Doherty, 2015). This method estimates a large number of sets of parameters following a probabilistic distribution function. Each simulation of the model running with a set of generated parameters is called realisation (Refsgaard et al., 2007). Afterwards, it is computed the predictions of the model using all the realisations estimated. Finally, the set of predicted values can be used to analyse the uncertainty of the model through statistical inferences. The uncertainty range of the hydraulic heads is the difference between the maximum and the minimum head calculated using all the Monte Carlo parameter realisations and the uncertainty standard deviation is the standard deviation calculated also using all the realisations (Sepúlveda and Doherty, 2015).

### 5.2.8 Spatialization of K

Cokriging is part of the suite of methods that composes the kriging interpolators. Cokriging is a multivariate interpolation method which estimates the main variable (in this case K), using another accessory variable (in this case $S_i$), given that there is a correlation between these two (Razack and Lasm, 2006). This method has a more reliable theoretical foundation because the correlation between these variables are determined through a cross semivariograms, in oppose to kriging with regression, whereas certain assumptions have to be made (Mace, 2001). The application of Cokriging needs two semivariograms, one for each variable, and cross semivariogram between the two variables. To validate the reproduction of the spatial variable predicted using Cokriging, a cross-validation technique is applied (Dalla Libera et al., 2017; Razack and Lasm, 2006). The geostatistical package included in the software ArcGis 10.2 was used to apply this method.

## 5.3 Results

### 5.3.1 Estimates of Hydraulic Conductivity

After analysing the pumping test data, 21 wells were selected in the confined region of the Beberibe formation. These wells had data available to the application of the Cooper-Jacob test, in order to determine the values of hydraulic conductivity and specific capacity index. The values of hydraulic conductivity varied from $4.43x10^{-2}$ m/d to 4.26 m/d, with a geometric mean of $4.79x10^{-1}$ m/d. The specific capacity index varied from $2.36x10^{-2}$ m/d to 1.08m/d, with a geometric mean of $2.12x10^{-1}$ m/d.

Afterwards, a log transformation of the pairs of data and a linear regression model were applied to determine the empirical relationship of hydraulic conductivity and specific capacity index for the steady-state. A linear correlation of logK – $\log S_i$ was determined with a correlation coefficient ($R^2$) of 0.69 with a 95% prediction interval. This interval spanned over one order of magnitude (Figure 5.3). The logarithmic transformation for the relationship can be presented as: $\log K = 0.9838(\log S_i) + 0.3426$

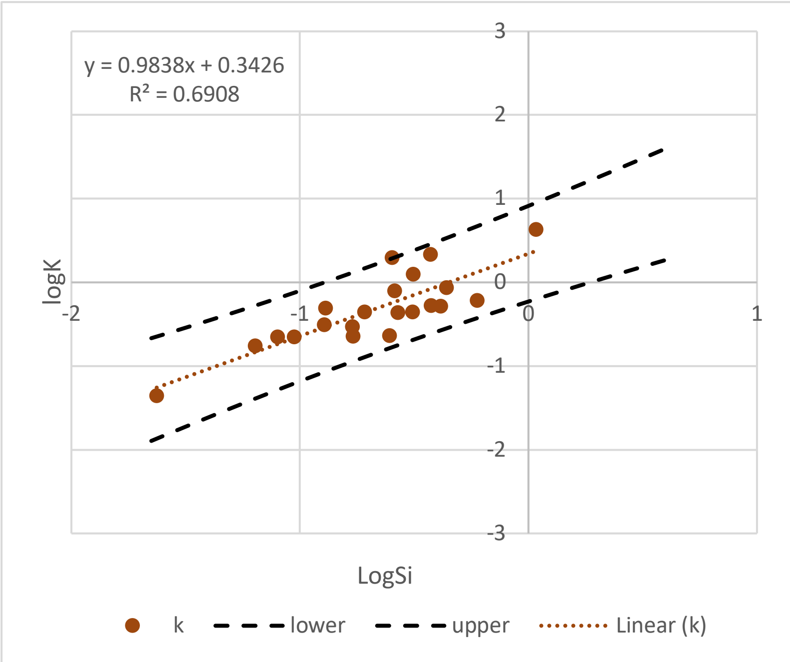

Figure 5.3 – Scatter plot with linear regression model

### 5.3.2 Numerical Model: Implementation and Calibration

A numerical model for the study area was implemented using the software FEFLOW. This model has one layer representing the Beberibe formation. The bottom is delimitated by the crystalline. At the west part of the system, the model is delimitated by the geological fault. Whereas in the east part, it is delimitated by the shoreline. The numeric model has 14935 elements and an average thickness of 280m. The thickness of the layer was interpolated using the data from 30 lithological profile and one cross-section available. The boundary condition at the southern border was considered as no flow, and it was assumed that this was a natural groundwater flow boundary. At the west, it was obtained from the heads of a larger groundwater model in the area. At the east, an interpolation based on the distance of the shoreline to the end of the confining formation, similar to the one used in Batista et al., (2011), was applied. Exploitation wells that belong to the water supply company (CAGEPA) were added, given that these were the only exploitation data available.

It is worth note that the region has a large number of clandestine exploitations wells which data are unknown and influences the values of hydraulic heads. Out of 12 wells measured, six presented negative values of potentiometric heads. Due to these influences, the observed values of hydraulic head ranged between 18.76 m.s.l to -10.70 m.s.l. In the first calibrated model (C1), the bounds of hydraulic conductivity varied from 0.25 to 12m/d. This range was determined from previous studies conducted in the study area (Batista et al., 2011; Braga et al., 2015; Costa et al., 2007). This bound was applied to the eight pilot points chosen. The correlation coefficient achieved was 0.91 with a root mean squared error (RMSE) of 5.08, mean absolute error (MAE) of 4.32m and Normalised Root Mean Square Error (NRMSE) of 17.25%. The scatter plot between observed and calculated heads shows in Figure 5.4. The second calibrated groundwater model (C2) applied values calculated from the prediction interval of the empirical relationship estimated as the hydraulic conductivity bounds for calibration. In each one of the eight pilot points, the lower and upper range was calculated from the specific capacity index in the location. The correlation coefficient was 0.929, with an RMSE of 4.615, MAE of 3.86m, and NRMSE of 15.65%. The scatter plot between observed and calculated heads is shown in Figure 5.5. Finally, the spatial distribution of the hydraulic conductivity calibrated for C1 and C2 are depicted in Figure 5.7 and Figure 5.6.

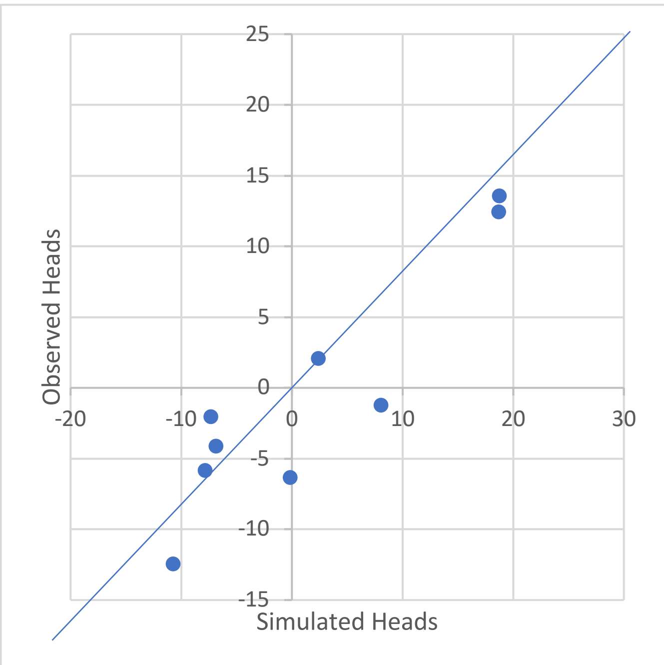


Figure 5.4 – Scatter plot calibration C1

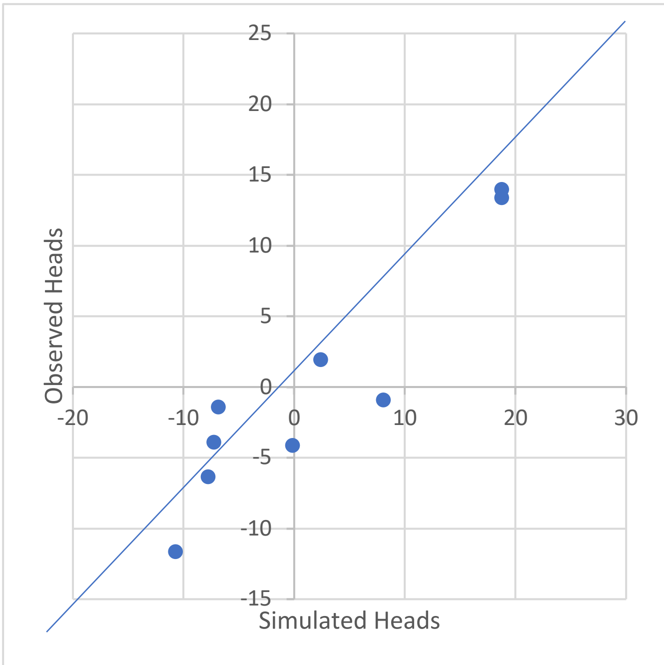


Figure 5.5 – Scatter plot of calibration C2

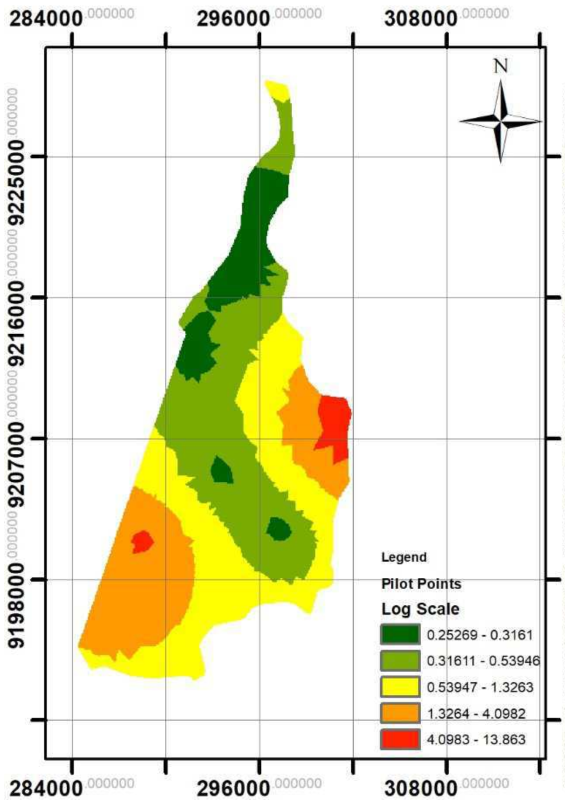


Figure 5.7 – Hydraulic conductivity calibrated in C1

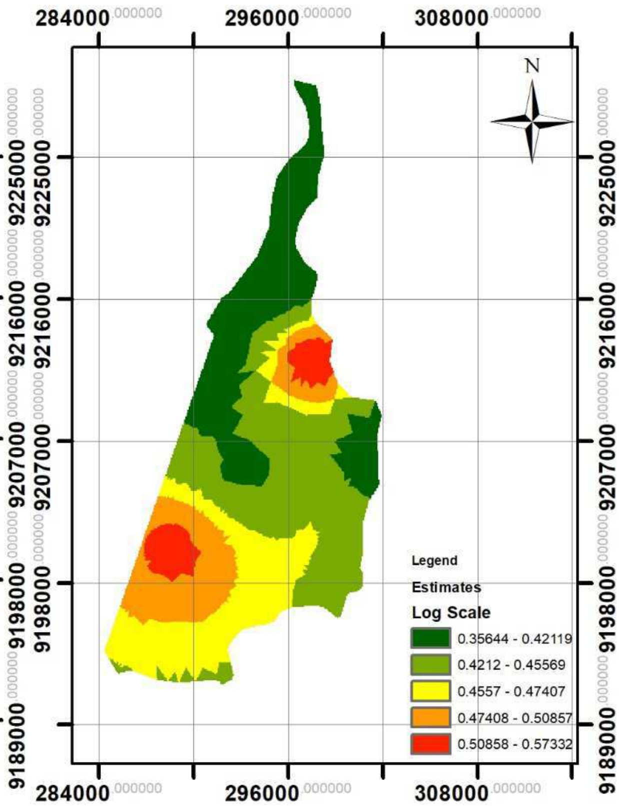


Figure 5.6 – Hydraulic conductivity calibrated in C2

### 5.3.3 Assessment of the predictive uncertainty

The Monte Carlo Method was applied for both calibrated models. Two hundred sets of parameters were generated using log-normal distribution. The determination of the number of realisations was evaluated by the cumulative absolute mean difference (Figure 5.8). It can be noted that after 100 realisations, the uncertainty of the model can be adequately characterised given that stability is reached, and the cumulative absolute mean converges to a value.

The uncertainty range of the hydraulic heads calculated using all the 200 Monte Carlo parameter realisations for the first calibrated model (C1) in the nodes with observation wells varied between 1.80m to 16.95m. The uncertainty standard deviation for C1 varied from 0.31m to 2.81m. For the C2, the uncertainty range varied between 0.20m to 2.61m with an uncertainty standard deviation varying from 0.07m to 0.57m. A boxplot summarising the calculated heads of the MC simulations for C1 and C2 is presented in Figure 5.9 and Figure 5.10.

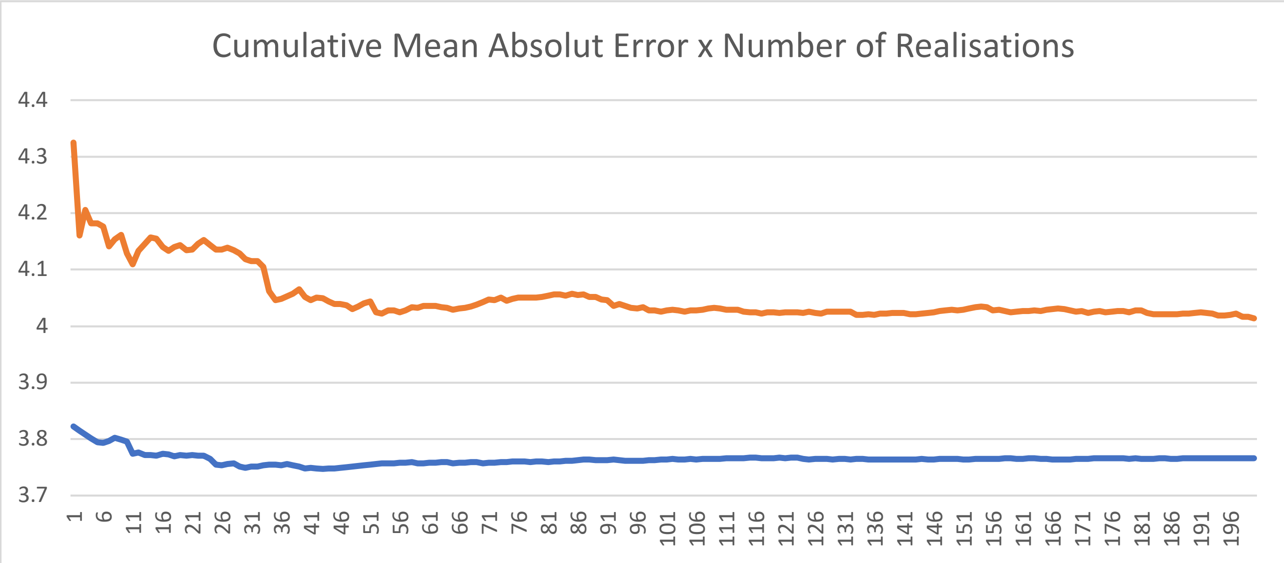


Figure 5.8 – Cumulative mean absolute error for C1 (orange) and C2 (blue)

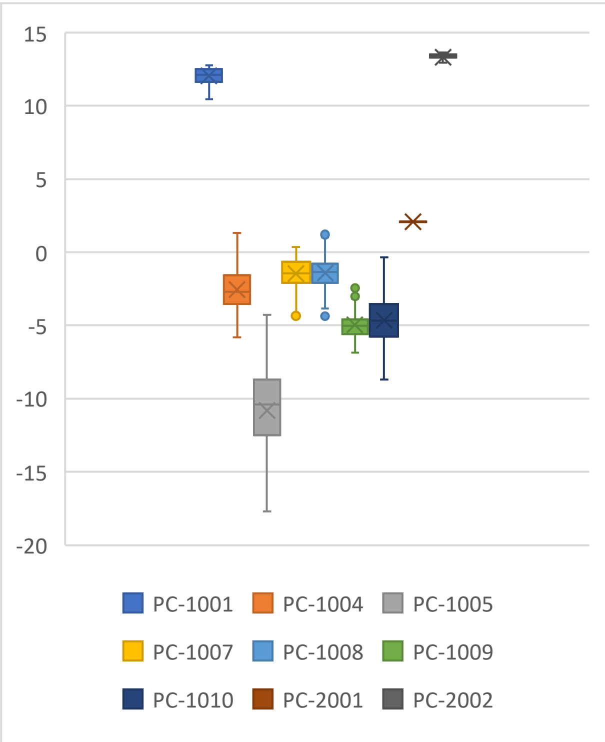


Figure 5.9 – Boxplot of calculated head for C1

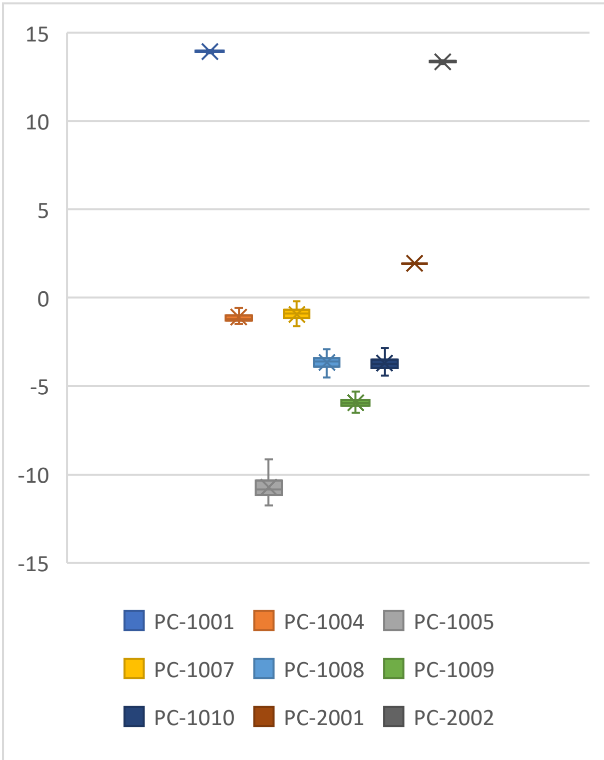


Figure 5.10 - Boxplot of calculated head for C2

### 5.3.4 Spatialization of K

Hydraulic conductivity data were estimated through empirical relationship using the available data of specific capacity index for 121 points. The available data of specific capacity index varied from $4.95 \times 10^{-2}$ m/d to 1.04m/d, with a geometric mean of $2.94 \times 10^{-1}$ m/d. The estimated values of K obtained from the relationship varied from $1.1 \times 10^{-1}$ m/d to 2.30 m/d, with a geometric mean of $6.53 \times 10^{-1}$ m/d.

These pair of data were then analysed according to its distribution. The data did not have a normal distribution, which is one of the requirements for the application of kriging methods. Therefore, a log-transformation was applied to both sets of data, in order to apply the Cokriging with lower error (Aboufirassi and Mariño, 1984). Afterwards, semivariograms and covariance using the exponential model were obtained. Finally, cross-validation of the cokriged data was conducted. The MAE of the predictions was 0.18m/d with an error variance of 0.05 $m^2/d^2$, and a correlation coefficient of 0.89. A scatter plot between the predicted and estimated values of K is presented in Figure 5.11. A map with the spatialization of the predicted value of hydraulic conductivity is presented in Figure 5.12.

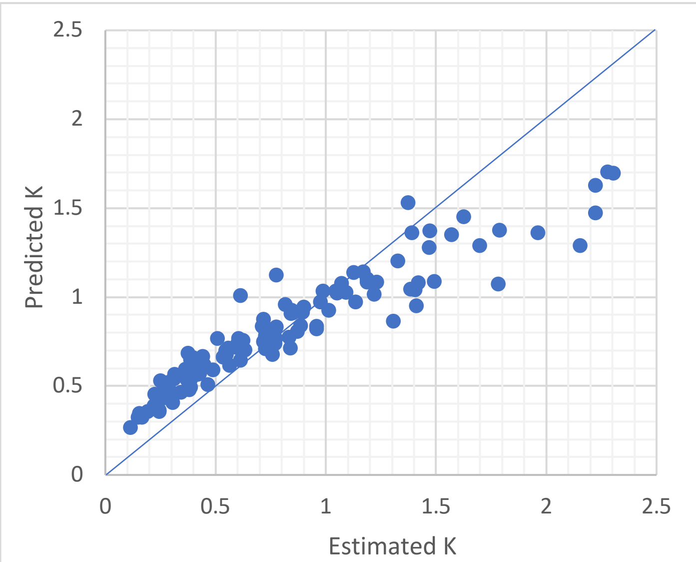


Figure 5.11 – Scatter plot of predicted K (cokriging) and Estimated K (empirical relationship)

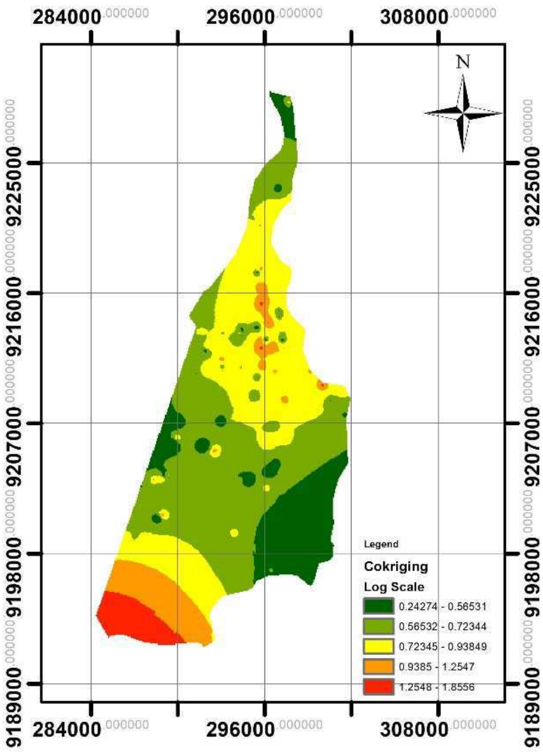


Figure 5.12 – Map of predicted K - Cokriging

## 5.4 Discussion

The empirical relationship between the hydraulic conductivity and the specific capacity index presented a good correlation coefficient and linear regression model similar to other studies conducted in different geological settings (Priebe et al., 2018; Richard et al., 2016;

Verbovšek, 2008) and sandstones (Jalludin and Razack, 2004; Mace, 2001). Certain factors could not be taken into account in the determination of the relationship due to the lack of data. These factors include the well loss correction, the heterogeneity of the aquifer, and the identification of the equilibrium between pumping rate and the late-time drawdown (Baye et al., 2013; Hsu and Chou, 2019; Richard et al., 2016; Srivastav et al., 2007). However, the correlation coefficient found is still within the range achieved by other studies (Mace, 2001). Equation 2 can be rewritten as a power function, and it is generally presented in this form in other studies: $K = 2.20(S_i)^{0.9838}$. Therefore, this relationship can be used to generate new information in the study area, given that this estimate is a site-specific relationship (Fontenele et al., 2014). This method allowed to increase the number of estimates from 21 to 121. A considerably higher number, more than five-fold. This resulted in a density of K values of 1 per 2.47 $km^2$ instead of 1 per 14.28 $km^2$. This represents a significant increase in the amount of information for the confined aquifer.

The lack of data is an obstacle to the calibration of any groundwater model (Alfaro et al., 2017; Candela et al., 2014). Furthermore, the region exhibits a high level of exploitation with many wells not registered, leading to a number of interferences in the behaviour of the observed heads. These interferences make the calibration process even more problematic. Despite these challenges, a satisfactory calibration was achieved. The calibration using empirical relationship (C2) presented better results than the calibration using literature values (C1). A reduction of approximately 10% in both MAE and RMSE and a higher correlation coefficient form C2 to C1 shows the validity of applying this estimate to improve the groundwater model calibration. Consequently, more reliable information can be generated using this tool. Pool et al., (2015) achieved a better calibration applying values of T derived from the Theis interpretation of pumping tests then applying values obtained of T from a relationship with SC. However, to obtain this second set of T values, it was used a relationship from another location, not a site-specific derived one (Pool et al., 2015; Razack and Huntley, 1991). Therefore, this set of values did not present a significant correlation. We argue here that in order to be applied has information in the calibration process, the relationship has to be locally derived, from a local set of data, given that this relationship is site-specific (Hsu and Chou, 2019; Priebe et al., 2018).

An improvement could also be perceived regarding the uncertainty of the information generated. In this study, the use of the prediction intervals of the relationship as parameters' bounds to the calibration resulted in a considerable reduction in both uncertainty range and uncertainty standard deviation. For example, the observation PC-1005 had a reduction in the uncertainty range from 16.95m (C1) to 2.61m (C2). This reduction is mostly explained due to the reduction in the bounds in which the parameters can vary (Refsgaard et al., 2012). Narrowing these limits can help the calibration process to converge to a more stable solution; evidence of that is presented in figure 5, where it can be seen that the stability of cumulative mean absolute error for C2 converged with a smaller number of realisations than C1. These findings go in line with what has been concluded by Hunt et al., (2019): how the parameters are handled can increase the performance of the groundwater model, and it is more important than the number of parameters itself.

The application of simple linear regression estimates using readily available data can substantially aid the calibration process leading to a more precise groundwater model. To effectively communicate this information, it is of paramount importance the inclusion of uncertainty analysis, given that these empirical estimates also have an inherent uncertainty (Neuendorf et al., 2018; Pezij et al., 2019). Although widely studied by the scientific community, the uncertainty analysis is still disregarded by water practitioners (Delottier et al., 2017). Among the reasons is the lack of relevant and reliable datasets (Delottier et al., 2017). The technique applied here can help bridge this gap by enhancing the dataset available while reducing the parameter uncertainty. Presenting uncertainty trough terms such as the ones used in Monte Carlo analysis can increase the understanding of forecast for decision-makers or stakeholders; therefore, the information provided from the model can be more readily accepted (Hunt, 2017).

Furthermore, the information obtained through estimates of hydraulic conductivity from the specific capacity index can also be spatialized through the application of geostatistical methods such as cokriging. A good fit between the predicted values using cokriging and the estimates from empirical relationships had been identified when Transmissivity (T) and Specific Capacity were used (Aboufirassi and Mariño, 1984; Razack and Lasm, 2006). A moderate fit was found by Fontenele et al. (2014), and a weak spatial correlation was found by Rotzoll and El-Kadi, (2008) using hydraulic conductivity (K) and specific capacity but applying ordinary kriging. In this paper, a good correlation was found using hydraulic conductivity and the specific capacity index. The importance of using K instead of T lies in that K is the input for groundwater models, and it can be used to compare different regions in the same aquifer, given that K does not depend on the aquifer thickness (Rotzoll and El-Kadi, 2008; Verbovšek, 2008). Estimates of T and further conversion in K requires knowledge on the geological characteristics of the aquifer (Priebe et al., 2018). Sometimes this can be challenging due to scarcity of data and information. On the other hand, the determination of the specific capacity index requires only the saturated thickness at the location where the data is obtained and can be used to inform the aquifer productivity without influence from the aquifer-thickness variation (Mace, 2001).

Finally, a comparison between the information from the map with calibrated values of K for C1 (Figure 5.7) and C2 (Figure 5.6), and the map with the spatialization of K using cokriging (Figure 5.12) presents some resemblances. Comparing the two maps obtained with the calibration of the model, it could be identified that the spatial pattern of hydraulic conductivity in certain regions, even though the values are different in more than one order of magnitude. The central and north regions presented the lowest values of K, while the largest was found in the southwest. These same patterns are also found when compared with the map obtained from the Cokriging method. While the maps from calibration used eight pilot points with values of $S_i$, the map from Cokriging used all 121 data of $S_i$. Hence, this information regarding the spatial pattern was kept using different methods and a different set of data. Therefore, it was consistently identified within the aquifer zones with higher and lower hydraulic conductivity. This pattern of hydraulic conductivity occurs due to the heterogeneity of the aquifer and the scale investigate with the head observations and pair of data used for determining the estimate. Woodward et al. (2016) could not obtain this information using groundwater flow modelling with high-resolution head data and slug test. In the present paper, the information was obtained using a more straightforward approach. A representation of this knowledge obtained from the combination of the information generated through the numerical model and geostatistic can be depicted through zoning of the system. Overlaying the map of C2 (given that this had the best results from calibration) with the map of the Cokriging application using normalised values can depict the zones with higher and lower hydraulic conductivity (Figure 5.13). This zoning can provide information for groundwater management. Regions with higher hydraulic conductivity in the confined aquifer have a wider radius of influence (Chesnaux, 2018). Thus, management measures can be determined according to this knowledge.

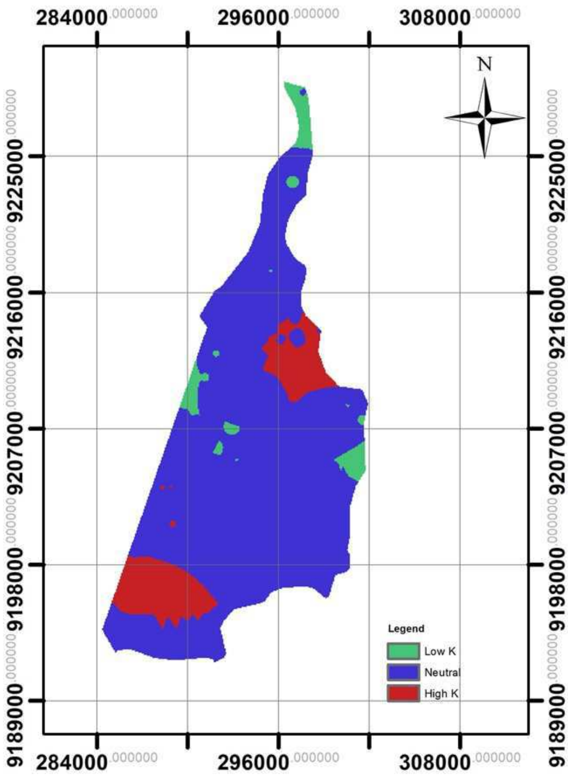


Figure 5.13 – Zones with high and low values of K

## 5.5 Conclusion

In this study, the knowledge of a groundwater system was improved using an empirical relationship between the hydraulic conductivity and the specific capacity index. Through this relationship, estimates of hydraulic conductivity were obtained. These estimates were applied to improve the calibration of a groundwater model and were spatialized for the study area. Finally, regions with higher and lower hydraulic conductivity were identified.

The calibrated model using the estimates had a considerable reduction in the uncertainty; therefore, it can provide more reliable information. This application can facilitate the application of groundwater model in scarce data areas. The estimates can generate new information to be applied in the calibration process. These estimates were also able to be spatialized using Cokriging with a satisfactory validation. Through the combination of this different information obtained from the estimates, it was possible to increase the existing knowledge about the groundwater system.

Further research could concentrate on the application of model-data interaction (MDI) to enhance the knowledge about the groundwater system, applying these estimates in a two-way process between the database and the models to both refine the numerical or geostatistical model and to identify areas with more scarcity of data. MDI methods, such as data fusion, can be applied to analyse more specific issues in groundwater systems, such as the heterogeneity. The increase in knowledge with these methods might be possible by a fusion of the data with expert knowledge.

# Chapter Six – Managing groundwater at the landscape scale: exploring the spatial and temporal dynamics

## 6.1 Introduction

The connection between groundwater and land use has been widely recognised (Han et al., 2017). Differing types of land-use cause different pressures on groundwater systems (e.g., alterations in recharge due to urbanisation, contamination of aquifers due to agricultural runoff) (Delkash et al., 2018; Zipper et al., 2017). As a result, groundwater management and land use planning need to be better integrated, especially regarding biophysical, socio-economic aspects, and spatial-temporal dynamics (Foster, 2018).

The management of groundwater system has usually been conducted looking to the whole hydrogeological unit (Zhang et al., 2019) or the intersection with a catchment (Remondi et al., 2016; Yu et al., 2018). Therefore, those analyses have been led without subdividing the system into more homogenous regions under different aspects. When such a perspective is sought, there has been applied hydrological sub-catchments as smaller units (Sahoo et al., 2018). Few studies analyse land use with conjunction with other aspects to define management units for groundwater, for example, most of the publications only include land use aspects for the determination of protections or buffer zones, but such zones are not used as a spatial basis for management (Mogaji et al., 2016; Neshat and Pradhan, 2017). An advance was made by the Sustainable Groundwater Management Act in California (USA) that applies the concept of groundwater sub-basins, but they are defined based on geologic factors, hydraulic considerations, or institutional boundaries (Winter et al., 2018).

Regarding the temporal dynamics in groundwater systems, the most common aspect addressed is the influence of land uses changes in the temporal variation of the recharge or discharge (Hugman et al., 2015; Mair et al., 2013). The temporal dynamics of the recharge refers to its seasonal variation due to rainfall patterns (Braga et al., 2015) or inflow from rivers or lakes (Ott and Uhlenbrook, 2010), while for the discharge, these temporal dynamics include the outflow as baseflow to springers, rivers, wetlands (Stigter et al., 2014) or as submarine groundwater discharge in coastal aquifers (Burnett et al., 2006). The importance of the temporal dynamics of the groundwater system has also been analysed taking into account the sustainability of the aquifer. For this, analyses have addressed the timescale applied for calculation of the sustainable yield, which determines the maximum amount to be withdrawn from the aquifer (Abrishamchi et al., 2020; Hugman et al., 2013). Other aspects, such as the time-lag and the response time of aquifers, have only been recently addressed (Carr and Simpson, 2018; Currell et al., 2016). However, its applicability in complex groundwater systems and its integration with the spatial dynamics has yet not been investigated.

Landscape Scale Planning can be an appropriate approach to support the integration between groundwater management and land use because it takes into account not only the distinctive features of the landscape but also its natural, cultural and socio-economic features (Selman, 2006). Hence, there is a potential for landscapes to be applied under multifunctional approaches aiding the data collection, policy delivery and partnership-based coalitions from a spatial and temporal perspective (Selman, 2009). In Chapter 2, this integration was investigated under the Landscape Scale Planning approach. One of the results was the proposition of guidelines for groundwater management based on such an approach. Central to the application of these perspectives is the concept of landscape units. As previously explained, landscape units are defined as integrative units that have more internal resemblance compared with the surrounding regions (Selman, 2006). Besides, the landscape units can also be associated with the sense of place; therefore, fostering social learning and land care (Selman, 2009). However, environmental delimitations such as topographical basin and groundwater boundaries do not coincide with administrative or institutional limits. The lack of integration of these sectors leads to a fragmentation of the holistic aspect of the landscape units (Antrop and Van Eetvelde, 2017).

Therefore, this chapter seeks to analyse to what extent spatial landscape units can be integrated to provide a whole-of-landscape analysis to support groundwater management, and to what extent the temporal dynamics of the groundwater system can be incorporated into the analysis of groundwater management.

## 6.2 Spatial Dimension and Delimitation of Landscape Units

### 6.2.1 Boundaries for groundwater management

The historical approach guiding groundwater management was focused on defining the limits for exploitation within the aquifer. First, the concept of safe yield was commonly used (Lee, 1914; Meinzer, 1923; Theis, 1940); this concept was defined by Todd (1959) as "the amount of water that can be withdrawn from a groundwater basin annually without producing and undesirable result". However, this concept was widely criticised as incomplete and has now fallen in disuse (Alley et al., 2002; Alley and Leake, 2004; Freeze and Cherry, 1979; Todd and Mays, 2005). As an alternative, the concept of sustainable yield raised as a determined pumping rate that would not cause unacceptable social, economic or environmental consequences (Alley and Leake, 2004; Zhou, 2009). Therefore, it includes the consideration of the reduction of discharge to springs, rivers, and wetlands, as well as the insurance of the maintenance of the water rights for other users.

Both of these concepts were thought to take into account a groundwater basin. A groundwater basin boundary can be determined by the surface and groundwater inflows and outflow of the hydrogeological formation, these inflow and outflows being based on flow lines derived from topographic aspects (Demiroglu, 2017; Toth, 1963). However, this relationship is not always true, and the boundaries can vary from small ridges to continental scales (Demiroglu, 2017). As a consequence, the groundwater basin can end up being the whole aquifer itself.

In contrast, the approach to groundwater management interventions has been traditionally focused on the well. The decline of the water table or the well-yield is generally the indicator that makes the stakeholders call for groundwater management interventions (Foster and Chilton, 2017). Among the most common interventions are the trigger levels, measured in wells, to regulate the use in that location, and the buffer zones surrounding determined sites and controlled by wells (Noorduijn et al., 2018). Although both these interventions are valid, they are mainly focused on a small location in the aquifer.

Foster and Chilton (2017) call for a shift in the approaches for groundwater management. From targets focusing on the wells and springs to the whole aquifer system as well as its connection with land use. This means that sustainability and safeguard must not be addressed only at the well but throughout the whole aquifer. The shift involves, and can be facilitated by, the application of the Integrated Groundwater Management and the principles of groundwater governance (FAO, 2016; Foster and Chilton, 2017). Aiming at this direction, some effort has been made, such as the Water Framework Directive and Groundwater Directive in Europe (European Commission, 2000; Rejman, 2007), the Sustainable Groundwater Management Act in California (Dumas, 2019) and the Global Framework for Action (FAO, 2016).

The Water Framework Directive (WFD) determines as a management unit for its implementation the "bodies of groundwater". The definition of the body of groundwater is ambiguous and insufficient to allow a precise application (Sánchez et al., 2009). Body of groundwater is defined as "distinct volume of groundwater within an aquifer or aquifers" (European Commission, 2000, p. 6). In order to clarify specific issues, the European Commission published horizontal guidance to the identification of water bodies (European Commission, 2003). In conjunction, both documents determine the steps for delimitation of bodies of groundwater. The first step is the identification of the aquifers. The second step is the delineation of the bodies of groundwater following these criteria: i) the limits of groundwater bodies should be based on the geological boundaries of groundwater flow; ii) bodies of groundwater must have only one chemical and one quantitative status; iii) if no data is available to the determine the chemical or quantitative status, an analysis of pressures and impacts (according to Article 5 of the WFD) can be applied as status; and iv) areas with high risk which do not reach the objectives of the WFD must be delineated as a separate body of groundwater (European Commission, 2003, 2000; Sánchez et al., 2009). A similar concept to the sustainable yield is defined in the WFD as a guide to determine the "available groundwater resource" – the indicator of qualitative status. The available groundwater resource is "the long-term annual average rate of overall recharge of the body of groundwater less the long-term annual rate of flow required to achieve the ecological quality objectives for associated surface waters specified under Article 4, to avoid any significant diminution in the ecological status of such waters and to avoid any significant damage to associated terrestrial ecosystems" (European Commission, 2000, p. 7). Therefore, such an indicator should be calculated for each body of groundwater and, management interventions can be applied supported by this data.

Recently, after many years of droughts and unmanaged groundwater exploitation, the state of California (USA) has decided to establish a framework for sustainability and management of groundwater resources. The Sustainable Groundwater Management Act (SGMA) became in force at the beginning of 2015. This framework applied a bottom-up approach where the means and method to be applied in the management of groundwater resources are delegated to the local agencies called Groundwater Sustainability Agencies (GSA) (Dumas, 2019). Each of the GSAs has the responsibility to comply with the SMGA directives. These directives refer to quantitively and qualitative criteria at a local and the groundwater basin level (Dumas, 2019). The groundwater basins are the management units defined by the SGMA in which the sustainability indices are monitored and evaluated (Thomas, 2019). These groundwater basins were defined primarily by using geological contacts or hydrogeological barriers. Among these barriers are impermeable bedrock, constrictions in permeable materials, faults, low permeability zones, and groundwater divides (California Department of Water Resources, 2003). A further delimitation in sub-basin is also possible.

This division uses geologic and hydrologic aspects, but the more commonly applied are institutional boundaries. These sub-basins are refined of a more extensive basin with the purpose to aid the collection and analysis of data (California Department of Water Resources, 2003). Therefore, although the SGMA attempts to implement an integrated approach to groundwater management, the definition of the boundaries for groundwater management does not include other factors beyond the physical characteristics of the system, such as land use.

The ASUB project, in Brazil, brought a new perspective by proposing three different levels of management: local, regional and basin (ASUB, 2010). The local level refers to the well and the nearby consequence of its exploitation. The regional refers to an intermediate level where the groundwater system is divided into management units; this could be related but is not the same to the groundwater basins in the SGMA or the bodies of groundwater from the WFD. Finally, the basin level refers to the whole aquifer system or the surface river basin.

The division of the groundwater system into management units was conducted in a bottom-up approach, through the consultation of professionals or experts in the fields of hydrogeology and water management, field trips to the study area, and the accumulated knowledge of researchers and water users. Finally, the process for delimitation of management units followed several criteria: topographic levels, geological faults, hydrogeological formations and groundwater demand in the zone (ASUB, 2010; Costa, 2009). This process was applied for the intersection of the Paraíba – Pernambuco Sedimentary Basin with the Paraíba River Basin; therefore, it comprises part of the study area of this thesis. In total, the application of this method resulted in seven management zones.

The ASUB project proposition presents an advance if compared with the WFD guidelines, given that it does not restrict the definition of the management unit to an area with one chemical or qualitative status. A region of the aquifer might have a similar status but very different demands for water use and, as a consequence, requiring different interventions to the management. When compared with the SGMA, even though the ASUB project predates this framework, it can be seen that the ASUB preposition is more comprehensive than the groundwater basin concept proposed by the SGMA. Both proposals include as an initial criterion the geomorphology and hydrogeological factors; however, while the former includes the water demand of the users to refine the delimitation, the latter applies institutional boundaries as a further criterion. The institutional boundaries (such as municipal limits) can aid the process of decision-making and implementation of strategies to groundwater management, but it is widely known that the environmental and water issues can extend beyond the municipal limits (Albrecht et al., 2017).

### 6.2.2 The conceptualisation of Landscape Units

The Landscape Scale Planning framework introduces the concept of *landscape units* that can be applied to delimit boundaries for, but not exclusively to, groundwater management. Therefore, such landscape units could be applied as management units. As defined in Chapter 2, landscape units are integrative units that have more internal resemblance compared with the surrounding regions. These landscape units can be achieved by sorting the landscape into homogeneous units according to biophysical, cultural, and socio-economic attributes (Selman, 2009). The alignment between socio-economic practices, environmental capacity and cultural acceptance can lead to a virtuous cycle, increasing the sustainability of socio-ecological systems (Selman, 2006). Therefore, the intersection of the environmental, economic, and social process within landscape units gives a multifunctional character to these units (Selman, 2009). Furthermore, given that landscapes have physical and informational flows, the landscape units can act as a locus for data gathering and analysis to integrated multi-attribute spatial datasets into information for the planners, managers or practitioners.

There is a significant amount of physical landscapes variables that are associated or correlated with the environmental variables; so, a first step is to analyse the presence of these variables and their attributes – such as geology, topography, and hydrology. These variables are correlated with the biophysical aspects of landscapes. These criteria have been traditionally used to define management units. They were used by the WFD, SGMA and the ASUB project.

Even though the biophysical aspects play an important role determining the natural boundaries of any system, the landscape units should also reflect the cultural aspects of the system, such as the sense of place and human identification with the landscape itself (Brunckhorst, 2013). These terms refer to the emotional and physical attachment that individuals have with a determined place and reflects the way that the community (stakeholders) accepts and uses the landscape (Seddon, 1972; Tapsuwan et al., 2011). Furthermore, the acceptance of the community has been demonstrated to be one of the key factors to a successful planning process (Tapsuwan et al., 2011). The main contributor to the experience of place is the spatial planning since it can shape part of the physical components of the space (Žlender and Gemin, 2020); therefore, the resulting land uses from the spatial planning can be correlated to the acceptance of the community to spatial changes and act as a proxy of cultural aspects (Simensen et al., 2018).

Socio-economic aspects are one of the main drivers affecting socio-ecological systems. The social aspect of the landscape refers to the people living in the system and the stakeholders with their demands and needs for consumptions, while the economic aspect refers to the production and trading processes of goods and services that are deeply embedded in the functioning of society; these processes require water and alter the land configuration (Selman, 2006). Therefore, the socio-economic aspects are a fundamental part of cultural landscapes. Three criteria can be used to represent the homogeneity of these aspects in a landscape: i) the water uses; ii) the land use; and iii) census sectors. The water uses involve the necessity of the stakeholders from the groundwater system. This can be estimated by the number of wells in a determined region. This criterion was used by the ASUB project to delimitate the management zones. The land use can be defined as the way that society uses and alters the biophysical cover (Fritz et al., 2017); besides, it can represent cultural aspects as shown earlier. The census sectors represent the smallest homogeneous spatial continuous area with populational information situated in an urban or rural sector within a municipality limit; thus, it condensates information regarding the characteristics of those living in such areas.

Therefore, to determine landscape units taking into account the homogeneity of multiple aspects, the following criteria and sequential steps are here proposed for application: i) topography/hydrology; ii) hydrogeological delimitations; iii) water use; iv) land use; v) census sectors. The application of these criteria did not follow a unidirectional process but an iterative approach. Meaning that, in order to further refine the delimitation of landscape units, some criteria have to be revisited. This occurs due to an expected mismatch between the different layers of information applied in the process. Hence, an iterative approach aids the process of refinement.

### 6.2.3 Topography/Hydrology

The first criteria applied was the topographical delimitation based on hydrological characteristics. River basins are a traditional approach to delimitate management units in regional aquifers, given that these aquifers usually contain more than one river basin. Another point is that the primary sources of recharge to an aquifer, precipitation and the rivers, are concentrated within a river basin. Furthermore, the potentiometric boundaries of a groundwater system usually follow the topographical limits of a large river basin, especially in sedimentary aquifers.

In the study area, the groundwater system comprises two river basins; therefore, for the first delimitation of the system, it was applied the topographical limit of the river basins. Hence, part of the aquifer was divided by the Paraiba River and the topographical limits of the basin, and another part consisted of the Gramame River basin itself (Figure 6.1).

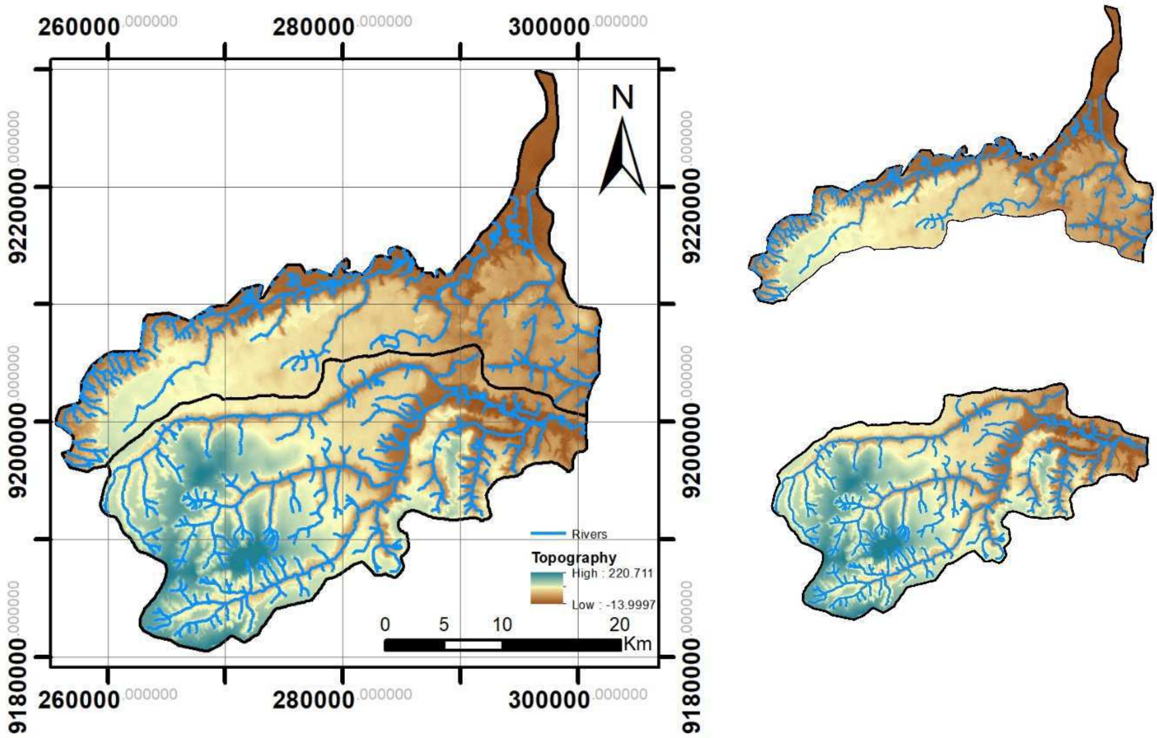


Figure 6.1 - Division according to the topographic/hydrological criterion

### 6.2.4 Hydrogeological delimitations

The groundwater flow occurs following the hydrogeological configuration of the aquifer. Thus, the type of formations, the hydraulic classification – if the system is confined or phreatic, and the geological faults. These aspects, combined with the hydrological ones, define most of the behaviour of the groundwater system, such as the recharge and discharge areas, the direction of flow and distribution of potentiometric heads.

The study area presents two different subsystems - one phreatic and one confined. The first division, according to the hydrogeological aspect, followed this aspect. The delimitation between the phreatic subsystem and the ones with the phreatic and confined formations happens in the geological fault (dashed green line – Figure 6.2). Another hydrogeological delimitation occurs at the Itabaiana geological fault (continuous red line – Figure 6.2) until the fault of Tibiri River (continuous blue line – Figure 6.2). This fault delimitates the horst of

Paraíba River, and the bedrock in this region is elevated from the rest of the aquifer. Thus, the alluvial deposits from the Paraiba river are situated directly on top of the bedrock, not on top of the Barreiras formation as it happens in other regions. The Itabaina fault extends until the western portion of the aquifer where it is located a granitic outcrop. This impermeable outcrop separates the alluvial deposits of Paraiba River from the Barreiras formations; therefore, the region delimitated by the Itabaina fault until the Tibiri River fault and the Paraiba River was adopted as another hydrogeological delimitation. East of the Tibiri river fault, the alluvial deposits from the Paraíba river still presents the same configuration (situated on top of the bedrock); however, there are two important lateral recharge contributions from the Tibiri river and the Sanhauhá River. Furthermore, these rivers have their alluvial deposits and are situated on top of the Barreiras formation. Because of this, it was decided to delimitate the alluvial deposits in that location. Therefore, the study area was divided into six hydrogeological units (HGU) according to the characteristics explained above (Figure 6.2).

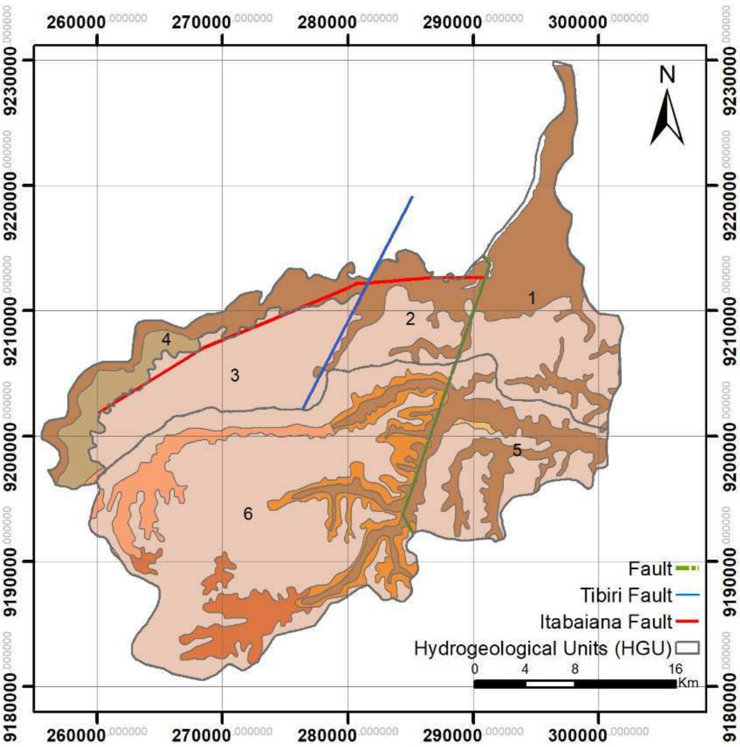


Figure 6.2 - Division according to HGU

### 6.2.5 Water Uses

The current water withdrawal is a clear indicator of the intensity of exploitation. The spatial distribution of the wells exploited can be used to identify zones of high concentration that might cause a saltwater upconing, saline intrusion and excessive drawdown of the water level. Therefore, the inclusion of this criterion to determine the management units can aid the implementation of groundwater management measures. In the ASUB project, this was one of the criteria adopted. Given that there are several clandestine wells, which locations are not identified, the groundwater rights conceded can be used as an indicator.

Figure 6.3 depicts the spatial distribution of groundwater rights for the study area. There is a high concentration of groundwater rights in the coastal area, northeast of the aquifer. The overexploitation in this area can lead to saltwater intrusion. As explained before, two subsystems are present in this region. On the one hand, the phreatic aquifer is less explored, and it is more susceptible to saline intrusion, on the other hand, the confined aquifer is less susceptible to saline intrusion due to the end of the Gramame formation situates far from the coast, but much more exploited than the phreatic aquifer. Another region with a high amount of groundwater rights is situated near the geological fault (dashed green line – Figure 6.3). Attention must be drawn to this region because this is where the recharge to the confined aquifer occurs. High exploitation here reduces the water availability for the rest of the confined aquifer. As a consequence, it can affect the sustainability of the whole confined subsystem. Finally, as the largest concentration of water rights are situated within the division so far defined, no further delimitation was conducted at this point.

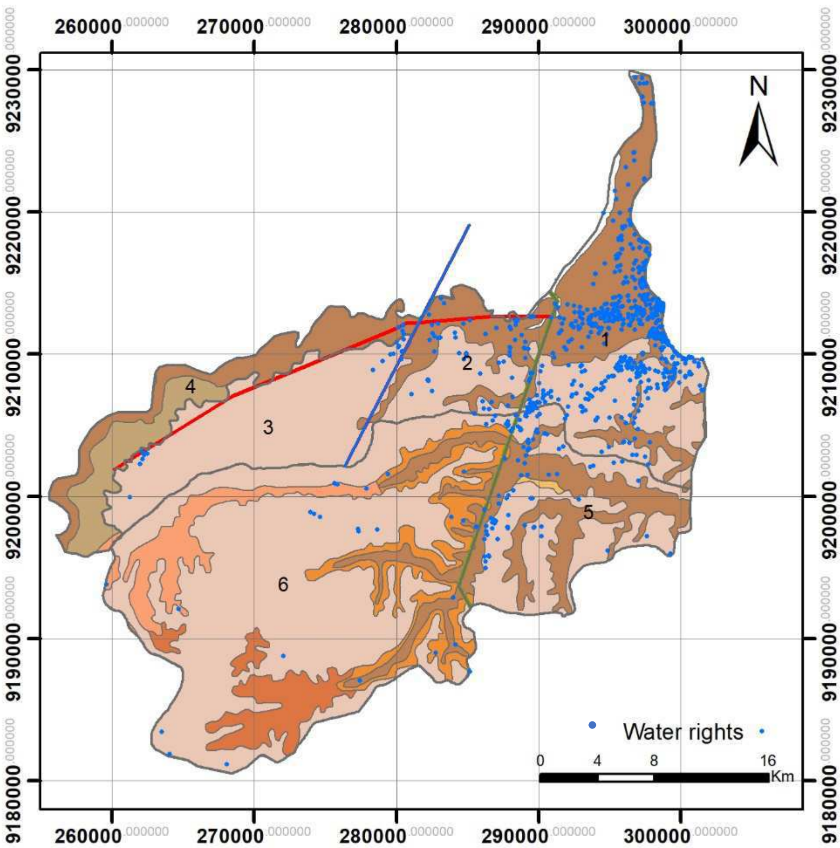


Figure 6.3 - Water rights in the study area and HGU

### 6.2.6 Typologies of land use

Land uses are one of the main drivers affecting groundwater resources and its integration with groundwater management has for long been defended. Both socio-economic and cultural aspects of landscapes are related to the land uses. As a consequence, it has a direct effect on water rights, given that residential, agricultural and industrial uses have different requirements and demands. Furthermore, land uses are also related to the contamination and pollution of groundwater. Agricultural land uses can have more contamination due to nitrate leaching; urban land uses due to sewage system leaking. In contrast, natural land use, such as

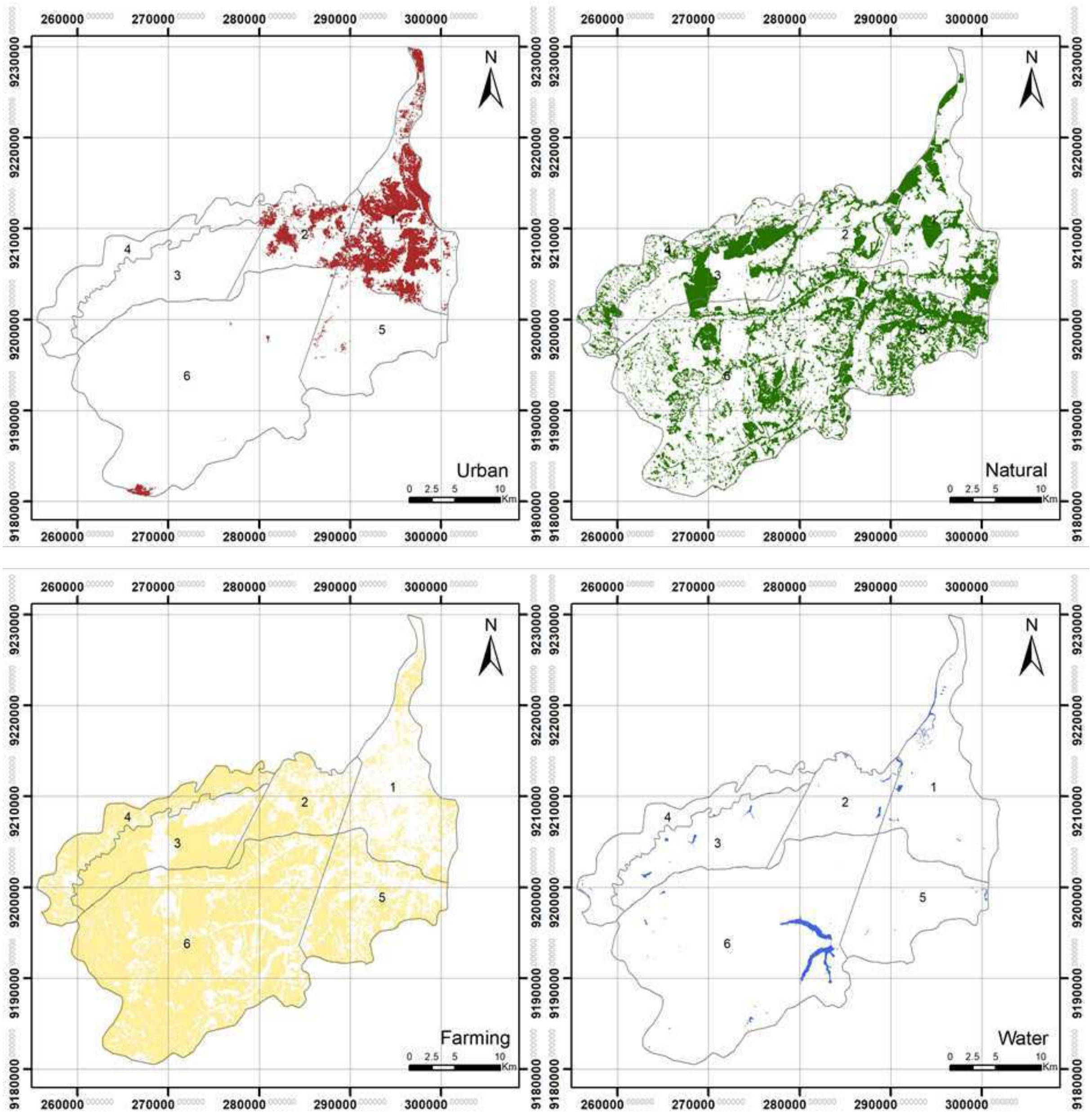


Figure 6.4 - Typologies of Land Use

forest, can have a positive effect on both qualitative and quantitative aspect of the groundwater resources.

For the delimitation of the landscape units, the diverse land uses existing in the study area were divided into four typologies: natural (forest and mangroves), farming (crops and pasture), urban, and water bodies (Figure 6.4). Coincidentally, the urban land uses are majority concentrated in one region of the aquifer. This region includes the confined part of the aquifer in the Paraiba River basin and the phreatic part of the aquifer in the same river basin, located between the geological fault and the Tibiri river fault. Natural land uses are distributed mainly following the river courses, although some patches can be identified. The Gramame-Mamuaba Dam is the main contributor to the water type of land uses, but some other small dams can also be identified. Farming uses are distributed throughout the study area and cover most of the region.

### 6.2.7 Census sector

Census sectors, or census tracts, census areas or census districts, are the smallest division units, generally within a municipality. These sectors are defined to conduct a census. Therefore, in each of these sectors, demographic data are collected. These census sectors are delimitated to keep approximately the same number of households; hence it varies in size. Urban areas have smaller census sectors, while in rural areas these sectors are wider. The application of census sectors to delimitate the landscape units can increase the information available in each landscape unit, and also aid the refinement of boundaries into more homogeneous units.

In the study area, 1404 census sectors are contained or intersect within its boundaries. Figure 6.5 shows the census sectors within the study area divided between those classified as rural sectors and urban sectors (green lines are the Tibiri fault and the geological fault and black line the river basin divisions). Most of the urban sectors are concentrated in three HGU limited by the geological fault and Tibiri fault; therefore, the census sectors were applied to refine the boundaries near these faults and the basin divisions.

### 6.2.8 Delimitation of Landscape Units

The final delimitation of the Landscape Units (Figure 6.6) was based on the criteria discussed above. Following is the description of each landscape unit as it was defined:

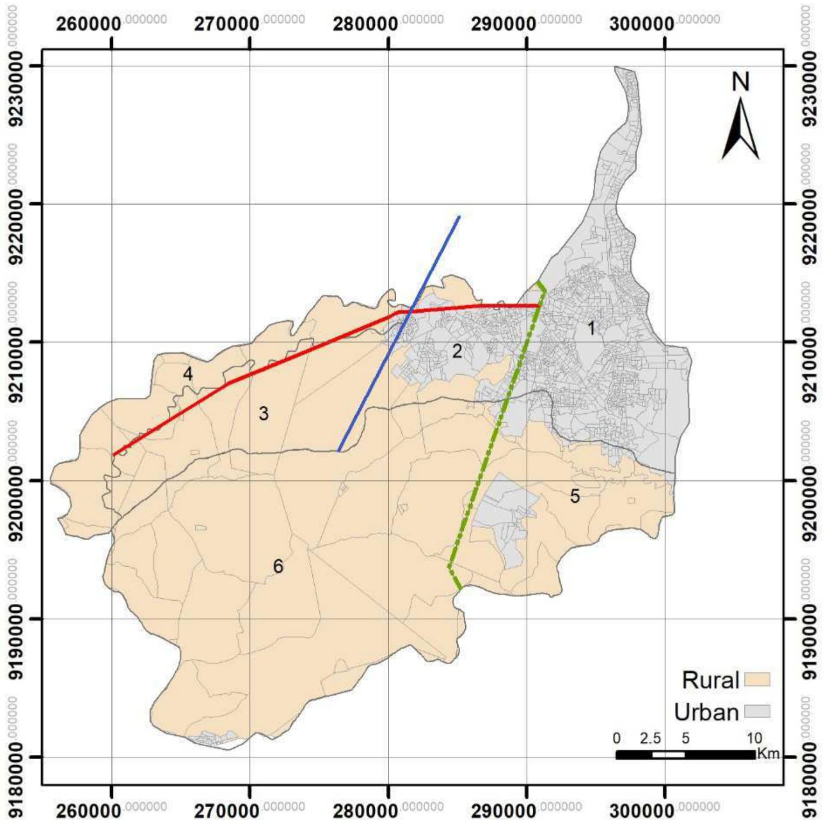

Figure 6.5 - Census sector (grey – urban; yellow – rural) and HGU

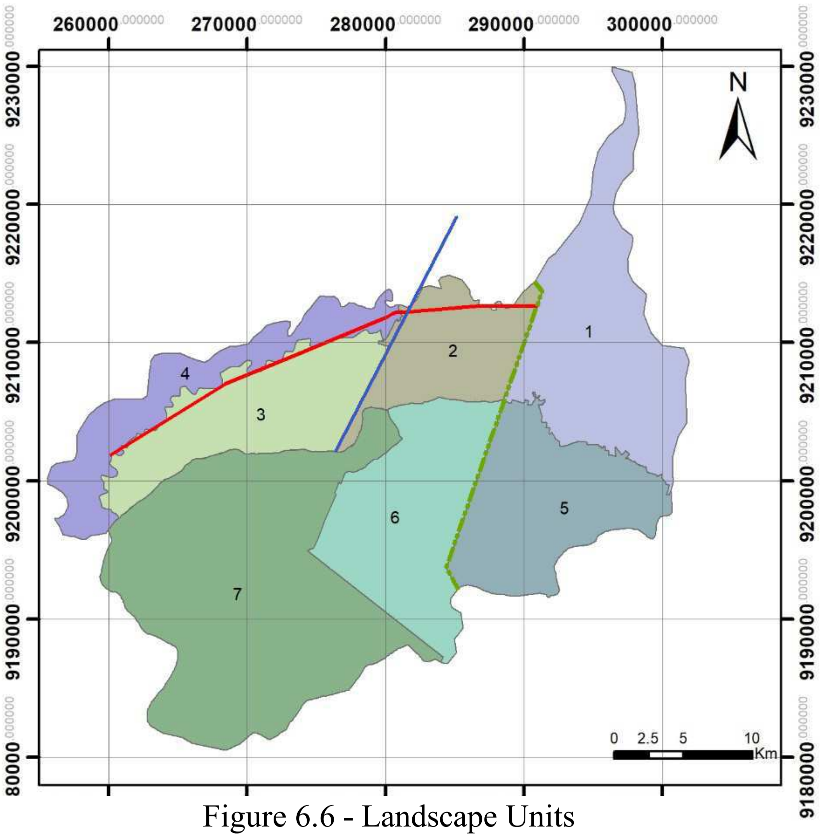

Figure 6.6 - Landscape Units

#### 6.2.8.1 Landscape Unit 1

The landscape unit 1 was defined based on the hydrogeological unit (HGU) 1. This landscape unit includes the overlapped phreatic and confined subsystems of the aquifer within the Paraíba river basin. This LU is similar to one of the management zones defined by the ASUB project. However, two changes were made. The first one regards the west delimitation of the LU. The reanalysis of the geological data allowed a slightly update on the position of the geological fault. The second change was the southern boundary: Figure 6.7 depicts the HGU 1, the census sector classified by type, and the urban land uses. There is a close match between the urban census sector and the southern boundary of the HGU 1. The delimitation of this boundary based on the urban census sectors with confirmed urban land uses can increase the demographic, social and economic data, and also homogenise the Landscape Unit as a region with high urban demands. Therefore, the following criteria were applied for the refinement of the southern boundary of the landscape unit 1:

i. The census sector that intersected with the boundary of the HGU 1
ii. The census sector located adjacent to the ones that intersected, and that had more of 50% of the current land use classified as urban.

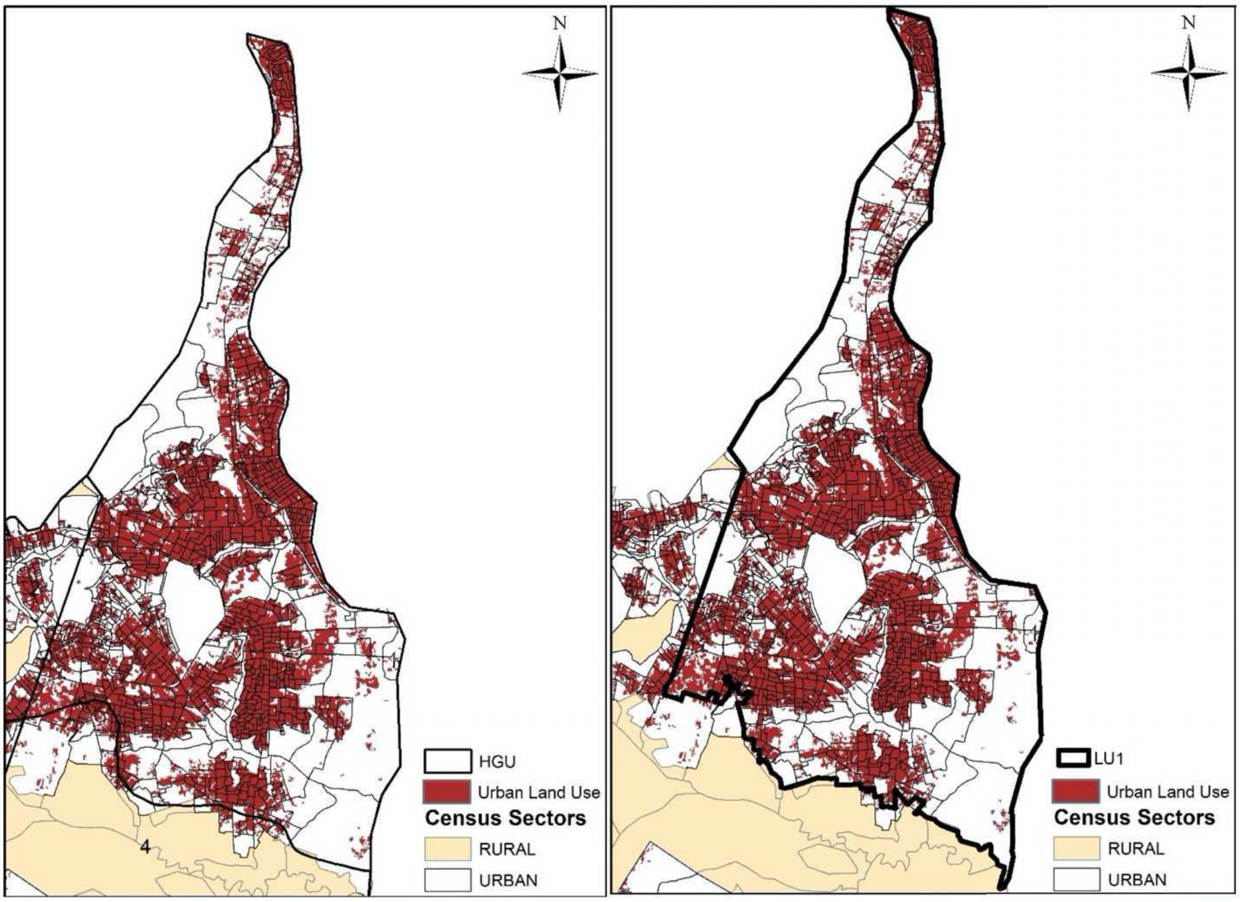


Figure 6.7 - Landscape Unit 1

### 6.2.8.2 Landscape Unit 2

Landscape Unit 2 was defined based on HGU 2. Hence, the LU2 has a phreatic subsystem with the formation Beberibe and the alluvial deposits of the Paraíba river on top of the bedrock, and the alluvial deposits of the Tibiri and Sanhaua rivers situated on top of the Beberibe formation. LU2 is the recharge area for the confined aquifer in the LU1. This LU also presents resemblance with a management unit defined in the ASUB project, such as the south border defined by the Paraíba river topographic limit and the east border by the geological fault; however, some alterations were made. First, the northern boundary of the LU2 is the Paraíba river itself. Second, the delimitation of the west border presented a situation similar to the south border of the LU1 (Figure 6.8). Therefore, the same criteria were applied to refine this border:

i. The census sector that intersected with the boundary of the HGU 2
ii. The census sector located adjacent to the ones that intersected, and that had more of 50% of the current land use classified as urban.

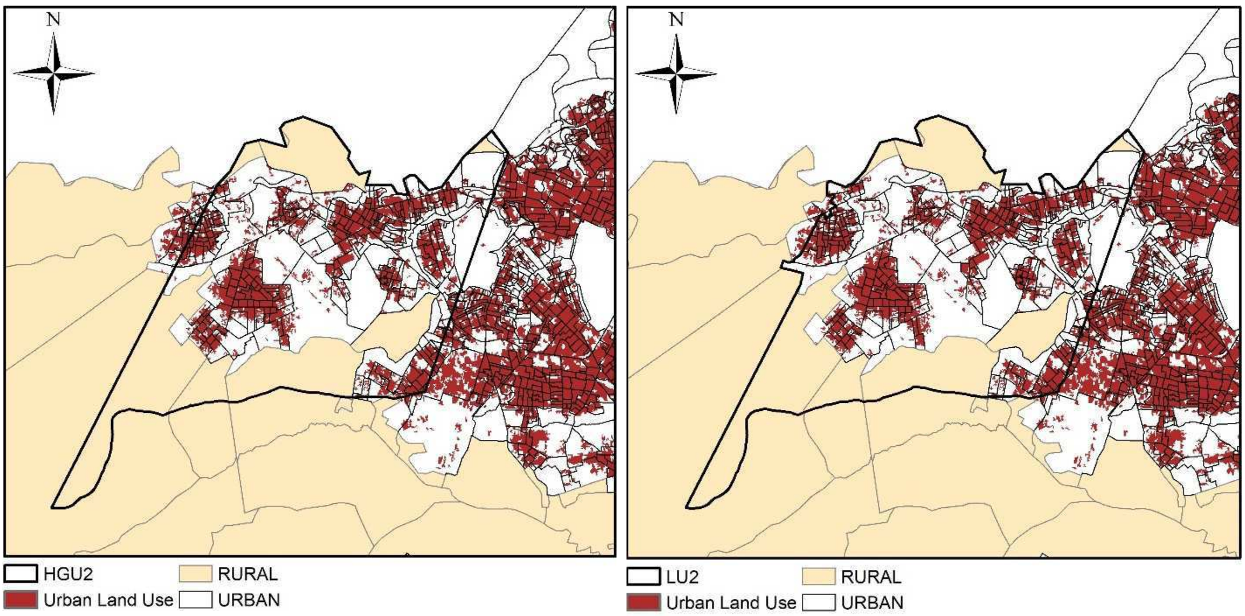


Figure 6.8 - Landscape Unit 2

#### 6.2.8.3 Landscape Unit 3

Landscape unit 3 was based in the HGU3. The southern boundary is the topographic limit of the Paraíba river basin. The east boundary is the limit with the LU2 and was adjusted due to the refinement of the previous Landscape Unit. The northern boundary is the alluvial deposits of the Paraíba river, and the west border is the impermeable granitic outcrop. Therefore, the LU3 is composed almost entirely of the Barreiras formation and, the aquifer is phreatic. The census sectors in this LU is all classified as rural. The land uses in the Landscape Unit 3 are in its majority farming (agriculture and pasture); however, there are a considerable number of forests, with the two on the northeast being preservation areas of springs.

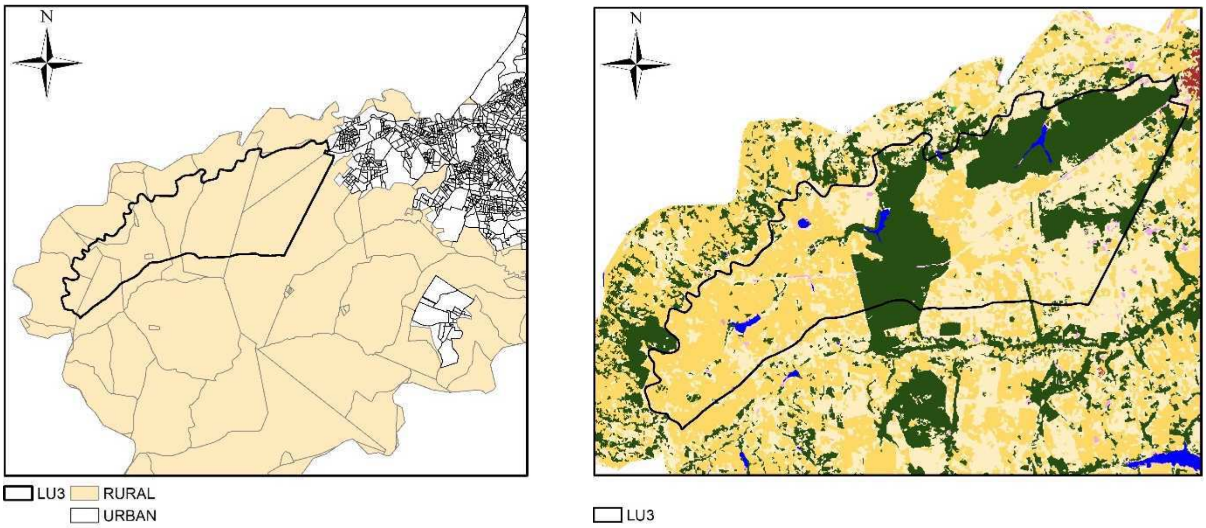


Figure 6.9 - Landscape Unit 3

#### 6.2.8.4 Landscape Unit 4

Defined based on HGU 4, Landscape Unit 4 presents a phreatic aquifer with crystalline outcrops in the southwest area. The east boundary was adjusted to comprise the refinement of the LU2. A vast majority of the census sector in this LU is rural. The land use in this LU are also mostly farms with few urban land uses are located in the urban census sectors. Natural land uses are presented scattered throughout the Landscape Unit 4; however, this distribution occurs due to the tributary rivers (blue line – Figure 6.10) of the Paraíba River.

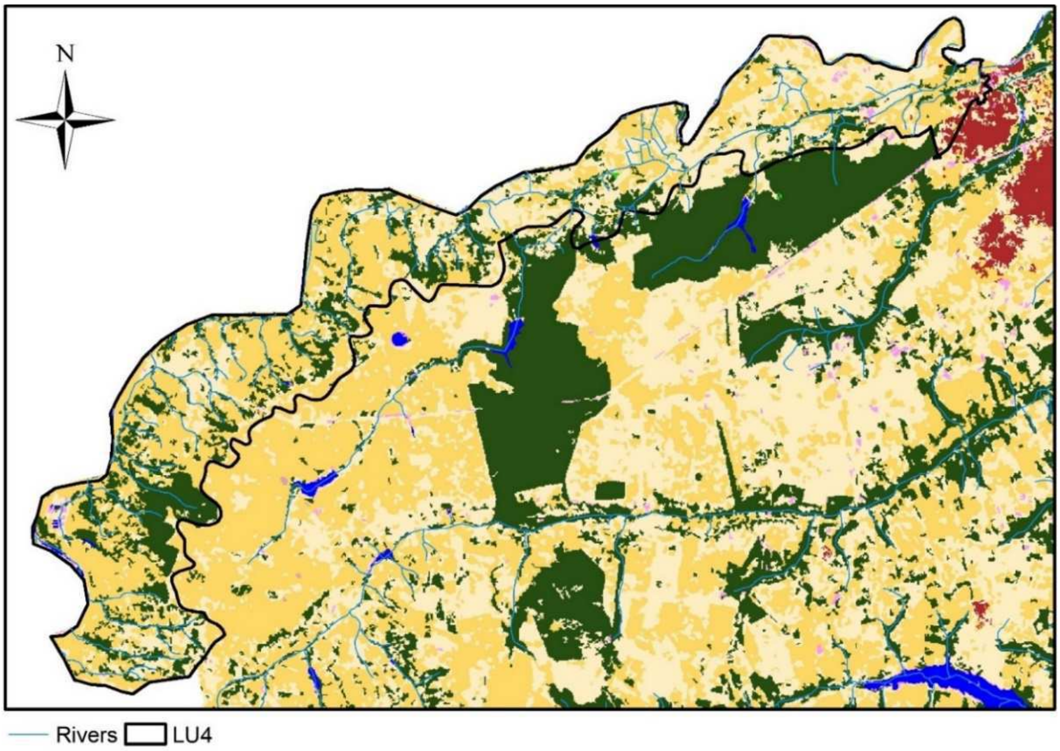


Figure 6.10 - Landscape Unit 4

#### 6.2.8.5 Landscape Unit 5

Landscape unit 5 is the first one contained in the Gramame River basin. This LU was based in the HGU 5. As the LU1, this Landscape Unit present two subsystems, one phreatic and one confined. In the phreatic subsystem, there are the alluvial deposits of the Gramame river, on top of the Barreiras formation. This LU presents both rural and urban census sector types (Figure 6.11). Although the level of urbanisation is low, this LU comprises a large number of industries situated next to the highway BR-101 that connects the city of João Pessoa to another metropolitan region nearby (Recife). Thus, most of the water rights are located in this region. As this region is also near the beginning of the confined aquifer and taking into account the influence radius of wells under this condition, the exploitation there might affect both subsystems. Furthermore, the land uses in this LU is mostly divided between farming (agriculture and pasture) and forest. A large number of forests is due to the Gramame river course and its tributaries (Água Boa, Salsa, and Ipiranga rivers).

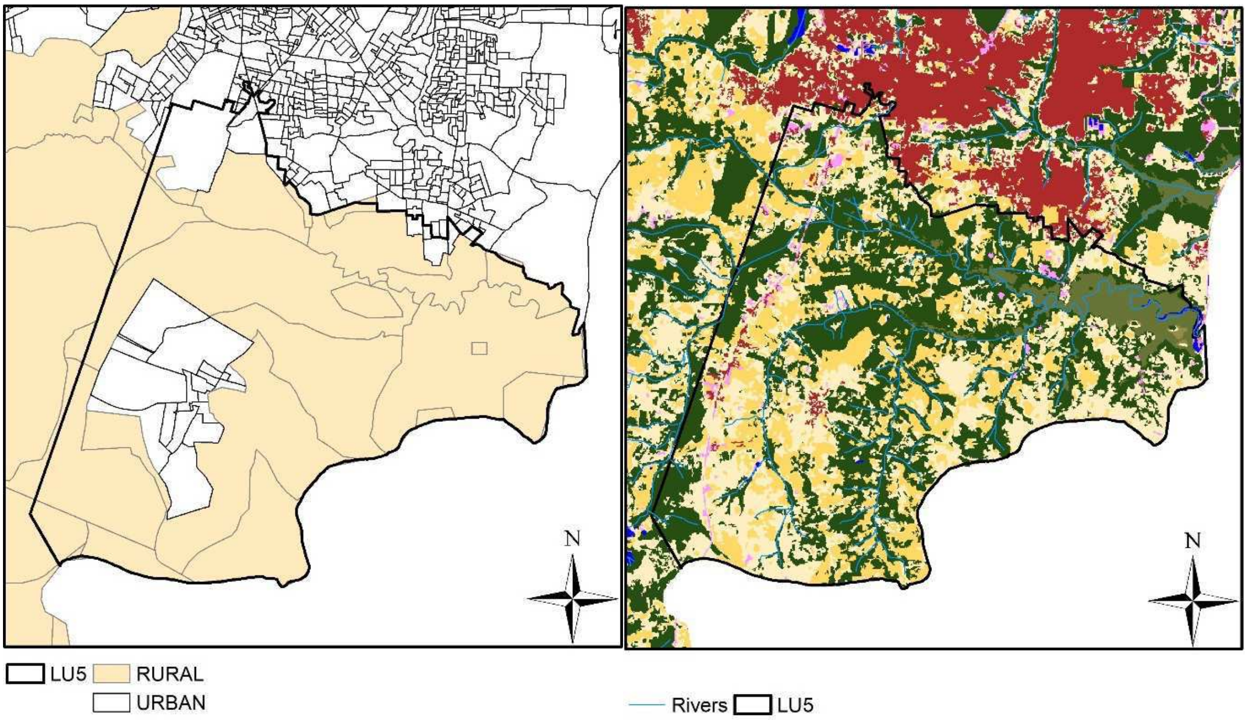


Figure 6.11 - Landscape Unit 5

#### 6.2.8.6 Landscape Unit 6

Landscape unit 6 was defined from a subdivision of the HGU 6. This subdivision was based in criteria relative to hydrogeological setting and land use. From the hydrogeological point, even though the whole HGU 6 is a phreatic subsystem, other aspects are equally important: i) there is the presence of outcrops of the Beberibe formation, so it emerges from the confined aquifer in LU5; ii) part of the HGU6 is a direct recharge zone to the confined aquifer in the LU5. Regarding the land use, the subdivision conducted sought to include the Gramame-Mamuaba dam within an LU, given that this is the primary source of water supply for the João Pessoa Metropolitan Region, and the dam influences, and is influenced by, the nearby groundwater system. The delineation of the boundaries in this subdivision, the census sectors limits was applied.

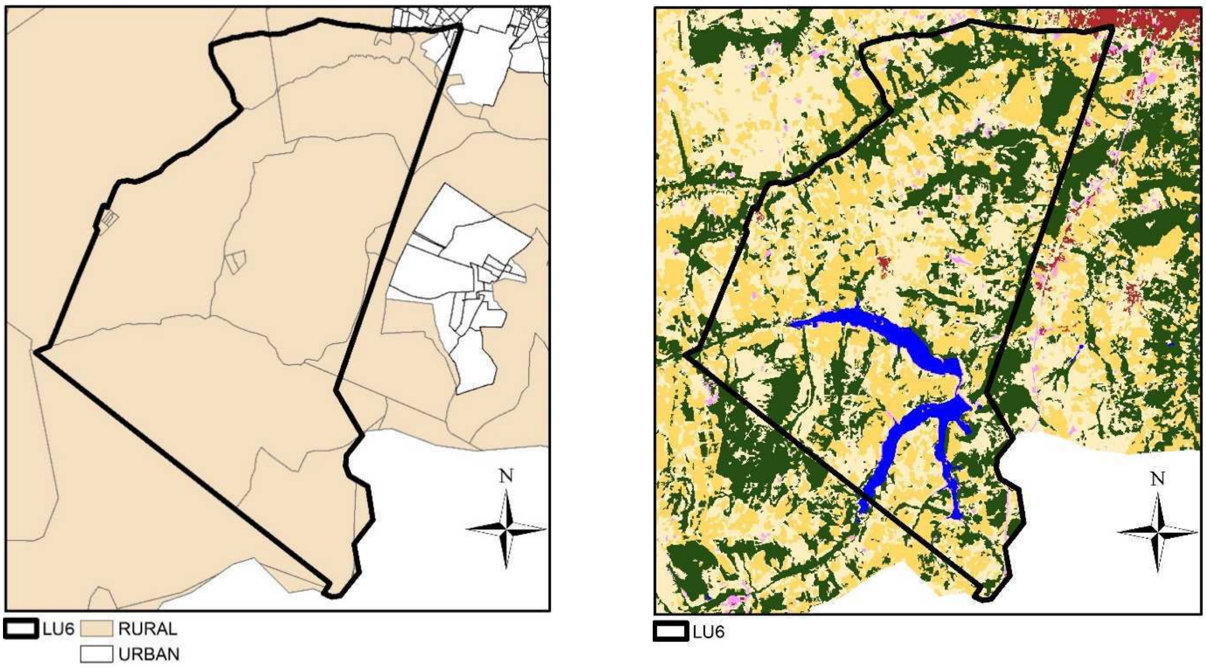


Figure 6.12 - Landscape Unit 6

#### 6.2.8.7 Landscape Unit 7

Landscape unit 7 represents the Barreiras formation of the HGU 6, other than that; there are two crystalline outcrops in the western portion. The census sectors are mostly rural with a small urban nucleus at southwest. This nucleus is located at the border of the state of Paraíba with the state of Pernambuco. This LU contains the springs of the three rivers (Gramame, Mumbaba and Mamuaba) that form the Gramame River Basin. However, analysing the land use patterns, the farming uses have a more markedly presence, and the natural land uses in spring areas are less than expected.

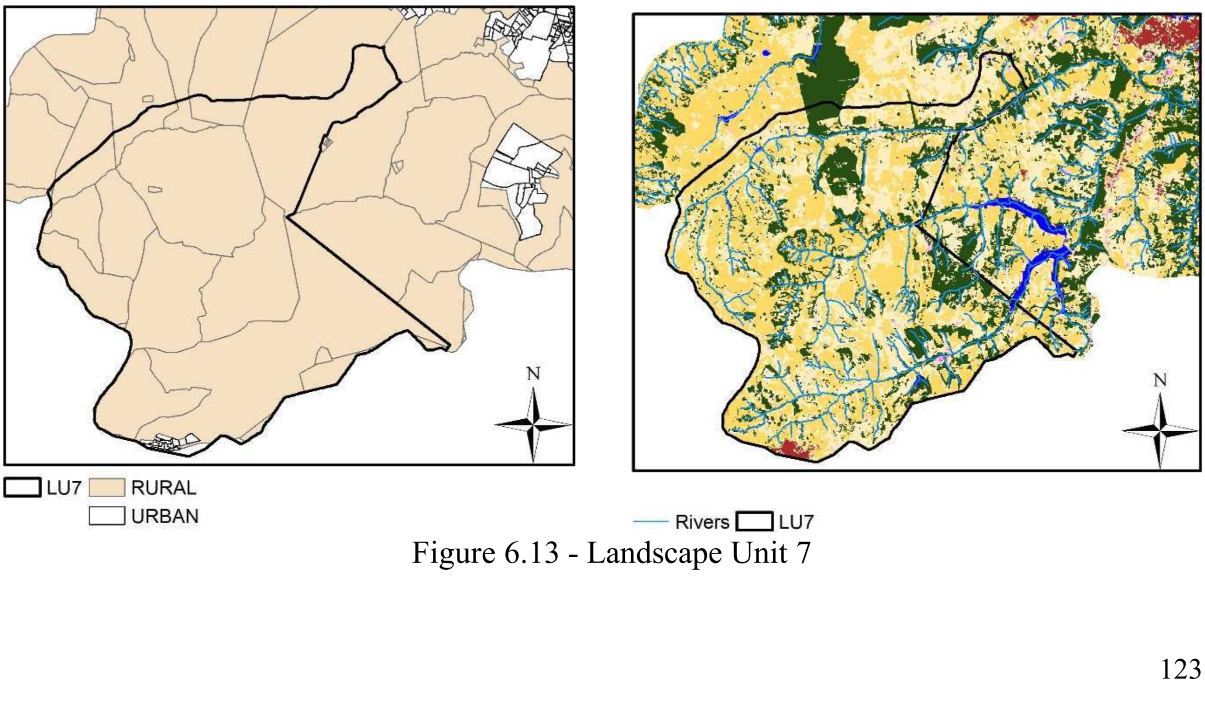


Figure 6.13 - Landscape Unit 7

### 6.2.9 Groundwater Budget and Landscape Units

The groundwater budget provides information about the diverse inflows and outflows of the system, taking into account the different boundary conditions as well as the possible sink or sources. The groundwater budget was calculated using the numerical model previous calibrated (see Chapter 4). The inflows and outflows obtained through the model represent the horizontal flow rates. Given that, an anisotropy between the horizontal hydraulic conductivity and the vertical hydraulic conductivity was assumed as 10%, given that it is expected the magnitude of the vertical flow to be in one order of magnitude less than the horizontal flow (Usman et al., 2020). Using the software FEFLOW, it was possible to select a set of elements and utilise the faces as boundaries. Each landscape unit was grouped as a set of elements. Though this application, the software calculated the variation of flow in each set of elements; hence, in each landscape unit. Furthermore, an equation was implemented to calculate the evapotranspiration in the groundwater system based on the phreatic level. This way, it was possible to estimate the net recharge reaching the saturated level of the aquifer. Groundwater budget analysis was conducted for the whole aquifer system and also for each of the defined Landscape Units.

For the period analysed (Feb/17 to Feb/18), the whole aquifer system had a recharge of approximately 320.3hm$^3$ (310mm) due to infiltration from rainfall. The average rainfall for the period was 1934.38mm. Hence, the average recharge rate was 16%. This value is similar to found in other studies (Costa et al., 2007; Fernandes, 2017). A diagram of the distribution of the recharge among the landscape units is presented in Figure 6.15. The outflow from the groundwater system due to evapotranspiration was estimated in 46.2hm$^3$ (44.7mm). Therefore, the net recharge to the aquifer was 274.1hm$^3$ or 265.64mm. Other sources of discharge are the wells from the State Water Company (CAGEPA). They represented a total outflow of 2.99 hm$^3$ during the simulated period. The total discharge from the phreatic aquifer to the sea during the period was estimated in approximately 24hm$^3$, whereas the discharge from the confined subsystem into the sea was approximately 10hm$^3$. Furthermore, it was possible to estimate the amount of from the phreatic to the confined subsystem: 32.10hm$^3$ for the period – average of 2.67hm$^3$ per month. A diagram of the flow between subsystem is presented in Figure 6.14 and the diagram of the groundwater flow among the LUs is depicted in Figure 6.16.

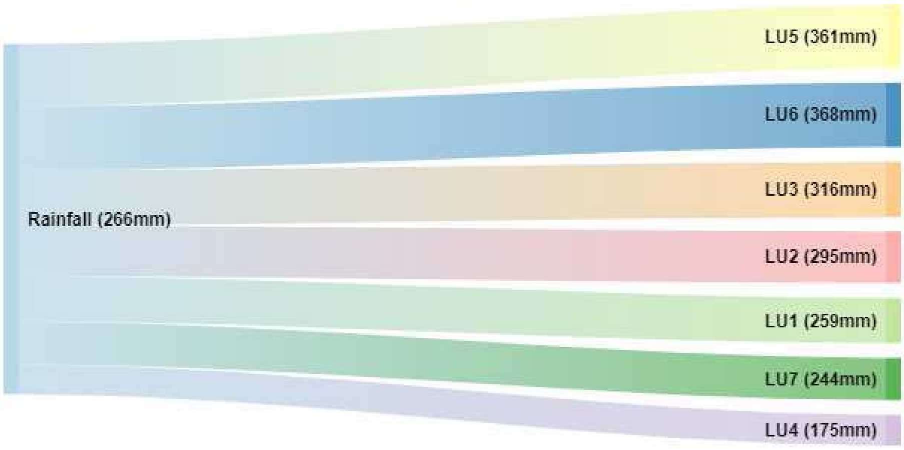

Figure 6.15 - Diagram of the distribution of recharge

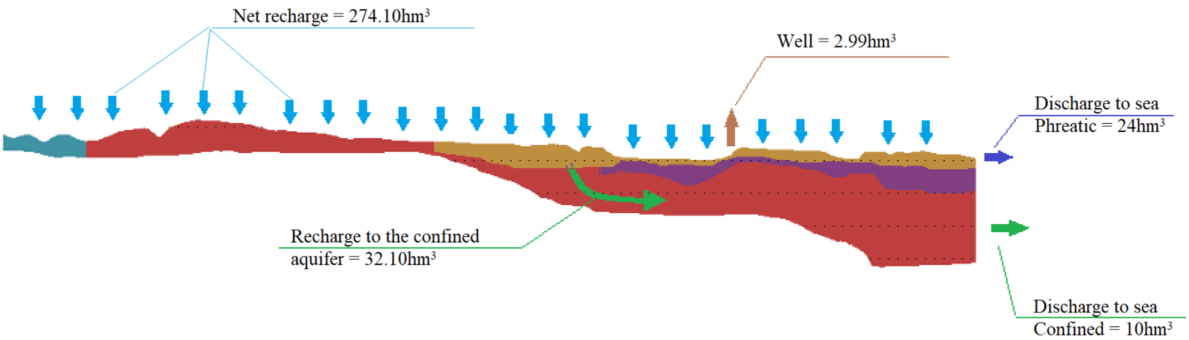

Figure 6.14 - Diagram of groundwater flow among subsystems and surface

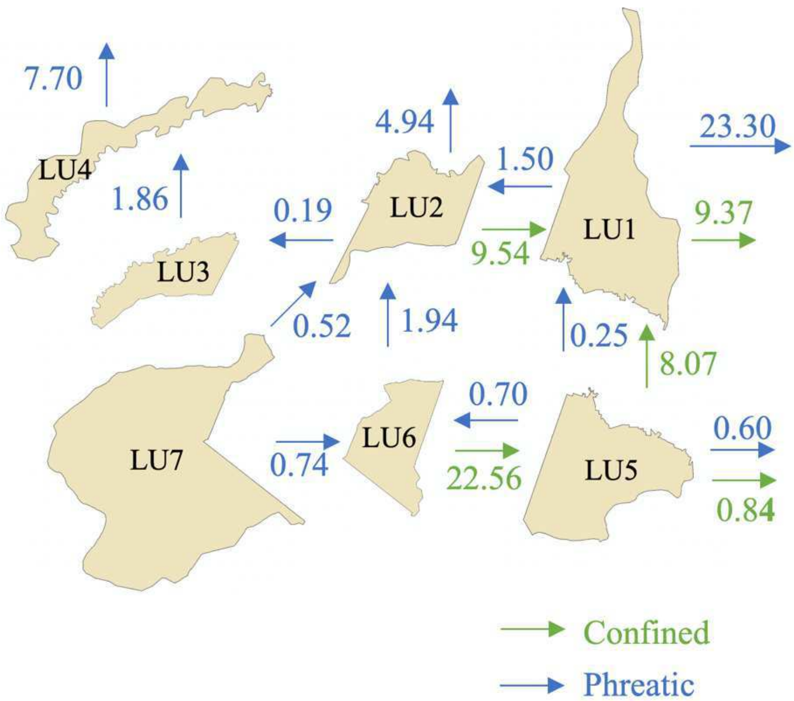

Figure 6.16 - Groundwater flow among Landscape Units (values in $hm^3$/year)

#### 6.2.9.1 Landscape Unit 1

The landscape unit 1 is composed by the phreatic and the confined subsystems. The phreatic subsystem has the rainfall as the primary source of inflow, with an estimated recharge of 43.80hm$^3$ (258mm) for the period. The discharge to the sea was estimated in 23.30hm$^3$, while the discharge as baseflow to the river over the period presented a net value of 0.17hm$^3$. The west boundary of the LU1 is delimited by the geological fault, over which the Marés River flows, and there is also the presence of the Marés dam. In this location, the simulation showed an outflow from the LU1 to the LU2 of 1.50hm$^3$. The isolines of hydraulic heads from Costa et al. (2007) (Figure 3.9) and the ones simulated (depicted in **Error! Reference source not found.**) shown that in this area, the groundwater flow is directed toward the Paraíba River and the shoreline. Also, a recharge (inflow) was estimated from the LU5 (Gramame river Basin) of 0.25hm$^3$. Hence, the recharge of the phreatic aquifer under the city of João Pessoa and Cabedelo happens from the infiltration of the rainfall, and the flow from the rest of the system has little contribution to it (Figure 6.16). This behaviour highlights the need for more integrated management of land use and groundwater.

The confined subsystem of the landscape unit 1 had an inflow from the LU2 for the period of 9.54hm$^3$; besides that, it was also estimated an inflow from the LU5 to the LU1 of 8.07hm$^3$. The total discharge to the sea was estimated in 9.37hm$^3$, and the wells representing the exploitation from the CAGEPA withdrawn 2.74hm$^3$ during the period simulated. Attention must be drawn to the flow transfer from LU5 to LU1: while in the phreatic aquifer the topographic limit has been acting as a groundwater flow limit, given that the flow in this region is only 0.25hm$^3$; for the confined aquifer the pressures resulting from the high concentration of exploitation wells have been showing their effects. The consequences of the clandestine wells were represented in the numerical model through the application of negative heads (measured in the field) as boundary conditions. As a result, this simulated an unexpected inflow, whereas it was considered a groundwater flow limit (Batista et al., 2011). This information provides more evidence that the confined aquifer under the city of João Pessoa is overexploited.

#### 6.2.9.2 Landscape Unit 2

The landscape unit 2 comprises a phreatic aquifer. The net recharge due to infiltration was 24.10hm$^3$ (294mm). Other sources of inflow to the LU2 come from the LU6, with a value of 1.94hm$^3$, and the LU7 (0.52hm$^3$), both located at the south of the LU2 and inserted in the Gramame River Basin. Therefore, it can be estimated a transfer of 2.46hm$^3$ from the Gramame River Basin to this LU. These are the sources of recharge to this LU. There is a small inflow of 0.47hm$^3$ from the LU4 to LU2, and the border between the LU2 and LU3 presented a small amount of outflow of 0.19hm$^3$. Hence, the LU had at the western boundary only a net inflow of 0.28hm$^3$. In general, the flow in this LU is directed toward the Paraiba River, as base flow, and to the LU1, as mentioned previously. The simulated outflow to the Paraíba River was 4.94hm$^3$ (Figure 6.16).

#### 6.2.9.3 Landscape Unit 3

The landscape unit 3 had a net recharge from the rainfall of 36.20hm$^3$ (316.02mm). The predominant land uses are agriculture and natural uses. Delimitated at west by the granitic outcrops, the flow in this LU occurs in the direction of the topographic slope towards the Paraíba River – from southwest to northeast. Hence, a flow occurs from the LU3 to the LU4 – the simulated value for the period was an outflow of 1.86hm$^3$ (Figure 6.16). There was practically no outflow at the south (0.01hm3); therefore, the topographic limit has also been acting as the groundwater flow limit in this LU.

#### 6.2.9.4 Landscape Unit 4

The alluvial sediments of the Paraiba River are mainly represented in the Landscape Unit 4. The net recharge for this LU was 8.4hm$^3$ (101mm); however, a part of the LU comprised a granitic outcrop where the recharge and infiltration processes are disregarded due to the lack of porosity in such formation. Therefore, excluding this area from the calculations, the net recharge was 175mm. As mentioned before, this LU received an inflow of 1.86hm$^3$ from LU3 and had an outflow of 0.47hm$^3$ to the LU2. Finally, this Landscape Unit provided an outflow for the Paraíba River of 7.70hm$^3$ during the simulated period (Figure 6.16).

#### 6.2.9.5 Landscape Unit 5

The landscape unit 5 presents a setting similar to the LU1; however, the LU5 has a much lower level of urbanisation. The net recharge for the phreatic aquifer of LU5 from rainfall was 46.90hm$^3$ (361mm), approximately 40% higher than the LU1. The culprit for this higher recharge rate is land use. Majority of the area is destinated to farming uses, followed by natural land-uses, such as forest and mangroves. This lower level of impermeabilization leads to more availability of groundwater. The outflow from the phreatic subsystem of the LU5 to the sea was estimated in 0.60hm$^3$. The baseflow to rivers were 2.48hm$^3$.

Furthermore, there are many industries in the western region of the LU5, where the inflow from the LU6 come. The net inflow from the LU6 simulated was 21.86hm$^3$. There is an outflow from LU5 to LU6 (as it happened among LU1 and LU2); however, on a lower scale: only 0.70hm$^3$, whereas the inflow from LU6 to LU5 was considerable: 22.56hm$^3$ (Figure 6.16). Two factors can be contributing for this high inflow: i) first, the Gramame-Mamuaba dam is located on top of an outcrop of the Beberibe formation, after which, the Gramame River follows the geological fault, this setting might increase the hydraulic gradient towards the confined aquifer; ii) the high amount exploited by the wells situated in the industrial zone (near the recharge area) can be inducing the recharge for the confined aquifer. Finally, the outflow from the confined subsystem of the LU5 to the sea was simulated in 0.84hm$^3$.

#### 6.2.9.6 Landscape Unit 6

The landscape unit 6 had a net recharge of 51.20hm$^3$ (368mm), the highest rate in the study area. On the one hand, the many natural and farming land uses aid this process; on the other hand, the presence of the rivers, as well as the Gramame – Mamuaba Dam, also helps to control the recharge process in the area. According to the simulations, during the period analysed the Gramame – Mamuaba Dam contributed as an inflow to this Landscape Unit providing an amount of 7.40hm$^3$. Furthermore, there is a small inflow from LU7 of 0.70hm$^3$. As for outflow, this LU discharges 1.94hm$^3$ to the LU2 and 22.56hm$^3$ (Figure 6.16) to the confined subsystem of the LU1, as mentioned before.

#### 6.2.9.7 Landscape Unit 7

The Landscape Unit 7 is delimitated at west by the topographic limit of the Gramame River Basin; therefore, there was no inflow from the surrounding systems. The primary source of recharge is the rainfall. The net recharge simulated was 64.00hm$^3$ (203mm); however, similarly to the LU4, the LU7 also has the presence of crystalline outcrop. The net recharge value corrected by the area was 243mm. As mentioned before, this LU discharged 0.70hm$^3$ to the LU6 and 0.52hm$^3$ to the LU2. Finally, another source of outflow from this landscape unit was as springs and baseflow, in the amount of 2.48hm$^3$ (Figure 6.16), for the Gramame, Mamuaba and Mumbaba rivers.

## 6.3 Temporal Dimension

### 6.3.1 The transience of groundwater systems and its influence on management

Groundwater systems have temporal aspects, such as response time to external pressures, which influences are often disregarded when analysing groundwater management strategies (Walton, 2011). This misconception emerged from the understanding that the recharge and discharge of groundwater system tend to compensate each other if the analysis comprises seasons or climatic cycles (Currell et al., 2016; Theis, 1940). For this reason, the systems are generally analysed taking into account only the temporal characteristics of the management plans and not the transience of the aquifer itself. Transience refers to the temporal (transient) dynamic behaviour of an aquifer.

Pressures on groundwater systems propagate in differing ways until the system reaches a new equilibrium (steady-state) due to their physical and hydrogeological aspects (Simpson et al., 2013). Several studies have proposed and analysed several indices for characterising the temporality of groundwater systems (Rousseau-Gueutin et al., 2013; Schwartz et al., 2010; Simpson et al., 2013).

Domenico and Schwartz (1998) proposed the *basin time constant* ($\tau$) that refers to the time necessary for a hydraulic head or flux in a groundwater system to readjust to a pressure and that $\tau$ can be calculated through the following equation:

$$\tau = \frac{L^2 S_S}{K} \qquad \text{Eq. 6.1}$$

Where L is the length of the flow system, $S_s$ is the specific storage and K is the hydraulic conductivity. Rousseau-Gueutin et al. (2013) presented a simplification of the basin time constant equation to be used for confined and unconfined aquifer:

$$\tau = \frac{L^2}{D_h} \qquad \text{Eq. 6.2}$$

Where $D_h$ is the hydraulic diffusivity calculated as the ratio between transmissivity and storativity.

Although similar to the previous time constant herein presented, another essential time constant for analysing the transience for groundwater systems was presented by Townley (1995) and referred to the aquifer's response time to transient pressures ($\tau_P$); hence, this constant includes a variable P that is the period of the fluctuations or pressure:

$$\tau_P = \frac{SL^2}{4TP} \qquad \text{Eq. 6.3}$$

According to Haitjema (2006), this index can be used to evaluate the necessity of transient or steady-state model to analyse a groundwater system depending on the pressure: if $\tau_P > 1$ is recommend a steady state model with time-averaged boundary conditions and recharge rates; if $\tau_P < 0.1$ is recommended a steady state with boundary conditions and recharge rates for a determined time-step, such as dry season condition; and if $0.1 < \tau_P < 1$ is recommended the use of a transient model (Haitjema, 2006). It must be highlighted that the indices previously shown refer to a one-dimensional model. Therefore, their applicability should consider the length of the flow (L) as the distance from recharge to discharge in one direction (Currell et al., 2016); for analysing a two-dimensional unconfined flow, it is necessary the application of numerical models due to the high nonlinear situation of the system (Rousseau-Gueutin et al., 2013).

Furthermore, Domenico and Schwartz (1998) assumed that a change in the hydraulic head followed an exponential form:

$$h(t) = h_o + (h_1 - h_0)^{1-e^{\left(-\frac{t}{\tau}\right)}} \qquad \text{Eq. 6.4}$$

where h = hydraulic head, $h_o$ = hydraulic head at t = 0; $h_1$ – new hydraulic head at t = $t_1$; t – time; $\tau$ – time constant. Therefore, equations 1 and 2 can be used to predict the hydraulic head response in a determined time (t) and a distance (L) from the pressure point.

Domenico and Schwartz (1998) defined the period of one time constant as the time t = $\tau$. This period represents that an adjustment of 63% from $h_o$ to $h_1$ has occurred. When t = 5 $\tau$, it is expected that 99% of the adjustment has occurred and the system have returned to a new equilibrium (Schwartz et al., 2010). Rousseau-Gueutin et al. (2013) estimated the time necessary for an aquifer reach the near-equilibrium (when 95% of the initial pressure has been dissipated) using an analytical solution of the flow equation, and Equations 1 and 3. The time near-equilibrium occurs when t=3$\tau$.

The calculation of indices relative to the transience of the aquifer is essential to understand and analyse the effects of the diverse pressures that change the aquifer. Both anthropogenic (Han et al., 2017) and natural (Klammler et al., 2020). It can take from a few years to millenniums until the system reaches a new equilibrium (Han et al., 2017; Rousseau-Gueutin et al., 2013). Hence, the management of groundwater might encompass only part of the effects on the systems. Therefore, the transient aspects of aquifers must be included when analysing groundwater management strategies.

### 6.3.2 Assessment Framework for Transience in Hydrogeological Systems

Currell et al. (2016) recently proposed a framework to reduce the gap between the temporal aspects and the management of groundwater. This framework is applied to determine the influence of transience in the groundwater system under management perspective. The framework takes into account three aspects: i) time-dependent hydraulic response of the system; ii) nature of the external pressure, given that it can cause temporary or long-term impacts; and iii) the management timeframe in which the response is considered important (Currell et al., 2016).

To aid the determination of the likely importance of transient behaviour, Currell et al. (2016) proposed the application of two graphical tools. The first one, Figure 6.17(a), regards the assessment of the influence of transient periodic pressures, such as climatic shifts or seasonal rainfall, while the second one, Figure 6.17(b), regards the assessment of the response to a rapid monotonic (step) change, such as deforestation and its resulting changes in the recharge rate.

A thorough explanation of the application of the framework can be found at (Currell et al., 2016). Herein is provided with a brief explanation of its application:

Using the graphical tool Figure 6.17(a), the influence of periodic pressure can be analysed following these steps: i) calculate the time constant of the system and selected it on the x-axis; ii) the period (time) of the pressure is selected on the left-hand of y-axis; iii) plot the point based on steps i and ii to evaluate if the influence of transience due to the pressure is likely (shadow grey zone); if negative: lower influence of transience, then simplified steady-state model can be used, if positive: iv) select the management timeframe on the right-hand of y-axis; if the timeframe is higher than the point plotted: pressures are driving the transience of the system, and transient modelling should be incorporated into the groundwater management.

The graphical tool in Figure 6.17(b) is applied to assess the response to rapid monotonic changes. The steps for its application are the following: i) calculate the time constant of the system and selected it on the x-axis; ii) select the management timeframe on the y-axis; iii) plot the point based on steps i and ii; iv) if the point is located above the line defined by $t = 3\tau$ (near-equilibrium) the system will reach the steady-state during the management time frame; if the point is located under the line, the system will be in a transient state of adjustment, and a transient model is required.

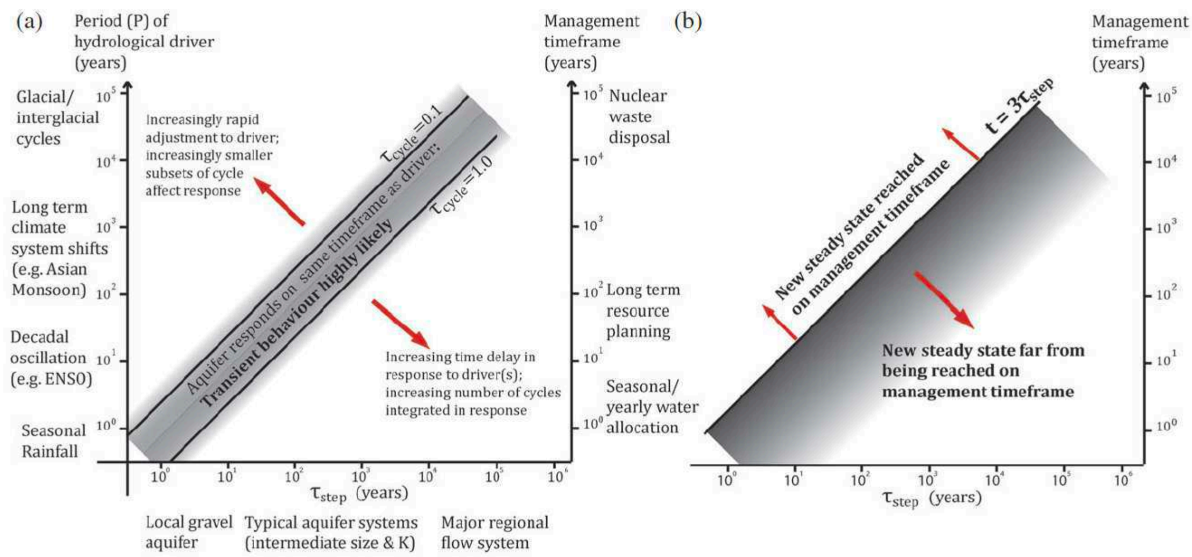


Figure 6.17 - Graphical tools for evaluation of the transience due to periodic (a) and monotonic (b) pressures (Currell et al., 2016)

### 6.3.3 Assessment of Transience in a Hydrogeological System

The temporal dynamics of the study area was analysed through the application of the framework previously explained proposed by Currell et al. (2016). Some simplifications were required in order to apply the framework, given that such a framework is a first assessment tool and not a strict analytical model. First, the flow of the system was considered in one direction; therefore, it was determined a predominant direction of the flow taking into account the recharge and discharge area for each landscape unit, as well as the isolines of hydraulic head in the groundwater system. Second, to calculate the transmissivity in the phreatic aquifer, it was considered the average of the saturated thickness (Rousseau-Gueutin et al., 2013). Third, when more than one value of hydraulic conductivity and specific yield was present, an average of the values was applied.

Figure 6.18 shows the study area divided by Landscape Units, the isolines of hydraulic heads and the major rivers. Given that the Landscape Units 1 and 5 comprise two subsystems, calculations were conducted for each of them. Following the steps of Currell's framework, initially, the values of basin time constants were calculated for each landscape unit using the equation 6.2. The results are presented in Table 6.1.

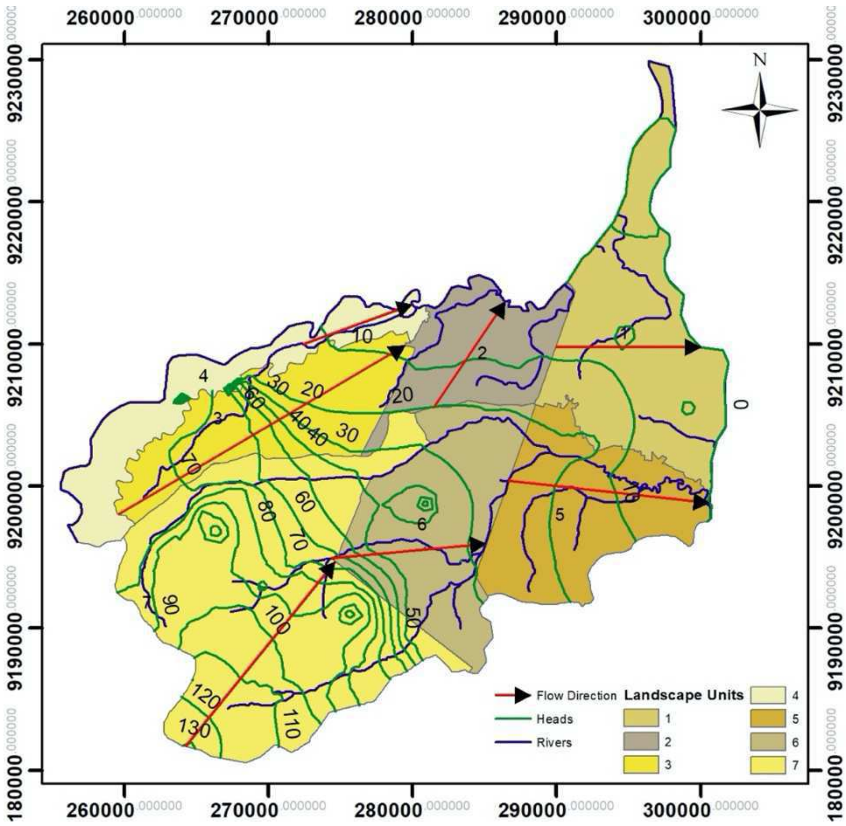


Figure 6.18 - Predominant direction of groundwater flow

Table 6.1 - Parameters of LUs and basin time constant

| Landscape Unit | Length of the flwo (m) | Hydraulic Conductivity (m/year) | Saturated thickness (m) | Tranmissivity (m²/year) | Effective porosity or Storativity | Basin time constant (year) |
|---|---|---|---|---|---|---|
| LU1 | 10021 | 4617.25 | 45.75 | 212393.50 | 0.1681 | 79 |
| LU1 - C | 10036 | 153.30 | 300 | 45990.00 | 0.0045 | 10 |
| LU2 | 8943 | 1697.25 | 231 | 392064.75 | 0.1193 | 24 |
| LU3 | 23168 | 1647.98 | 85 | 140077.88 | 0.13465 | 516 |
| LU4 | 8093 | 4252.25 | 26 | 110558.50 | 0.2658 | 157 |
| LU5 | 14041 | 4617.25 | 99 | 457107.75 | 0.2702 | 117 |
| LU5 - C | 14041 | 153.30 | 315 | 48289.50 | 0.004725 | 19 |
| LU6 | 10828 | 4584.40 | 143 | 655569.20 | 0.2759 | 49 |
| LU7 | 16650 | 4255.90 | 66 | 280889.40 | 0.125 | 123 |

The shortest basin time constant was in the confined portion of the LU1 (approximately ten years) and the LU5 (approximately 19 years). This happens due to the effect of confinement in the subsystem (Rousseau-Gueutin et al., 2013). The most extended time constant was found for the landscape unit 3. This LU has the most extended length of the flow, low values of hydraulic conductivity and saturated thickness when compared to the other LU. The plot of the time constant relative to each landscape unit according to graphical tool 1 (Figure 6.17a) is depicted in Figure 6.19. These data were plotted considering a decadal oscillation relative to 10 years. In orange is the line referring to the timeframe of management, defined as 20 years, given that this is the period usually applied in the management plans.

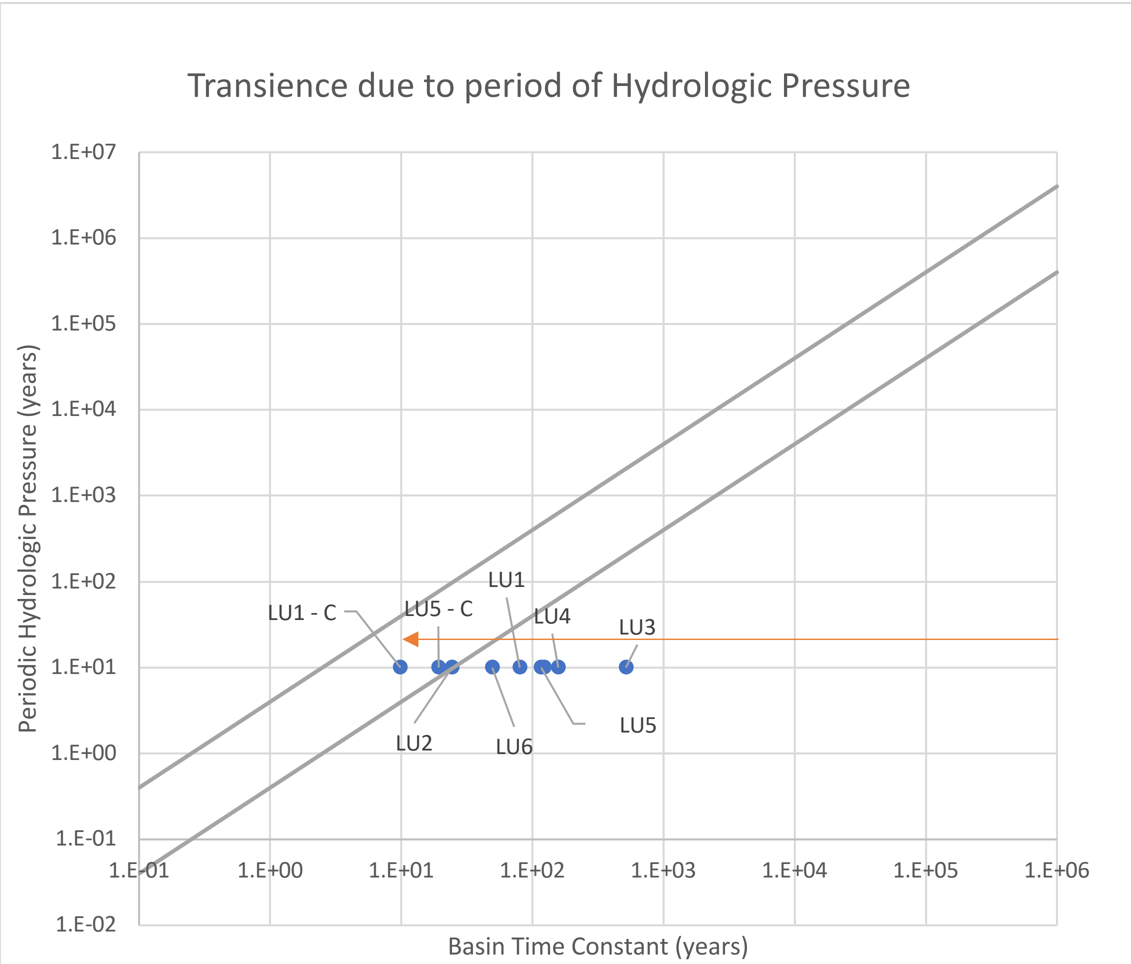


Figure 6.19 - Time basin constant and transience due to periodic pressures

The landscape units LU2, and the confined portion of the LU1 and LU5 are situated within the transience zone defined by the framework. Also, these Landscape Units are situated under the line of the management timeframe. This means that these Landscape Units need transient modelling and that the temporal dynamics of the system as a response to this pressure (a decadal oscillation in climate) needs to be incorporated to the management plans. The remaining Landscape Units (LUs 6,1,4,5,3) are located under the transience zone. Therefore, for these LUs the temporal dynamics also need to be included in the groundwater management; however, this inclusion can be through the application of steady-state models that considering, for example, the maximum and/or minimum situations, average wet and/or dry conditions.

Afterwards, the graphical tool 2 (Figure 6.17b) was applied to investigate the temporal dynamics due to rapid monotonic pressures. The time constant previously calculated were plot against the management timeframe (20 years). The graph is shown in Figure 6.20.

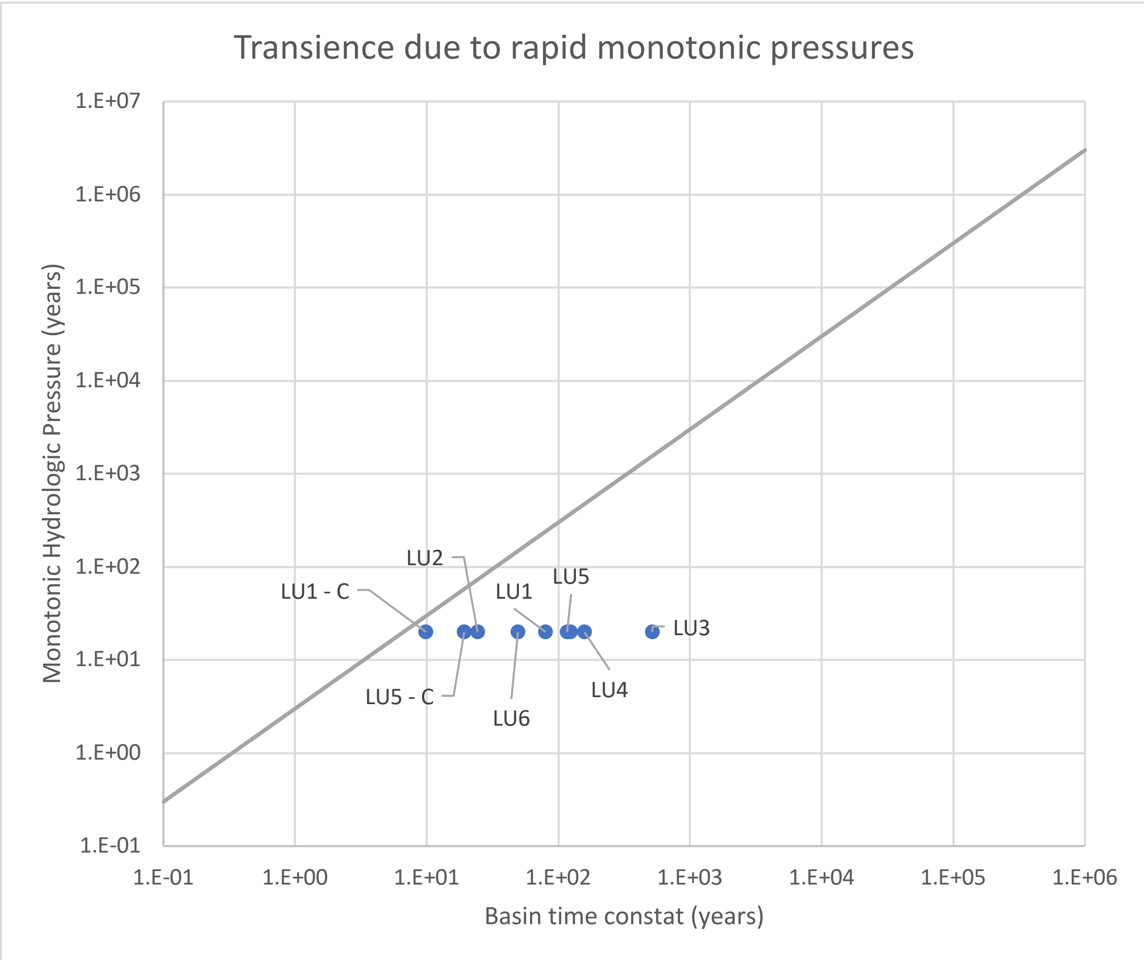


Figure 6.20 - Basin time constant and transience due to rapid monotonic pressures

All the landscape units were situated under the line proposed by the framework that defines the near-equilibrium of the system. Therefore, rapid changes, such as land-use alteration (deforestation or urbanisation), are still going to be felt during the management timeframe. Hence, the system will be adjusting to the pressures within the management timeframe, and transient simulations are necessary to analyse the aquifer's response.

## 6.4 Spatial and temporal dynamics

The sustainable yield is a standard metric for groundwater allocation. The determination of this metric has been widely acknowledged as dependent on the natural recharge and the dynamics of the capture zone (Demiroglu, 2019; Kalf and Woolley, 2005; Zhou, 2009). Besides, the sustainable yield also depends on the spatial scale applied. This yield is generally calculated taking into account the whole scale of the system, which is also the boundaries adopted for management purposes. The sustainable yield should be defined at an appropriate small spatial scale, to take into account local effects as the discharge of the system and the adjustment to pumping stresses (Maimone, 2004). However, the dependence of the sustainable yield on the spatial scale has not been taken into account, and defining the sustainable yield over a large area can lead to an underestimation of the value (Abrishamchi et al., 2020). Loaiciga (2017) and Zhou et al. (2017) are examples of the application of this concept over large areas. The former applied a mass balance while the latter applied numerical modelling for the calculation. Although valid, these studies were conducted on a coarse spatial scale. As a consequence, specific aspects as well as their effects on the system, have not been analysed. One of such aspects is the land use patterns and its influence on the recharge of the aquifer.

Embracing such analysis through the use of landscape units can be a powerful tool. The results obtained by this research using numerical groundwater modelling showed that the determination of the groundwater budget for the whole groundwater system as well as for each of the landscape units can be a feasible process and can shed light on the behaviour of such hidden resources like groundwater. Therefore, landscape units could be an alternative to the delimitation of the boundaries for groundwater management. The numerical model presented a value of 265mm of net recharge for the whole aquifer. However, the system has an inherent spatial complexity with different land uses. As a consequence, such value would be an overestimation for three landscape units (LU4, LU7 and LU1) and an underestimation for the four remaining ones (LU5, LU6, LU3, LU2). This result goes in line with the findings presented by Maimone (2004) and Abrishamchi et al. (2020).

Furthermore, the analysis through the landscape units highlights the influence and importance of land use patterns for the sustainably of groundwater. The landscape unit 1 and landscape unit 5 are closely located and have similar biogeographic characteristics. Yet, very different land-use patterns. While the latter presents mostly forest and farming uses, the former is dominated by urban land uses. As a result, the values obtained from net recharge due to rainfall varied considerably: for the LU1 the recharge was 258mm, while for the LU5 the recharge was 361mm, approximately 40% higher than the LU1. The influence of land use in the recharge processes has been widely discussed in the literature (Hall et al., 2020; Usman et al., 2020; Zipper et al., 2017), and even though the recharge is not translated in groundwater availability for exploitation (Zhou, 2009), the recharge can be considered a useful proxy for the availability (van der Gun and Lipponen, 2010). Besides that, the results here presented go further in highlighting the importance of the spatial variation of the recharge (Hu et al., 2019) and its influence on groundwater management.

Another advantage of the application of the landscape units to support the groundwater management comes from the possibility of analysing the system taking into account its internal specificities. One of the results of this application was the improvement of the conceptual model for the study area from the indications of interbasin groundwater flow, not considered previously, might be occurring. Interbasin groundwater flow (IGF) occurs when the recharge from one basin flow beneath the surface limit and discharges in another basin (Frisbee et al., 2016). This assumption is typically disregarded with the conceptualisation that the flow occurs within the surface basin (Frisbee et al., 2016) and difficult to quantify (Zanon et al., 2014).

At total, the analysis of the groundwater system showed an inflow from the Gramame River Basin to the Paraíba River Basin of 2.71hm$^3$ for the period in the phreatic subsystem. Such estimate could also be achieved using other methods or even the numerical model considering only the adjacent basins (Ye et al., 2016; Zanon et al., 2014). The benefit of analysing the system through landscape units was the identification that an inflow from LU6 (located within the Gramame River Basin) was occurring to the LU2 (located at the Paraíba River Basin), in the amount of 1.94hm$^3$, which means that more than 70% of this IGF happens at a specific region. Moreover, another IGF was identified among the confined subsystems of the LU5 and the LU1 (again from the Gramame to the Paraíba River Basin) in the amount of 8.07m$^3$. Almost three times the IGF of the whole phreatic subsystem. Fan (2019) suggest as factors for the occurrence of IGF the scale, drainage position, climate gradient, substrate properties and geological structure. However, the two basins are similar regarding such factors. If the LUs from the two basins is compared, some differences can be pointed out: the land use and well's location. The LU1 and LU2 have the majority of urban land use with a high concentration of wells, while in the LU5 and LU6 farming and natural land uses are present with a low concentration of well. Such configuration might be inducing the occurrence of this IGF. Therefore, this finding brings vital information to support the groundwater management in the area.

The temporal dynamics was assessed using Currell et al. (2016)'s framework. Even though the results obtained from this method represents a first simplified estimate, the findings can be applied as valuable guidance for the inclusion of temporal dynamics into groundwater management. For all the landscape units, the analysis of the transience due to rapid monotonic pressure (e.g. land-use change) showed that the transience needs to be informed to the groundwater management. The LU2 presented relative short basin time constant (24 years) indicating that the effect of land-use change might interfere in the discharge for the Paraíba River within a planning horizon. The confined subsystems of landscape units LU1 and LU5 also have a short basin time constant (10 and 19 years respectively); hence, the effect of land-use change in the recharge zone of the confined subsystem might reduce the availability of groundwater for allocation near the shoreline even faster than a full planning horizon of 20 years. Coincidentally, the areas that presented the shortest basin time constant are the ones more exploited; this highlights the need for a more adaptive groundwater management. This finding is restricted to the quantitative aspect (Rousseau-Gueutin et al., 2013). The calculation of indices relative to the transience of the aquifer is essential to understand and analyse the effects of the diverse pressures that change the aquifer, both anthropogenic (Han et al., 2017) and natural (Klammler et al., 2020). It can take from a few years to millenniums until the system reaches a new equilibrium (Han et al., 2017; Rousseau-Gueutin et al., 2013). Hence, the management of groundwater might encompass only part of the effects on the systems. Therefore, the transient aspects of aquifers must be included when analysing groundwater management strategies.

The analysis of the spatial and temporal dynamics of the groundwater system through the landscape unit approach has shown that this method can present a further step in the delimitation of boundaries for groundwater management. Unlike the surface water resources, where the catchment is the traditional management unit, groundwater resources can span to different hydrological, political, and even institutional boundaries. Therefore, management units for groundwater need more refinement in their delimitations. The landscape units can merge diverse aspects into more homogeneous management units, aiding the provision of information for groundwater management.

## 6.5 Conclusions

A cultural landscape was divided into landscape units underpinned by the concepts of the Landscape Scale Planning approach. These landscape units were delimited according to defined criteria, and their purpose was to analyse the spatial and temporal dynamics of a groundwater system as well as their capacity to provide information to support groundwater management analysis. Landscape units were delimited using criteria relative to topography/hydrology, hydrogeology, water uses, land uses and census sectors. The integration of these diverse layers of information into homogenous areas shows the potential that landscape units have as a locus for analysis and its feasibility to be applied as management units for groundwater management. Therefore, seven landscape units comprising similar environmental, cultural and socio-economic settings were delimited. Thus, environmental information (from the hydrological and hydrogeological aspects) could be analysed in the same place as the socio-economic information (water use, land use and census sector). In this chapter, this analysis was conducted focusing on the underlying groundwater system.

With the seven landscape units defined and the aid of the numerical groundwater model, the spatial dynamics of the system and the integration of LUs to provide information for the whole-of-landscape were analysed. The groundwater budget for the whole system was determined. The estimated net recharge was 274.10hm$^3$, corresponding to 265.64mm, with a discharge to the sea of 24hm$^3$ from the phreatic system and 10hm$^3$ from the confined subsystem. Besides, a recharge of 32.10hm$^3$ from the phreatic to the confined subsystem was also determined. Framing this system into the seven landscape units defined provided further and more detailed information. As a result of the breakdown into landscape units, the net recharge in each LU was determined, which showed the difference caused by the many land-use patterns of the LUs, such as more recharge in natural and farming land uses then in urban land uses. Another significant result was the indication of interbasin groundwater flow (IGF) occurring from the Gramame River Basin to the Paraíba River Basin. In the phreatic subsystem, a large part of this flow (70%) occurs from the LU6 to the LU2; this inflow is equivalent to 1.94hm$^3$. Another IGF was also detected but in the confined subsystem. This inflow is substantially higher and happens from the LU5 to the LU1 in the amount of 8.07hm$^3$. Both LU1 and LU2 presents similar land-use patterns and water uses. Therefore, this information has the potential to guide groundwater management measures that also takes into account the land use influence.

The temporal dynamics of the groundwater system and its inclusion in the groundwater management was assessed considering the different landscape units (including the confined and phreatic subsystems). The results showed a wide span in the basin time constant, meaning that the consequence of disturbances for some landscape units can take up to 500 years to have an adjustment of approximately 60%, while for others it can take 10 years. Pressures from rapid monotonic changes, such as land-use alterations, can affect the discharge of the aquifer within 20 years planning horizon. Coincidentally, the areas that presented the shortest basin time constants are the ones more exploited; this highlights the need for a more adaptive groundwater management. It is worth note that the transience framework guiding the analysis of the temporal dynamics requires several simplifications from the system. Therefore, these estimates are first indications to raise awareness regarding the inclusion of the temporal dynamics to groundwater management.

The analysis conducted in this chapter were restricted to the quantitative aspect of both spatial and temporal dynamics. Further research could comprise the analysis regarding the qualitative aspects such as the time lag from nitrate leaching and its occurrence through the groundwater system as well as its occurrence in each of the landscape units. Furthermore, the landscape units themselves also have spatial and temporal dynamics. The delimitation of LUs applied criteria such as land use, water use and census sectors. The spatial distribution of these three criteria changes during a determined time. Therefore, the evolution of landscape units, from the past until possible futures, might carry information that could support the groundwater management.

# Chapter Seven: Summary and Conclusions

The hidden nature of groundwater has made this resource often overlook. The added demand by the population growth and the negative impacts of climate change resulted in unprecedented pressure on groundwater resources, leading to diverse issues. On top of that, groundwater management has proven to be a complex and difficult task. Traditional approaches based only on technical aspects, such as the hydrogeology, have been incapable of dealing with the issues faced by groundwater management. As a consequence, an integrated approach for groundwater management comprising other aspects, such as the socio-economic and cultural, have been advocated. Besides, with this integrative approach comes a wide range of data and information that need to be organised and incorporated in a compatible form to support the groundwater management process. Even though such approaches have been sought, an essential factor has still not been fully included: the land-use changes. While land-use changes have a strong biophysical relationship with groundwater system, they are mirrors of socio-economic and cultural development. Hence, the integration of land-use change with groundwater management has paramount importance.

This research has hypothesised that Landscape Scale Planning – a land-use planning approach – is capable of underpinning the integration of land-use change with groundwater management while adding the provision of information for groundwater management.

The concepts embraced by the Landscape Scale Planning framework (spatial, temporal and modification dimensions) and their connections with land-use changes were analysed with respect to groundwater management concepts (Chapter Two). This research has found that there is an intrinsic connection between Landscape Scale Planning and groundwater management. This connection was confirmed by how these dimensions relate to the characteristics of groundwater systems. On top of that, no methodological barriers for the application were found. Hence, the challenge lies in the frame of analysis of groundwater management from the Landscape Scale Planning perspective. This also allowed this research to identify a further challenge that has not been fully considered in the current groundwater management approaches for coastal aquifers: the dynamics of land-use change. Therefore, framing the analysis of groundwater management under the Landscape Scale Planning perspective can provide more accurate information, given that it takes into account differing spatial and temporal characteristics of the aquifer, as well as the dynamics of change through the land use. Upon these findings, guidelines for groundwater management based on the Landscape Scale Planning approach was suggested.

Central to the implementation of the Landscape Scale Planning framework is the concept of landscape units. In order to verify the applicability of landscape units, a cultural landscape with a groundwater system was chosen as a study area. This cultural landscape situated on the coast of Northeast of Brazil was analysed from different aspects (Chapter Three). The biophysical and socio-economic aspects of the study area such as climate, geology, topography, population, and water demand, were presented. Furthermore, an analysis of the institutional setting regarding the land use planning and groundwater management in the area was conducted. The analysis of the institutional aspect of land use planning resulted in the following possible perspectives for the integration of land use with groundwater management using the instruments constrained by the institutional setting: i) basic urban infrastructure; ii) green areas and public open spaces, and iii) permanent preservation areas and natural landscape protection areas. Finally, the institutional framework for groundwater management was presented and analysed. According to the existing literature, this institutional framework has several gaps that have led or could lead to diverse issues for the sustainability of the groundwater system. As a solution, diverse alternatives have been proposed by past studies to address these gaps. Among them, the ASUB project proposed the implementation of management zones, that can base the application of the landscape units.

As a tool to evaluate the potential landscape units to provide information for groundwater management, a groundwater model was built (Chapter Four). The challenge in setting up this tool for the study area lies in a considerable lack of biophysical data and information. However, part of this challenge was overcome. For steady-state, the numerical groundwater model was calibrated, presenting a reasonable agreement between the heads calculated and simulated. Unfortunately, for the transient state, such calibration was not achieved due to the lack of data. As an alternative, a parametrisation process was applied. This process allowed the groundwater model to simulate the behaviour of the groundwater systems with good matching.

This challenge led to a further investigation seeking to increase the knowledge of the groundwater system through the use of data commonly available (Chapter Five). The increase in knowledge of the aquifer results in the provision of more reliable information for groundwater management. Estimates of hydraulic conductivity were generated from specific capacity data and spatialized through the application of geostatistics methods. As a result, the combination of these data generated information on most likely zones with higher hydraulic conductivity. Furthermore, the calibration of the groundwater model was improved with a considerable reduction in the uncertainty. Therefore, more reliable information can be produced to support the analysis of groundwater management.

Finally, the landscape units were defined, delimited, and applied to evaluate to what extent these landscape units could be integrated to support groundwater management taking into account the spatial and temporal dynamics (Chapter Six). This research applied as criteria to define the landscape units the following aspects: topographical/hydrological, hydrogeological, water demand, land use and census sector. Seven landscape units were delimited taking into account the internal resemblance of the criteria adopted. Two of these landscape units comprised the two subsystems of the groundwater system (phreatic and confined). Analysis of the groundwater budget in the landscape was conducted with the assistance of the numerical groundwater model tool. It was possible to simulate the groundwater recharge for a period of one year, including its distribution among the landscape units, as well as to identify the influence of the land use in the recharge of two adjacent landscape units. The application of landscape units also provided a whole-of-landscape integrated analysis of the spatial dynamics of the system that led to the identification of interbasin groundwater flow (IGF) from the Gramame River Basin to the Paraiba River Basin. This IGF occurs between two sets of landscape units (LU5 to LU1, and LU6 to LU2). A possible reason for this IGF is the different land uses and consequent characteristic water demand. This provided valuable information to support groundwater management. Furthermore, the temporal dynamics of the groundwater system was analysed applying a framework for the determination of the importance of the transience to groundwater management that uses the basin time constant as an indicator of the response time of the aquifer. The analyses were carried for each of the landscape units. Three landscape units presented a time constant of less than 30 years, indicating the changes, such as in land use, can have more than 60% of its disturbances adjusted within a management timeframe; a possible consequence would be alteration in the groundwater availability for allocation.

Therefore, the findings of this research confirmed the hypothesis. Landscape Scale Planning framework can underpin the groundwater management by the incorporation of land-use changes while aiding the provision of information. Given that: i) Landscape Scale Planning is intrinsically connected with land-use change and groundwater management; ii) Landscape Scale Planning can guide the groundwater management; iii) the analysis through landscape units can integrate land-use change aspects providing information to support groundwater management taking into account the spatial and temporal dynamics.

Further research on the topic can concentrate in the evaluation of groundwater management strategies comprised by all the three dimensions of Landscape Scale Planning, and the effects of the dynamic aspects included in the modification dimension on typical coastal groundwater management issues such as saltwater intrusion and submarine groundwater discharge. Regarding information, the application of data-fusion techniques capable of coupling the measured data with expert knowledge can significantly enhance the capacity to provide more reliable information to support groundwater management and overcome the data scarcity issues. This research has been restricted to the quantitative aspect of groundwater resources. The evaluation of the qualitative aspect can bring valuable insights. Finally, the landscape units themselves also have spatial and temporal dynamics. Landscape units can be determined according to evolving land-use changes, water demands and census sectors. The spatial distribution of such criteria changes throughout time. Therefore, landscape units could be able to capture both the groundwater system dynamics and changes along a timeline; with their evolution, from the past until possible futures, carrying information that could support the groundwater management.

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# Appendix A

**1. Delimitation of keywords**

Several query searches were conducted to analyse the sensitivity of the terms relative to i) groundwater and ii) landscape scale planning; and also to filter the results for publications linked to coastal areas. All the searches were initially made using the Scopus database. These searches were limited to reviews, peer review papers, book chapters in the English language up to December 2018. The fields searched were title, abstract and keywords. When a filter is applied to a result, the search comprises all other fields, as source title and references used, but not the full-text.

Relative to groundwater, the first query search used the terms: groundwater management, groundwater recharge, groundwater availability and groundwater sustainability. Relative to landscape scale planning the terms used were: agricultural planning, forest planning, regional planning, urban planning, land use planning, land use change, landscape scale, landscape scale planning. After the first search, the filter "Coastal" was applied (Table 1).

Table 1: Keywords applied in the first search

| Groundwater Management | Search | "Coastal" |
|---|---|---|
| Landscape Scale | 0 | 0 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 29 | 18 |
| Land Use Change | 26 | 13 |
| Urban Planning | 10 | 1 |
| Regional Planning | 13 | 2 |
| Forest Planning | 0 | 0 |
| Agricultural Planning | 1 | 0 |
| Total (removing duplicates) | 76 | 34 |

| Groundwater Recharge | Search | "Coastal" |
|---|---|---|
| Landscape Scale | 14 | 3 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 28 | 5 |
| Land Use Change | 194 | 43 |
| Urban Planning | 24 | 8 |
| Regional Planning | 24 | 8 |
| Forest Planning | 0 | 0 |
| Agricultural Planning | 2 | 1 |
| Total (removing duplicates) | 266 | 32 |

| Groundwater Sustainability | Search | "Coastal" |
|---|---|---|
| Landscape Scale | 0 | 0 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 6 | 3 |
| Land Use Change | 6 | 3 |
| Urban Planning | 0 | 0 |
| Regional Planning | 0 | 0 |
| Forest Planning | 0 | 0 |
| Agricultural Planning | 0 | 0 |
| Total (removing duplicates) | 10 | 4 |

| Groundwater Availability | Search | "Coastal" |
|---|---|---|
| Landscape Scale | 1 | 1 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 1 | 0 |
| Land Use Change | 14 | 1 |
| Urban Planning | 3 | 1 |
| Regional Planning | 4 | 3 |
| Forest Planning | 0 | 0 |
| Agricultural Planning | 0 | 0 |
| Total (removing duplicates) | 23 | 6 |

Considering the two terms that provided more results (groundwater management and groundwater recharge), the total unique results were 333. The concept of groundwater management comprises the terms groundwater sustainability and groundwater availability, and because few results were obtained, these terms were removed. The terms Urban Planning, Regional Planning, Forest Planning, and Agricultural Planning are more related to spatial subdivision of territories than to specific concept under the perspective of Landscape Scale Planning. These terms were then replaced by spatial planning, which presents a stronger correlation with this perspective.

In order to make the search broader, the subsequent query searches applied the terms “Management” and “Planning” in alternation. An exception was made to Landscape Scale Planning, once this concept refers to a specific approach. Hence, for groundwater management the keywords became: groundwater management, groundwater planning and groundwater recharge; and for landscape scale planning: landscape scale, landscape scale planning, land use planning, land use management, land use change, spatial planning, spatial management, spatial change (Table 2).

Table 2 – Second search of keywords

| Groundwater Management | Total | "Coastal" |
|---|---|---|
| Landscape Scale | 0 | 0 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 29 | 18 |
| Land Use Change | 26 | 13 |
| Land Use Management | 8 | 3 |
| Spatial Planning | 2 | 1 |
| Spatial Management | 0 | 0 |
| Spatial Change | 0 | 0 |
| Total (removing duplicates) | 59 | 31 |

| Groundwater Recharge | Total | "Coastal" |
|---|---|---|
| Landscape Scale | 14 | 3 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 28 | 5 |
| Land Use Change | 194 | 43 |
| Land Use Management | 13 | 1 |
| Spatial Planning | 5 | 1 |
| Spatial Management | 0 | 0 |
| Spatial Change | 4 | 2 |
| Total (removing duplicates) | 241 | 50 |

| Groundwater Planning | Total | "Coastal" |
|---|---|---|
| Landscape Scale | 0 | 0 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 2 | 0 |
| Land Use Change | 0 | 0 |
| Land Use Management | 0 | 0 |
| Spatial Planning | 1 | 0 |
| Spatial Management | 0 | 0 |
| Spatial Change | 0 | 0 |
| Total (removing duplicates) | 3 | 0 |

The terms related to groundwater were conceptually expanded to a higher level. Another search was conducted taking into account the terms relative to water: water resources planning, water resources management, water planning and water management; afterwards the filter “groundwater” was applied to the results obtained (results in the second column). Then, the filter “coastal” was applied (third column) (Table 3).

Table 3 – Third search of groundwater

| Water Management | "GW" | "Coastal" |
|---|---|---|
| Landscape Scale | 19 | 7 |
| Landscape Scale Planning | 1 | 0 |
| Land Use Planning | 128 | 51 |
| Land Use Change | 325 | 115 |
| Land Use Management | 32 | 11 |
| Spatial Planning | 26 | 9 |
| Spatial Management | 2 | 0 |
| Spatial Change | 16 | 5 |
| Total (removing duplicates) | 508 | 181 |

| Water Resources Planning | "GW" | "Coastal" |
|---|---|---|
| Landscape Scale | 0 | 0 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 5 | 0 |
| Land Use Change | 20 | 4 |
| Land Use Management | 3 | 1 |
| Spatial Planning | 1 | 0 |
| Spatial Management | 0 | 0 |
| Spatial Change | 0 | 0 |
| Total (removing duplicates) | 28 | 5 |

| Water Planning | "GW" | "Coastal" |
|---|---|---|
| Landscape Scale | 3 | 1 |
| Landscape Scale Planning | 0 | 0 |
| Land Use Planning | 27 | 8 |
| Land Use Change | 34 | 9 |
| Land Use Management | 2 | 0 |
| Spatial Planning | 7 | 1 |
| Spatial Management | 0 | 0 |
| Spatial Change | 0 | 0 |
| Total (removing duplicates) | 65 | 18 |

| Water Resources Management | "GW" | "Coastal" |
|---|---|---|
| Landscape Scale | 4 | 1 |
| Landscape Scale Planning | 1 | 0 |
| Land Use Planning | 33 | 12 |
| Land Use Change | 106 | 35 |
| Land Use Management | 8 | 1 |
| Spatial Planning | 5 | 2 |
| Spatial Management | 1 | 0 |
| Spatial Change | 4 | 0 |
| Total (removing duplicates) | 147 | 47 |

The following flowchart (Figure 1 and 2) depicts the grouping of results gathered when applying terms relative to water resources and groundwater with the three groups of terms relative to Landscape Scale Planning, before and after the application of the filters 'groundwater' and 'coastal'. There is a higher concentration of publications regarding land use (especially in planning and change) than when the other terms were applied. The term "landscape scale planning" did not generate any results when searched in combination with "groundwater management", "groundwater planning" or "groundwater recharge". When searched in combination with the second group, "water" and variations, two results appeared. However, these papers did not focus directly on the concepts proposed, or referenced, by Selman (2006).

Therefore, the selected keywords to conduct the systematic literature review were: Relative to "Groundwater management", it was also adopted "Groundwater Recharge". Relative to "Landscape Scale Planning", the terms were: "landscape scale", "land use planning", and "land use change". These keywords were joined in the query searches using Boolean operators. Afterwards, the filter "coastal" was applied to search for the presence of the term among the titles, abstracts and keywords. These searches were done using the Scopus and Web of Science (Table 4) database, until December 15$^{th}$ 2018.

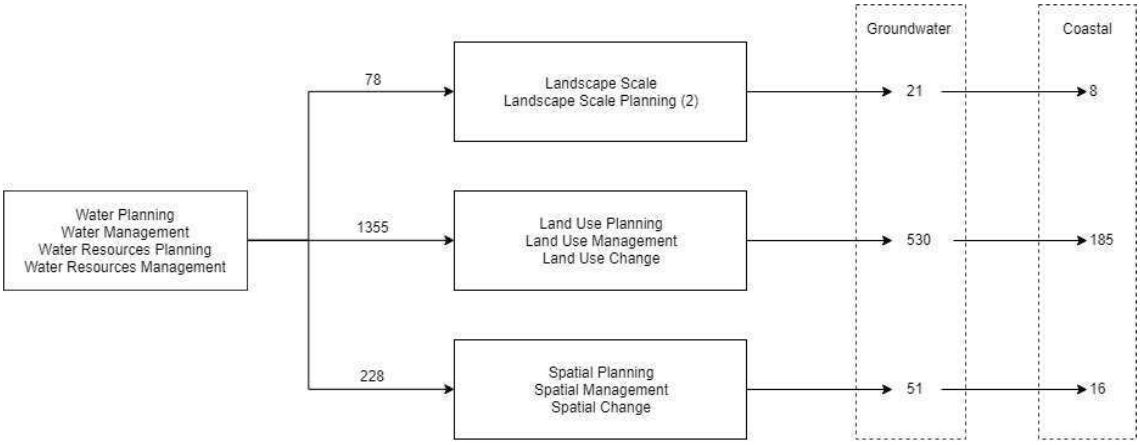



Figure 9.2 - Results of the terms referred as "water"

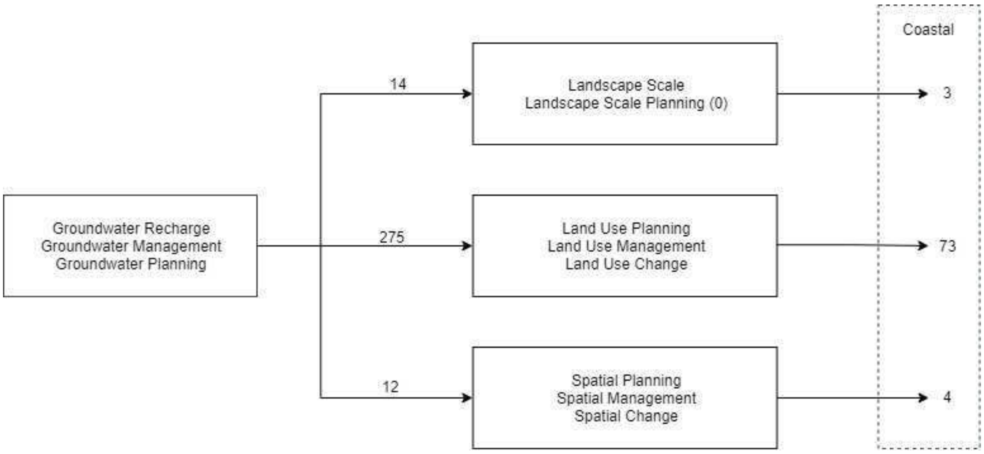



Figure 9.1 - Results of the terms referred as “groundwater”

Table 4 – Databases, strings for query and results

| Database | String |
|---|---|
| Scopus | TITLE-ABS-KEY (("Groundwater Management" OR "Groundwater Recharge”) AND ("Landscape Scale” OR "Land Use Planning" OR "Land Use Change")) |
| Web of Science | TS = ( ( "Groundwater Management"  OR  "Groundwater Recharge" )  AND  ( "Landscape Scale"  OR  "Land Use Planning"  OR  "Land Use Change")) |

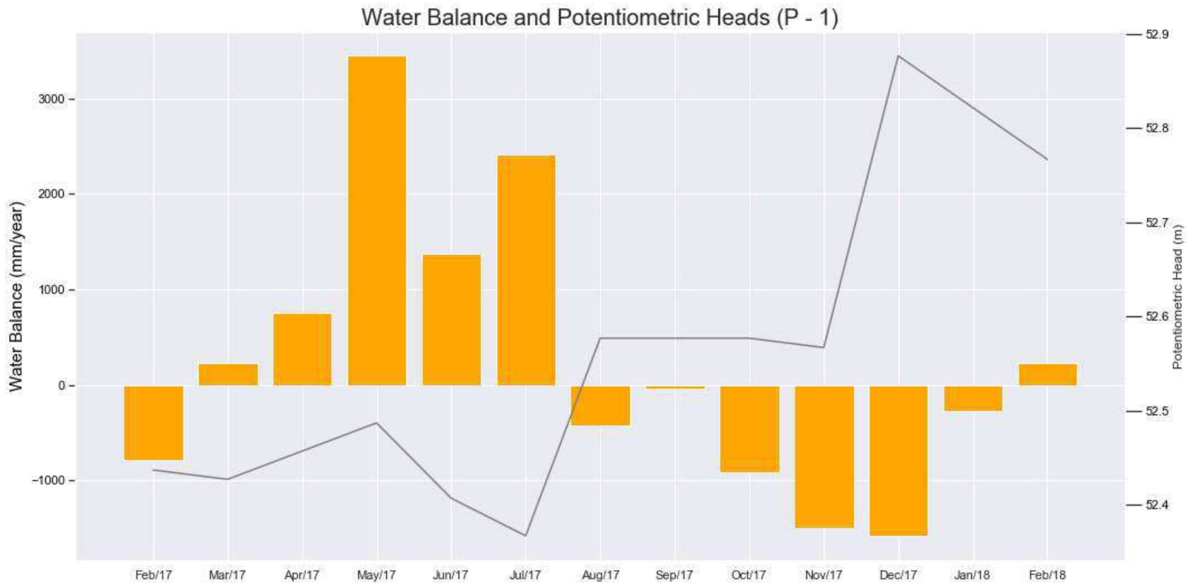
**2. Selection of the publications for analysis**

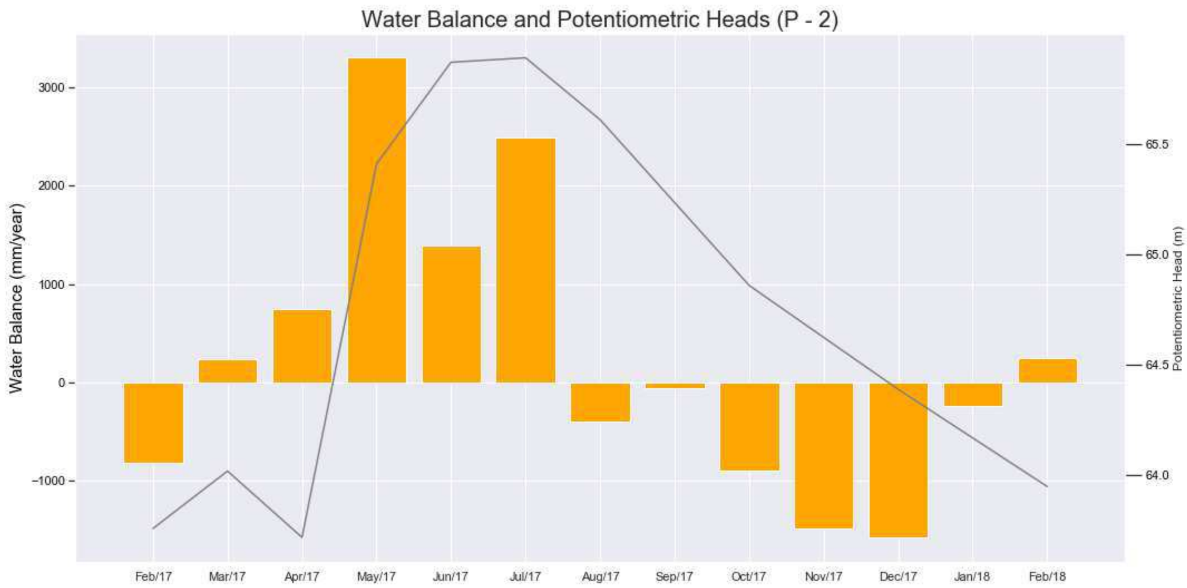
The searches using the determined keywords generated a total of 480 results. In the Scopus database, the search resulted in 278 documents, in the Web of Science the results were 202. After the application of the filter “coastal”, the number of results was narrowed down to 55 documents, among which 30 were unique results. These were subjected to the screening.

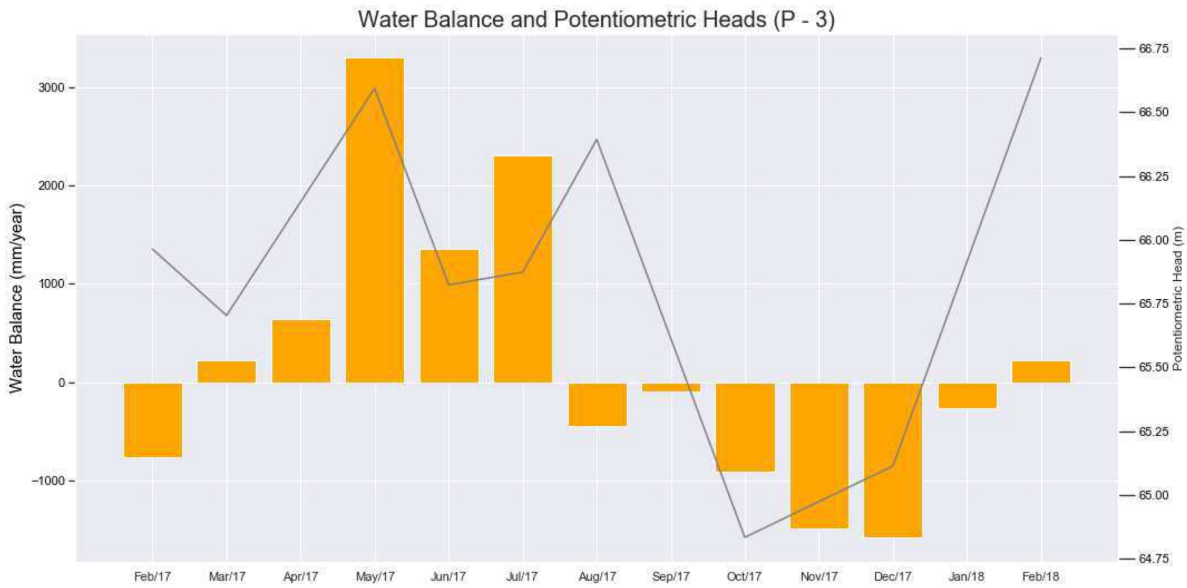
The screening was conducted through title and abstract evaluating the following criteria: groundwater management being one of the objectives of the article, and land use patterns were analysed. Only one article had to be excluded after the screening because it did not fit any of the criteria. Afterwards, a content analysis was conducted to determine the eligibility of the publications; this step was applied in the full-text of the selected records. The purpose of this analysis was to evaluate which publications investigated the aspects relative to the research question. Therefore, two criteria were adopted: at least one of the three landscape scale planning dimensions was analysed in the paper; land use presented at least one non-natural type of use. Only one record was excluded in this step given that the full-text of the publication was not available. Finally, the number of publications selected were 28 (see Figure 2.1 - PRISMA Flowchart).

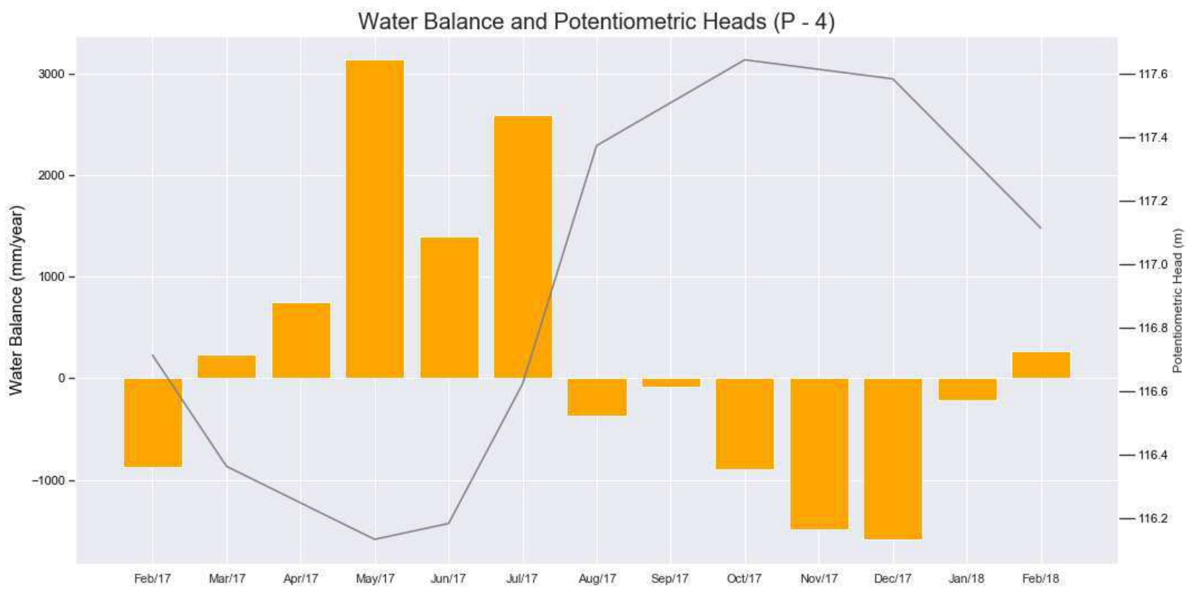
# Appendix B

**1. Figures: Water Balance and Potentiometric Heads**




Water Balance and Potentiometric Heads (P - 3)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
66.75
66.50
66.25
66.00
65.75
65.50
65.25
65.00
64.75
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 4)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
117.6
117.4
117.2
117.0
116.8
116.6
116.4
116.2
Feb/17
Mar/17
Apr/17
May/17
Jun/17
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Aug/17
Sep/17
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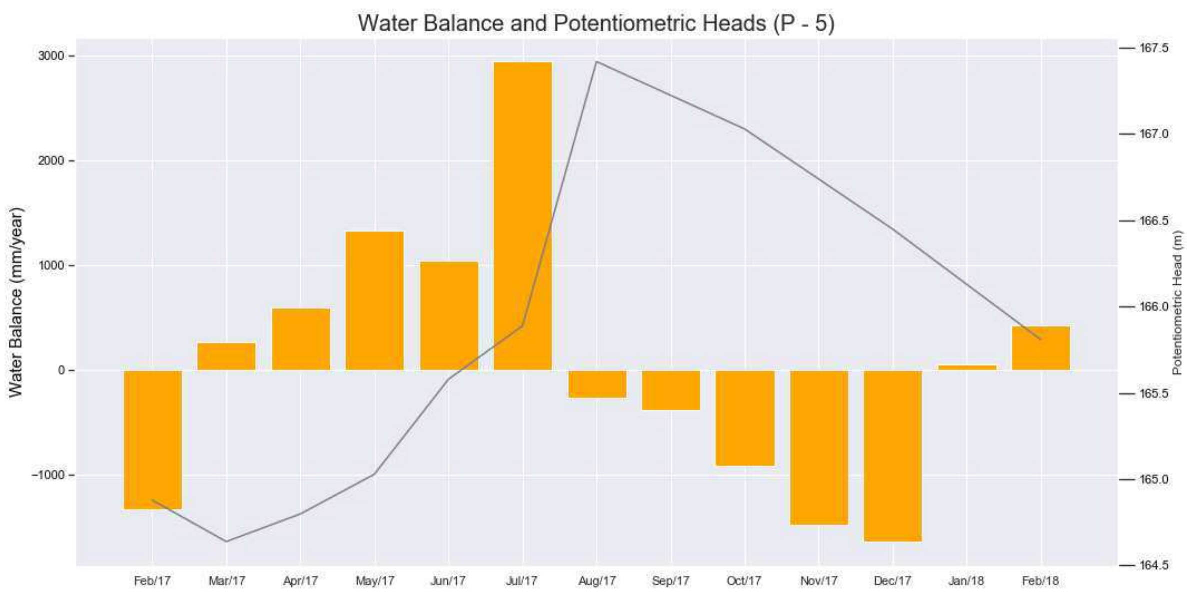
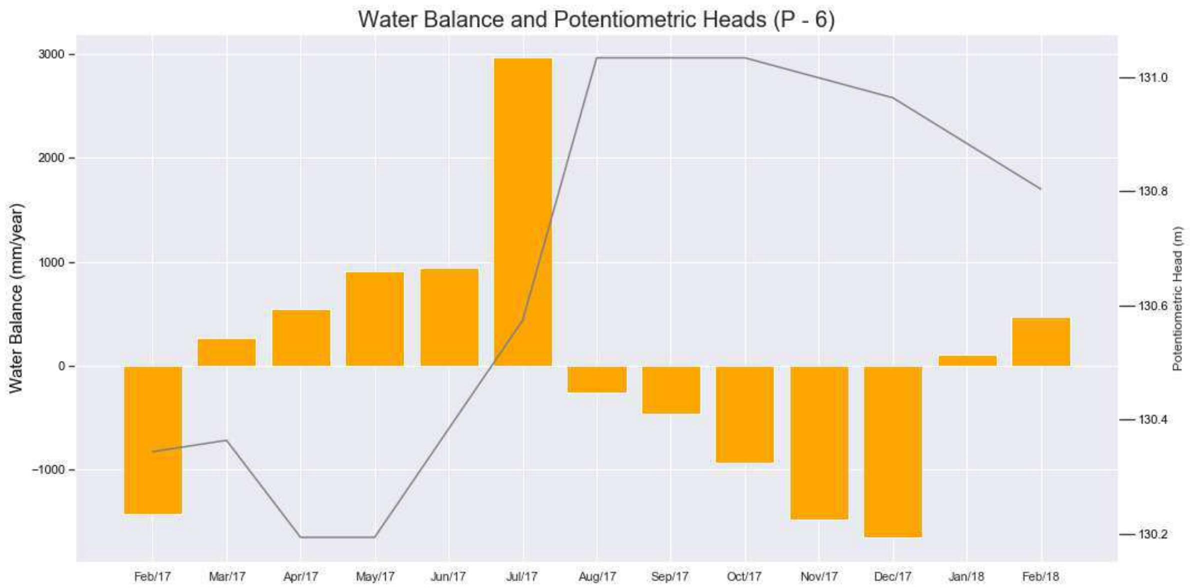
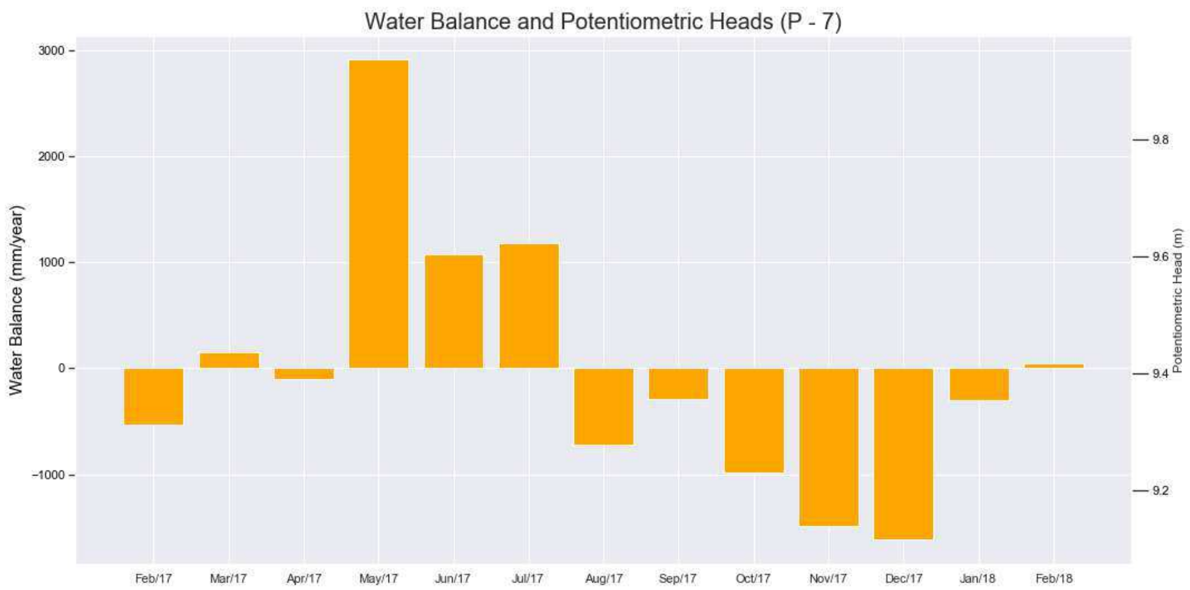
Nov/17
Dec/17
Jan/18
Feb/18

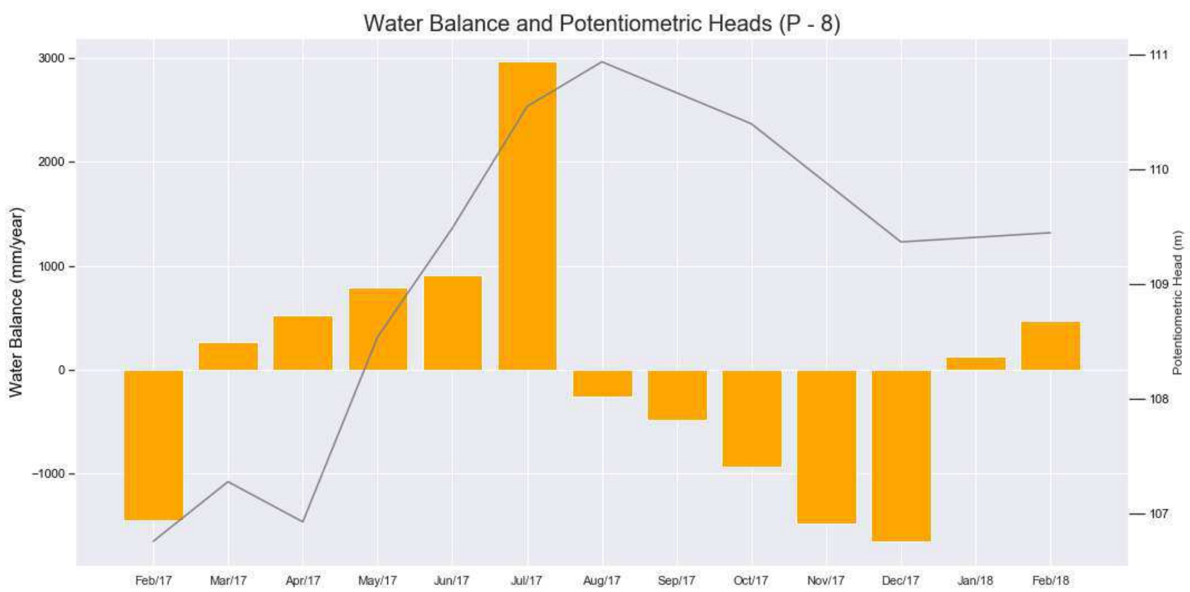

Water Balance and Potentiometric Heads (P - 5)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
167.5
167.0
166.5
166.0
165.5
165.0
164.5
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18



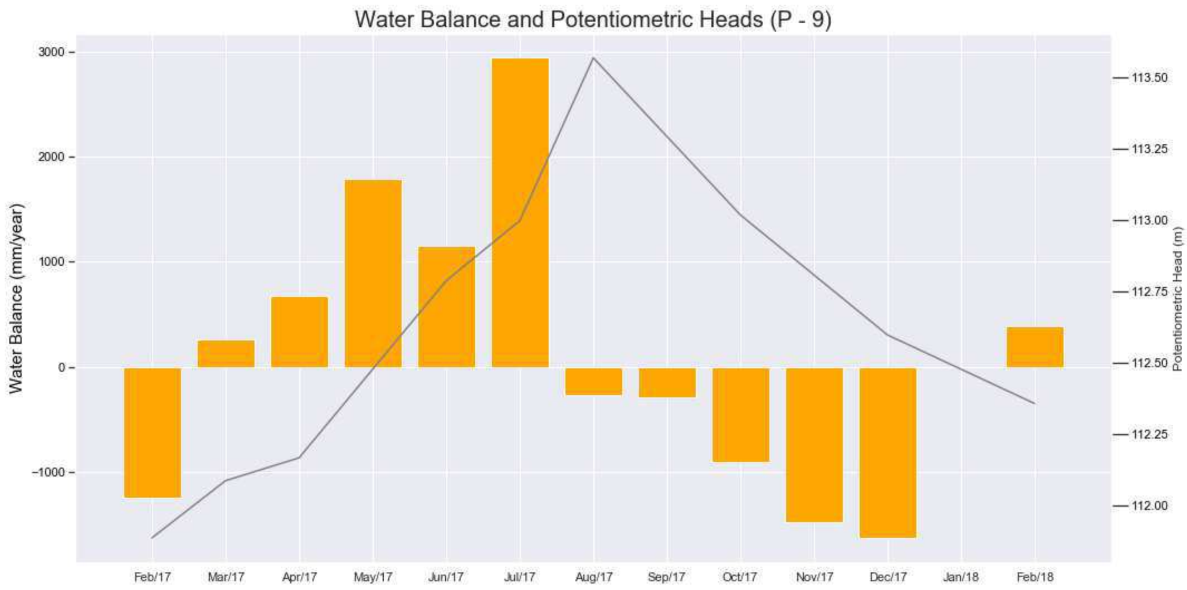
Water Balance and Potentiometric Heads (P - 6)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
131.0
130.8
130.6
130.4
130.2
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

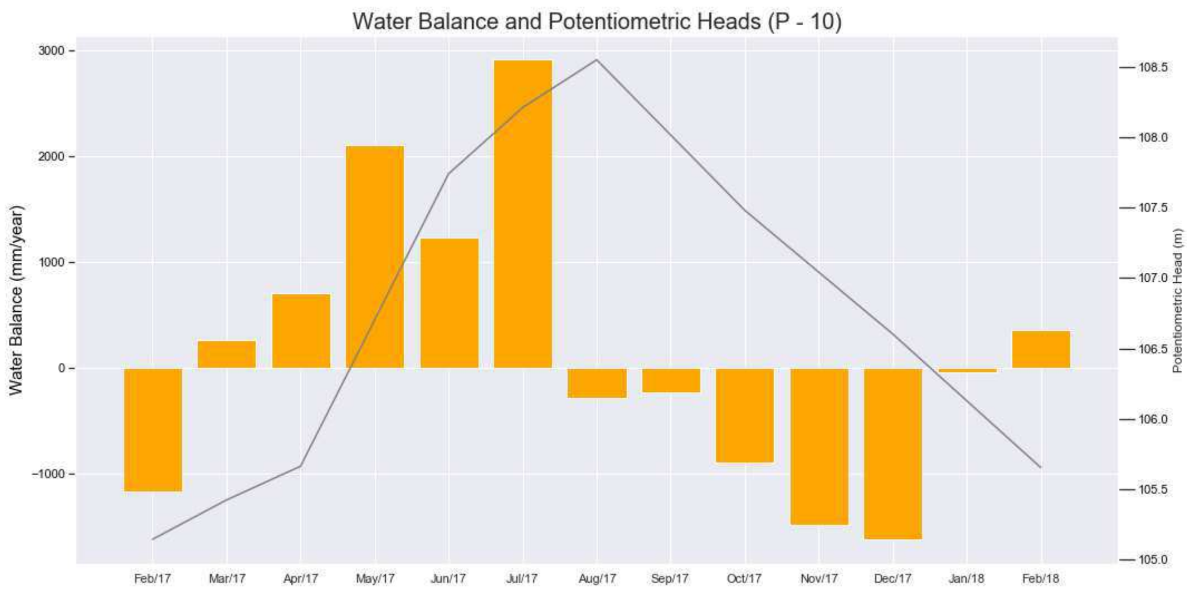



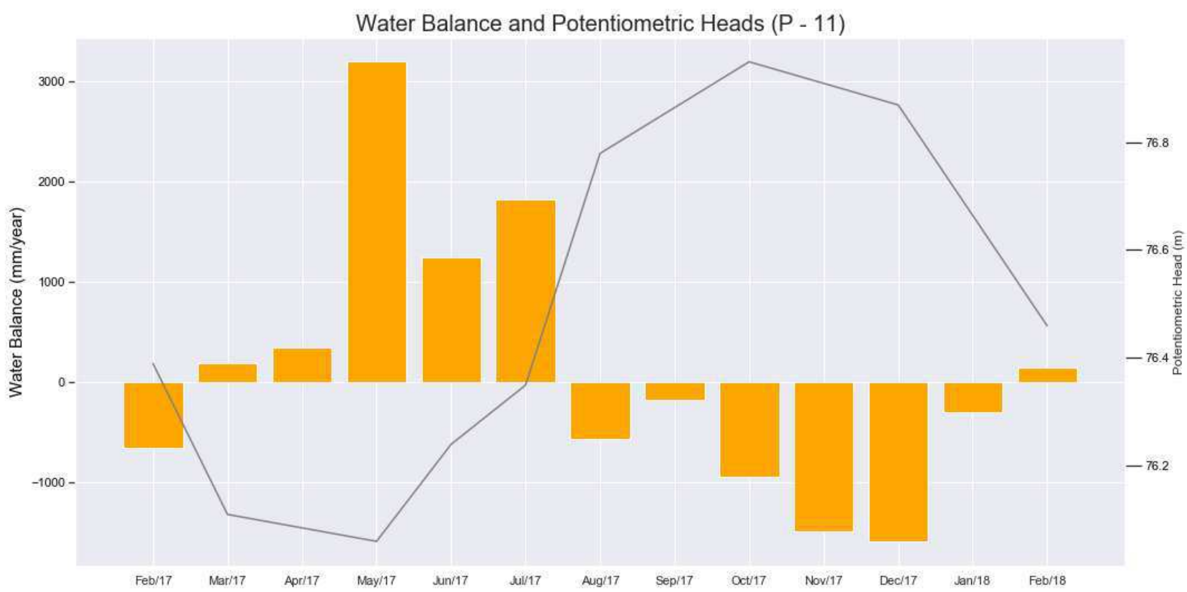
Water Balance and Potentiometric Heads (P - 7)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
9.8
9.6
9.4
9.2
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

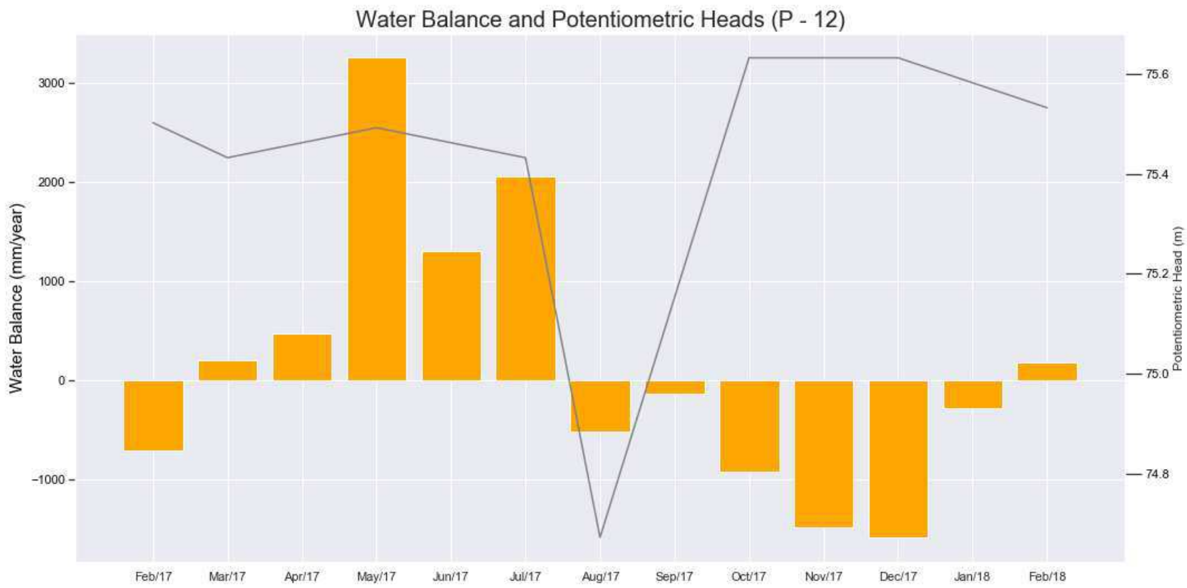



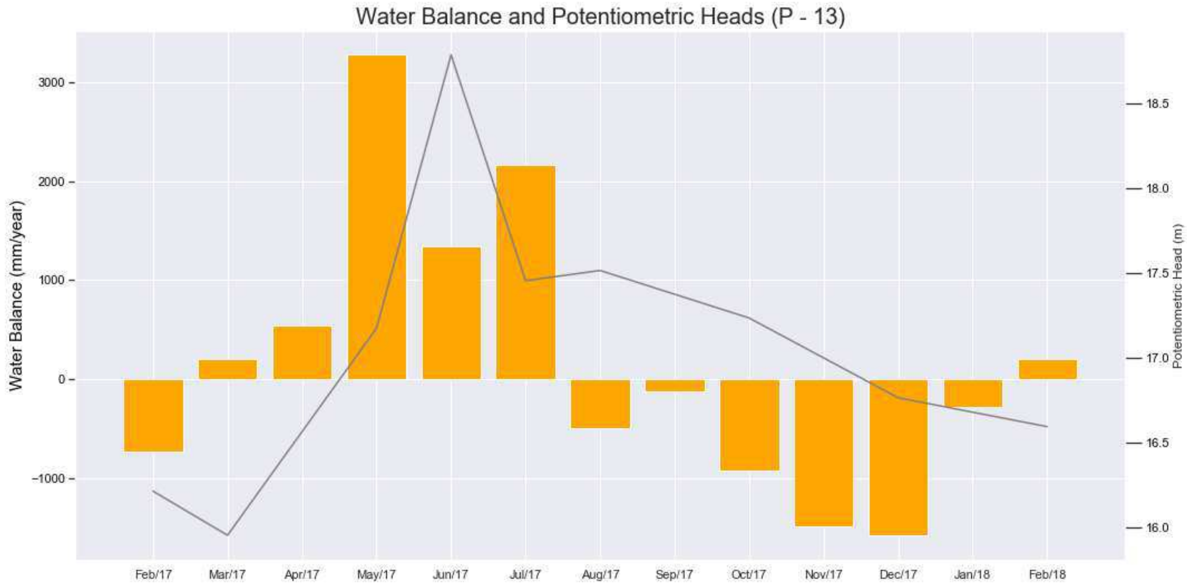
Water Balance and Potentiometric Heads (P - 8)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
111
110
109
108
107
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

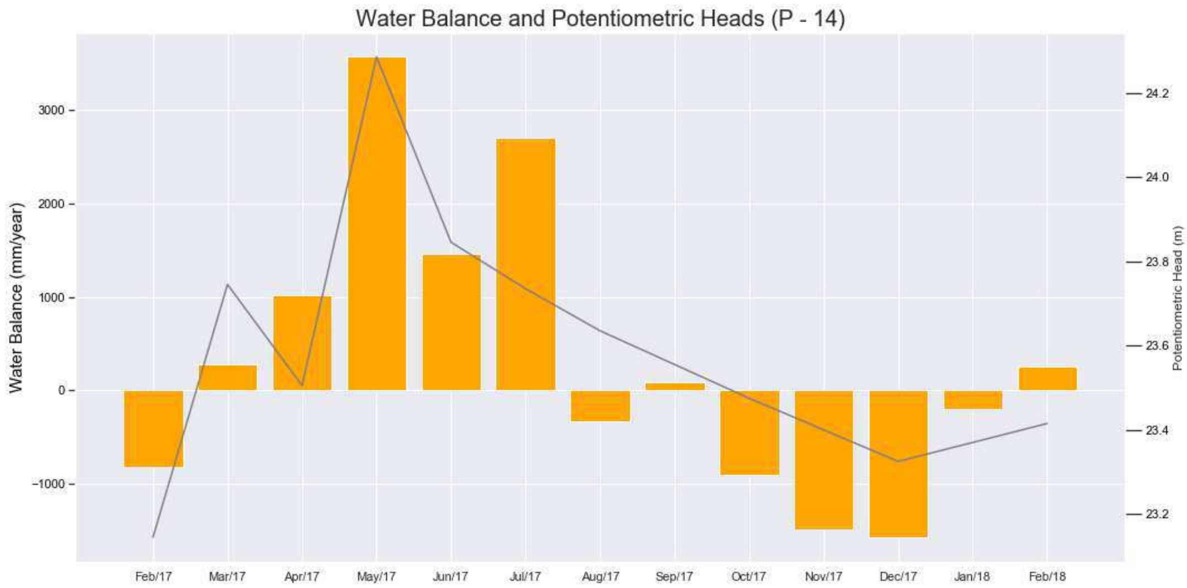

Water Balance and Potentiometric Heads (P - 9)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
113.50
113.25
113.00
112.75
112.50
112.25
112.00
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

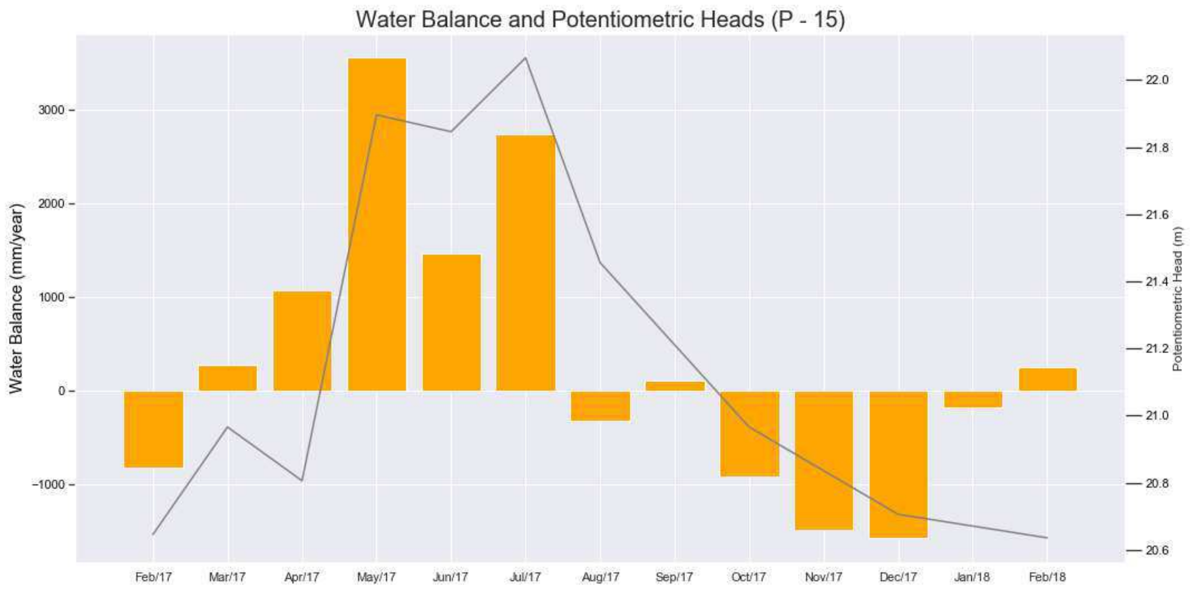

Water Balance and Potentiometric Heads (P - 10)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
108.5
108.0
107.5
107.0
106.5
106.0
105.5
105.0
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 11)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
76.8
76.6
76.4
76.2
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 12)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
75.6
75.4
75.2
75.0
74.8
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 13)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
18.5
18.0
17.5
17.0
16.5
16.0
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 14)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
24.2
24.0
23.8
23.6
23.4
23.2
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 15)
Water Balance (mm/year)
Potentiometric Head (m)
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
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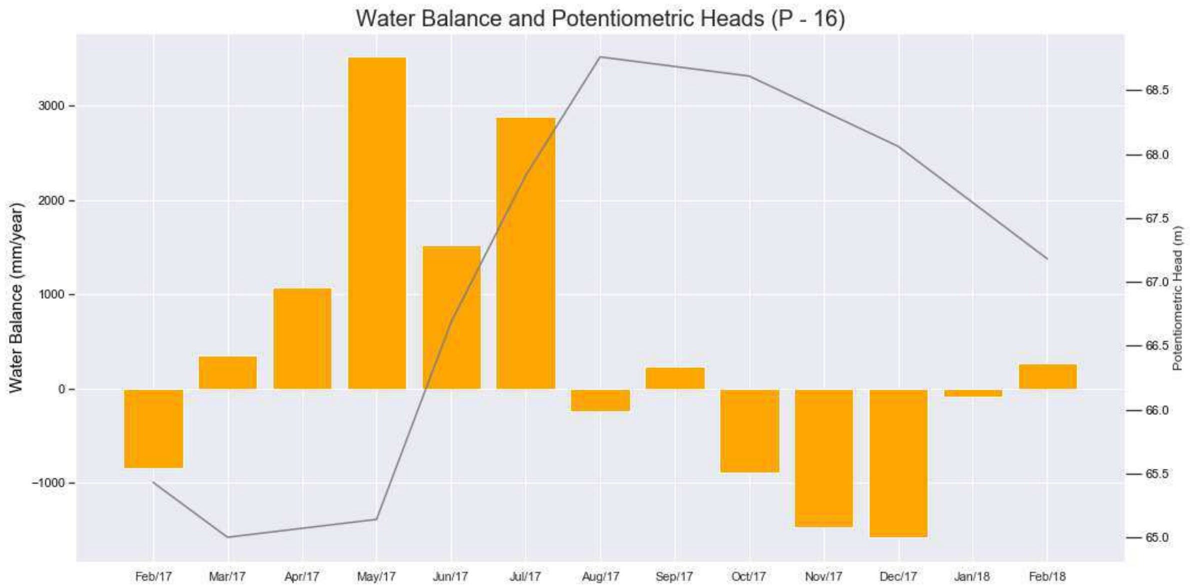
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 16)
Water Balance (mm/year)
Potentiometric Head (m)
Feb/17
Mar/17
Apr/17
May/17
Jun/17
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Sep/17
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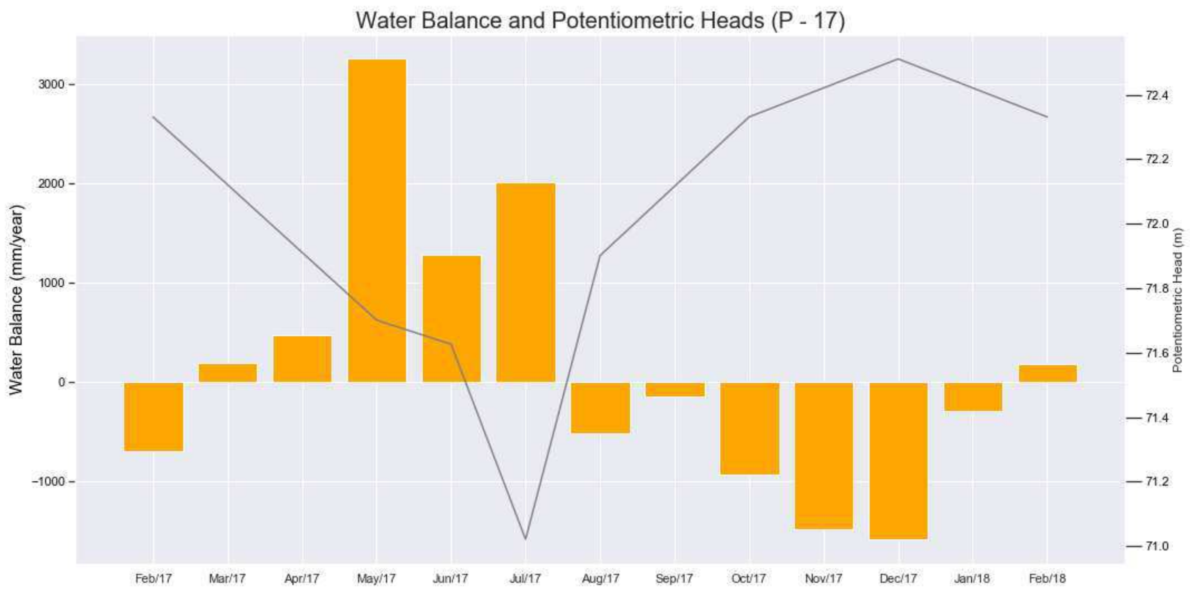
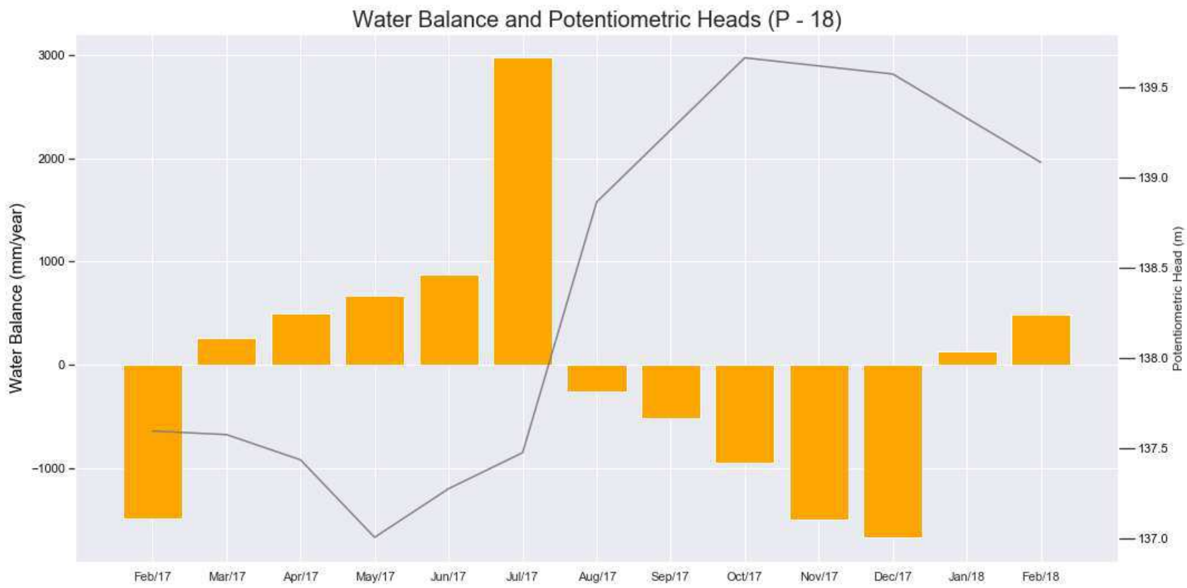
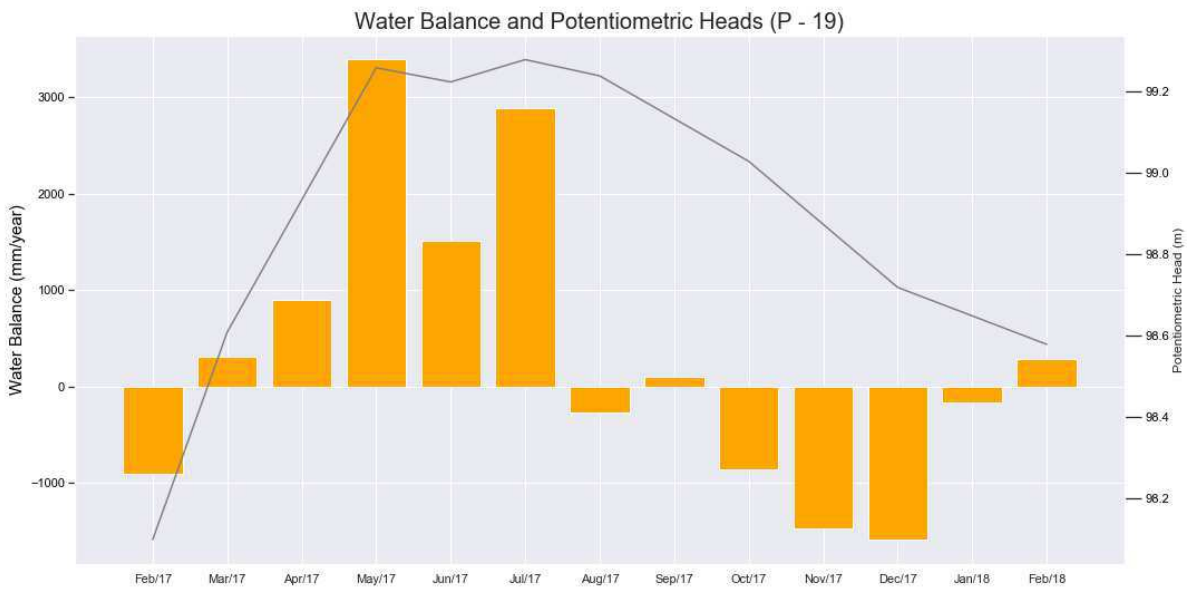
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 17)
Water Balance (mm/year)
Potentiometric Head (m)
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 18)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
139.5
139.0
138.5
138.0
137.5
137.0
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 19)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
99.2
99.0
98.8
98.6
98.4
98.2
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
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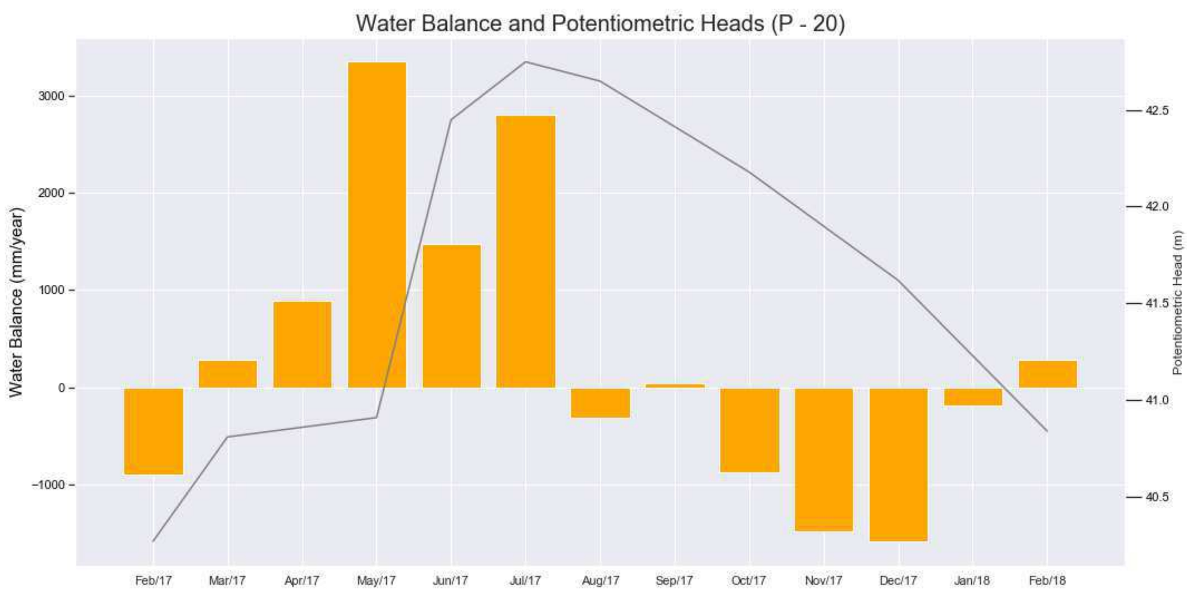
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 20)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
42.5
42.0
41.5
41.0
40.5
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
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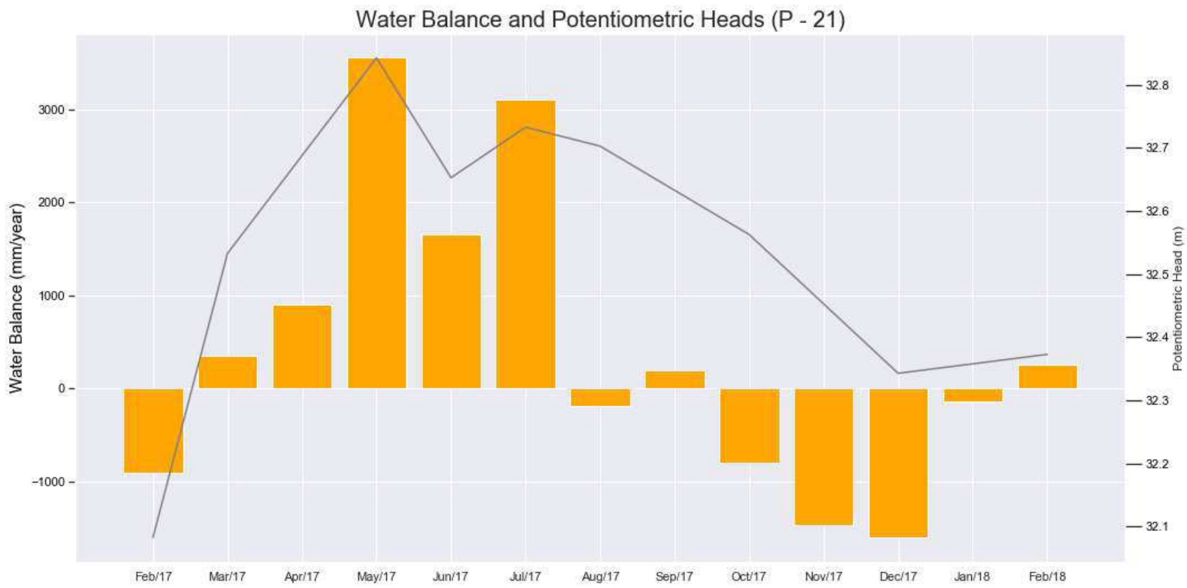
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 21)
Water Balance (mm/year)
Potentiometric Head (m)
Feb/17
Mar/17
Apr/17
May/17
Jun/17
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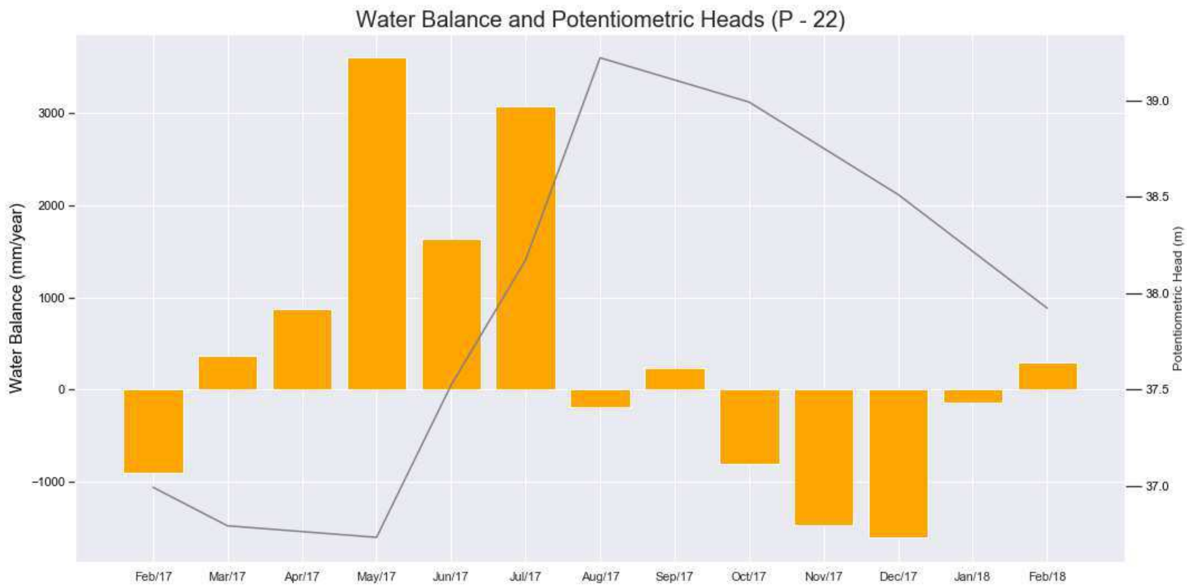
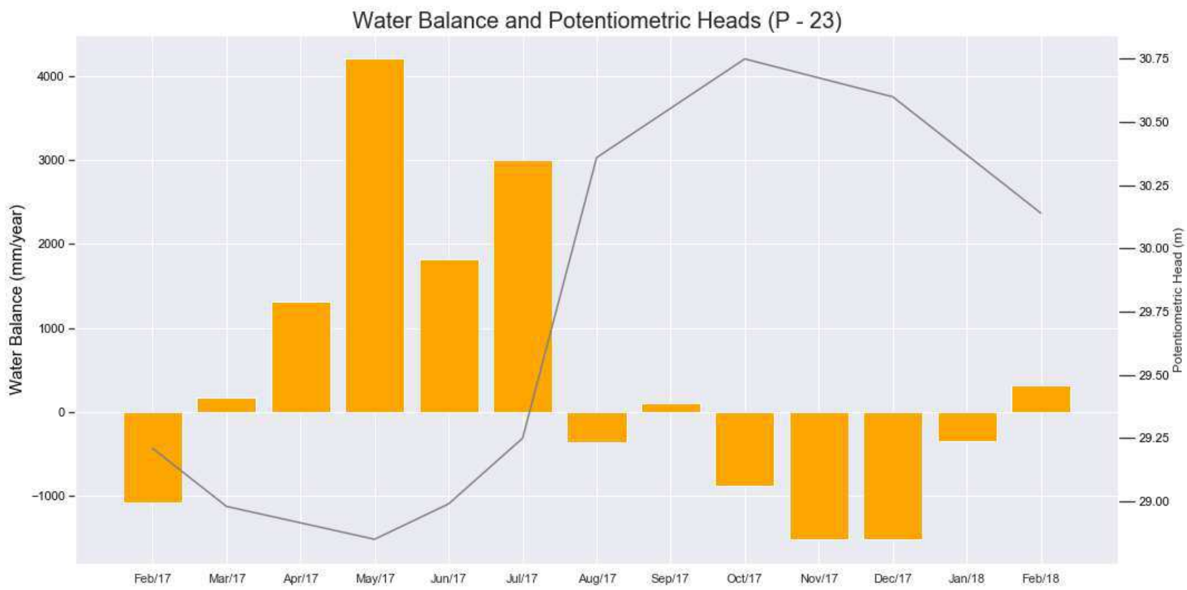
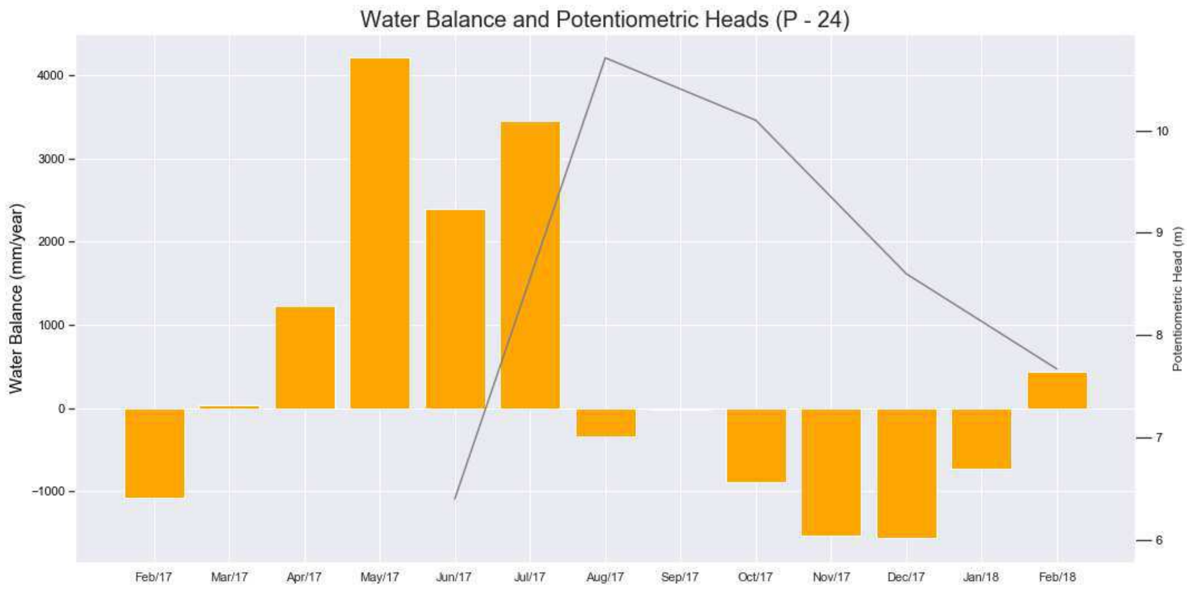
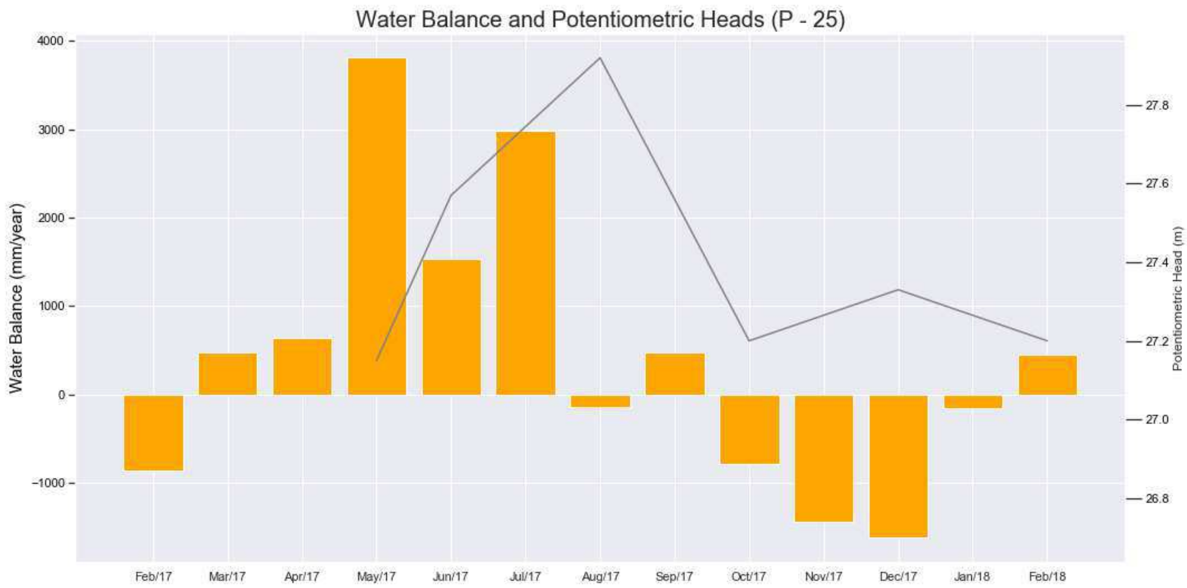
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 22)
Water Balance (mm/year)
Potentiometric Head (m)
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

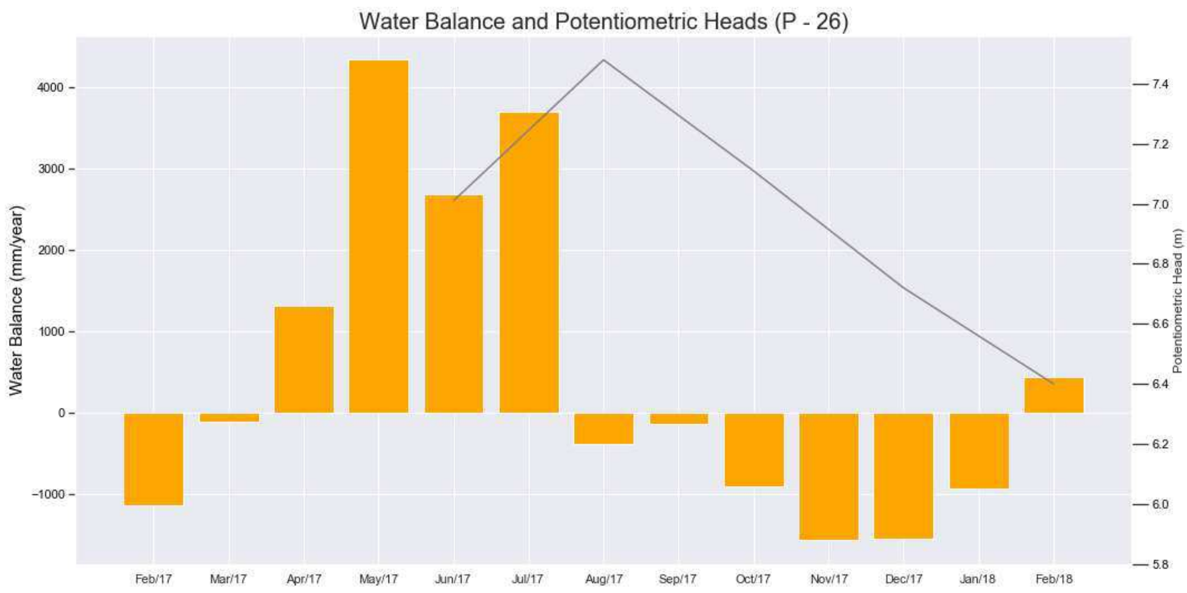

Water Balance and Potentiometric Heads (P - 23)
Water Balance (mm/year)
Potentiometric Head (m)
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18



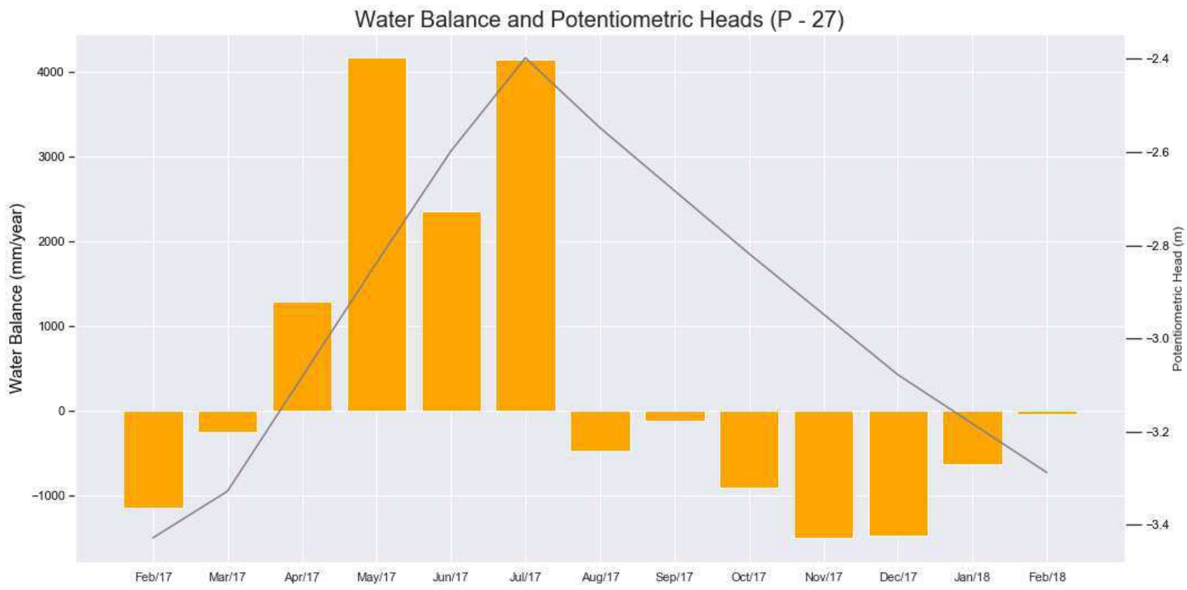
Water Balance and Potentiometric Heads (P - 24)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
10
9
8
7
6
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

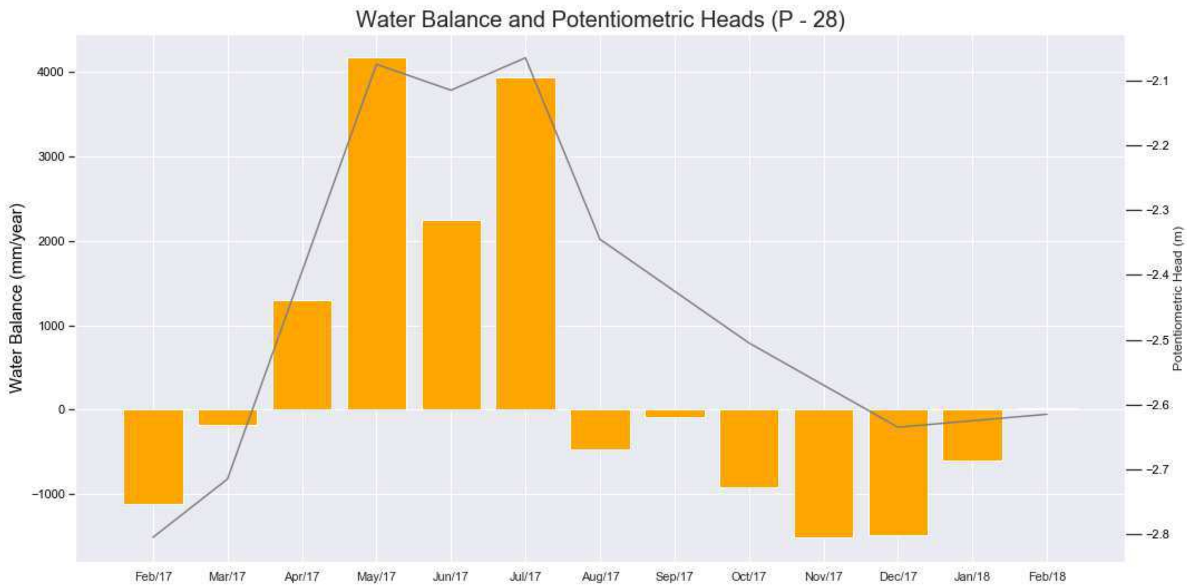
Water Balance and Potentiometric Heads (P - 25)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
27.8
27.6
27.4
27.2
27.0
26.8
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

Water Balance and Potentiometric Heads (P - 26)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
7.4
7.2
7.0
6.8
6.6
6.4
6.2
6.0
5.8
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18



Water Balance and Potentiometric Heads (P - 27)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
−2.4
−2.6
−2.8
−3.0
−3.2
−3.4
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

Water Balance and Potentiometric Heads (P - 28)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
−2.1
−2.2
−2.3
−2.4
−2.5
−2.6
−2.7
−2.8
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
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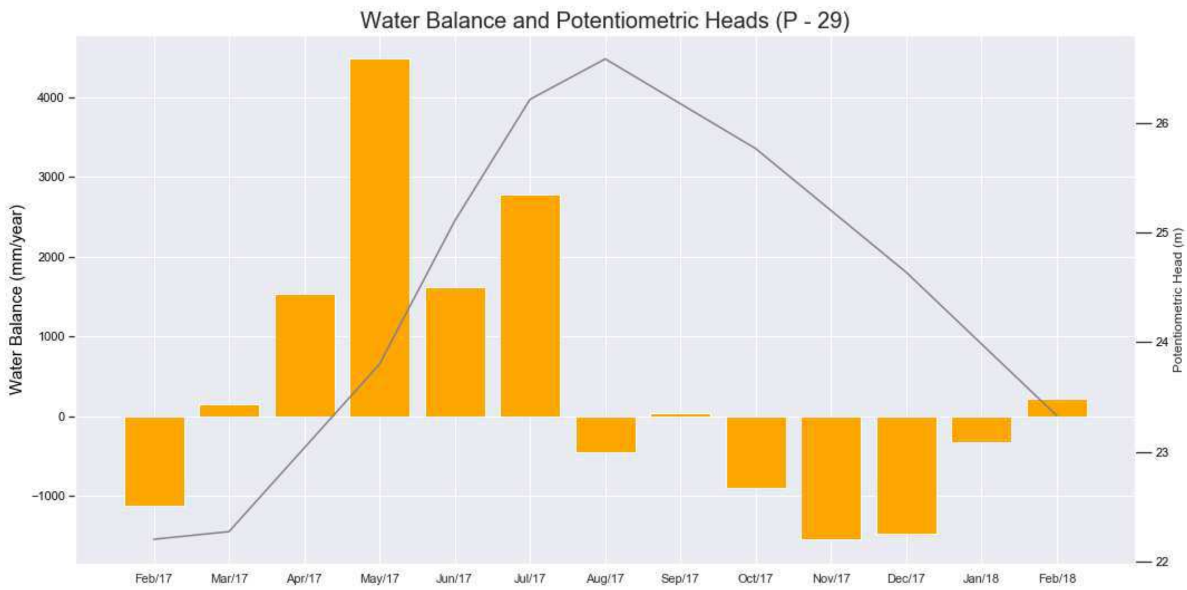
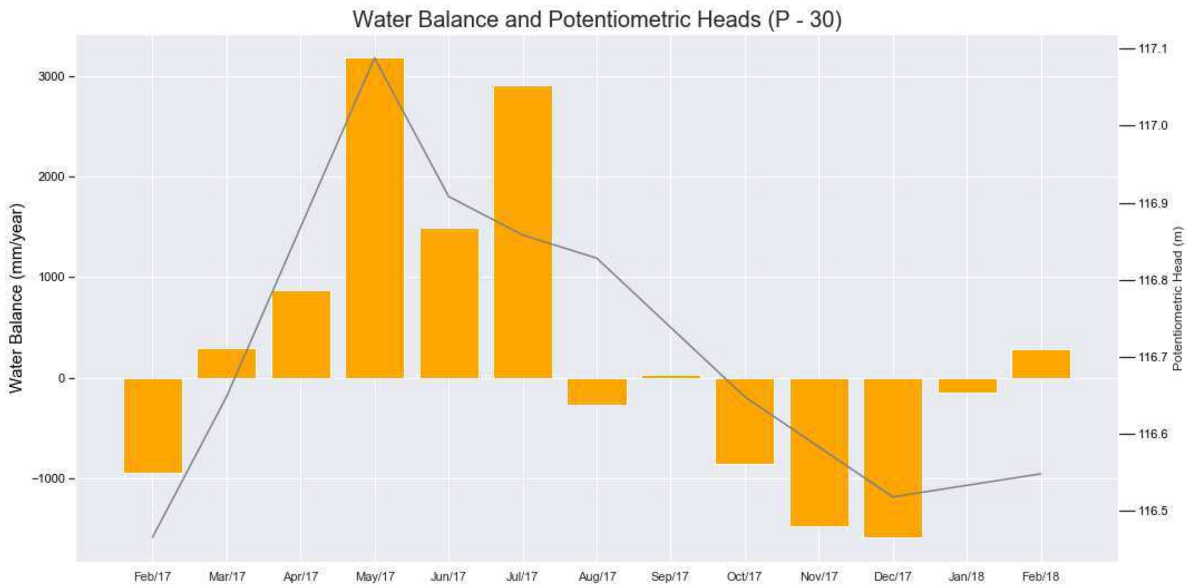
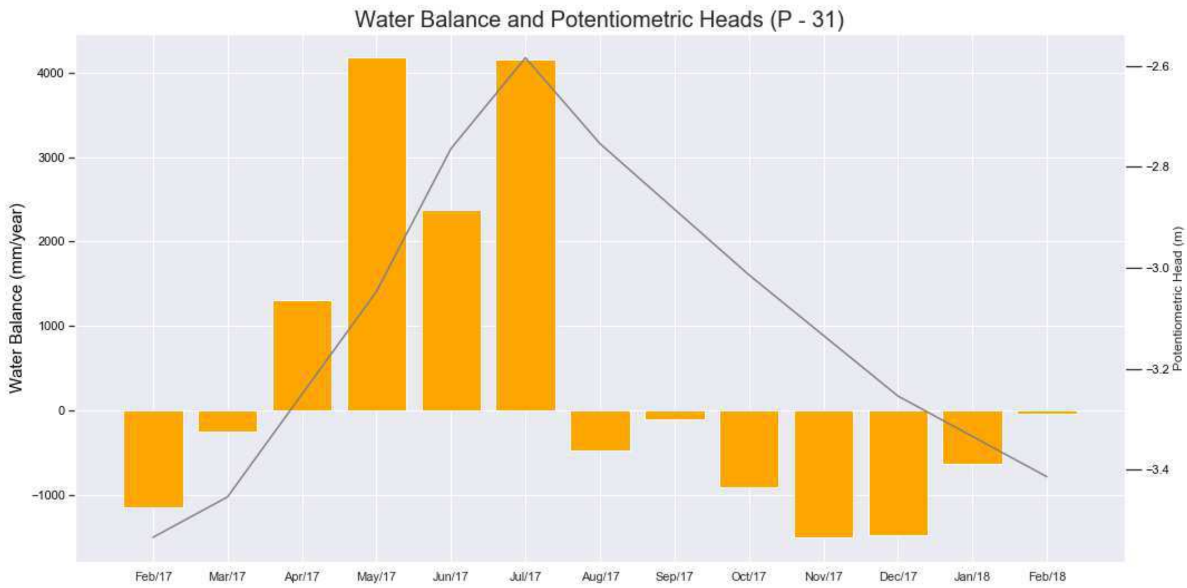
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18

Water Balance and Potentiometric Heads (P - 29)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
26
25
24
23
22
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 30)
Water Balance (mm/year)
Potentiometric Head (m)
3000
2000
1000
0
−1000
117.1
117.0
116.9
116.8
116.7
116.6
116.5
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
Feb/18


Water Balance and Potentiometric Heads (P - 31)
Water Balance (mm/year)
Potentiometric Head (m)
4000
3000
2000
1000
0
−1000
−2.6
−2.8
−3.0
−3.2
−3.4
Feb/17
Mar/17
Apr/17
May/17
Jun/17
Jul/17
Aug/17
Sep/17
Oct/17
Nov/17
Dec/17
Jan/18
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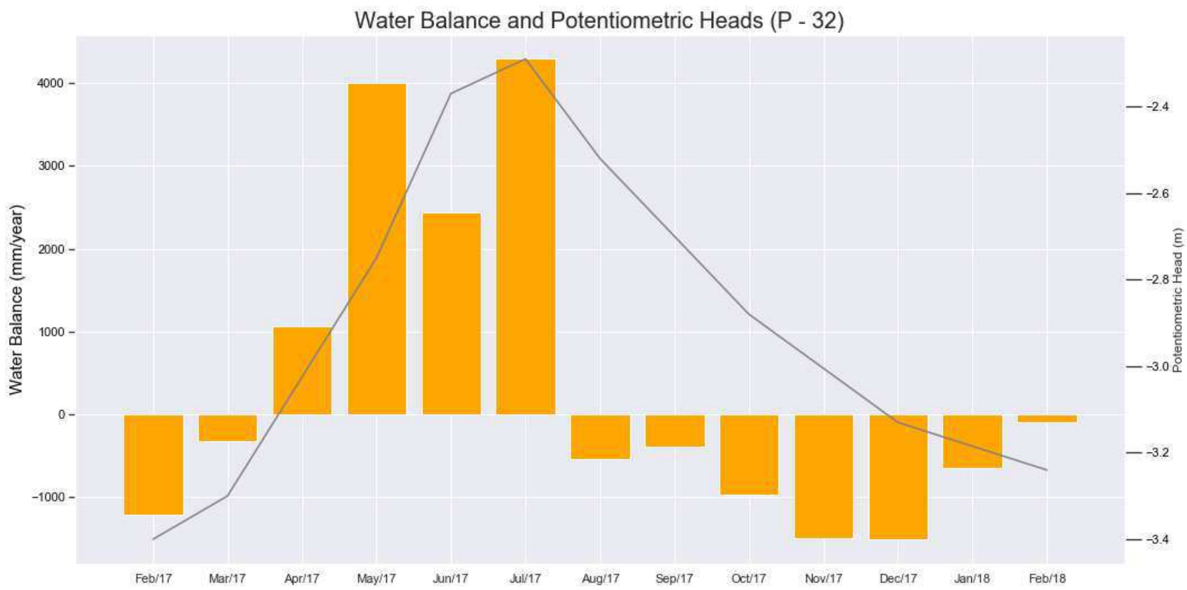
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Potentiometric Head (m)
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3000
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−2.6
−2.8
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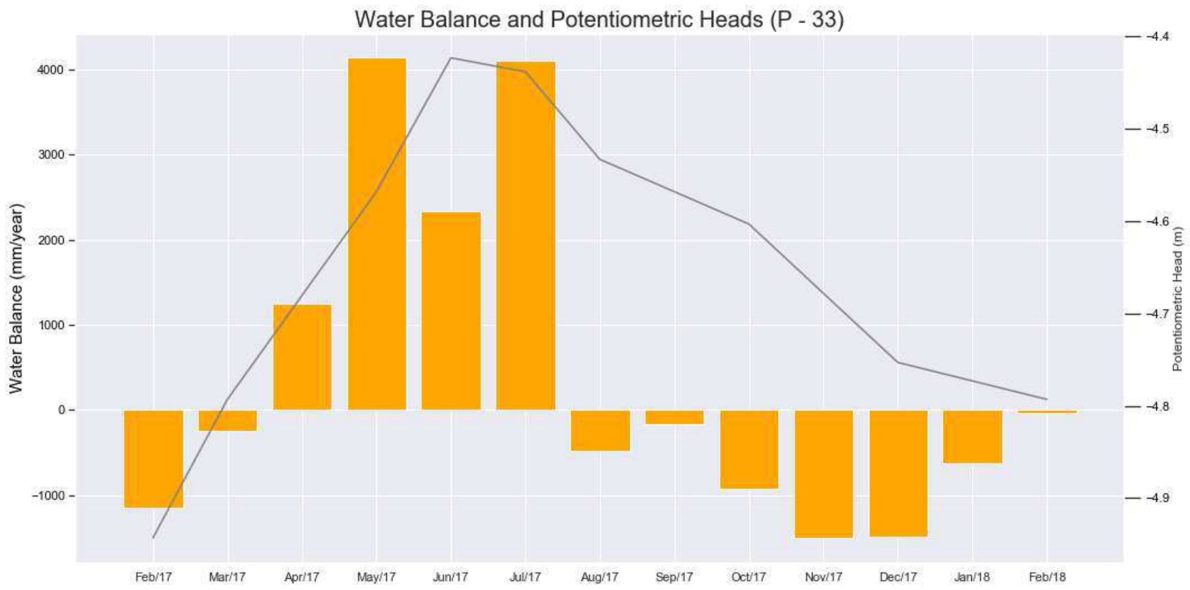
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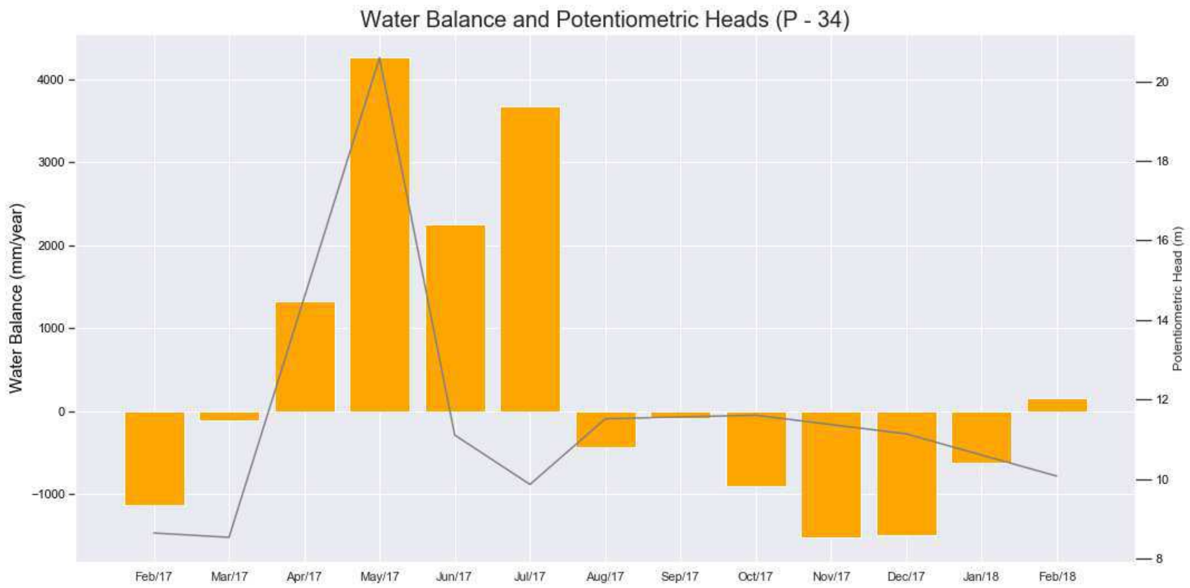
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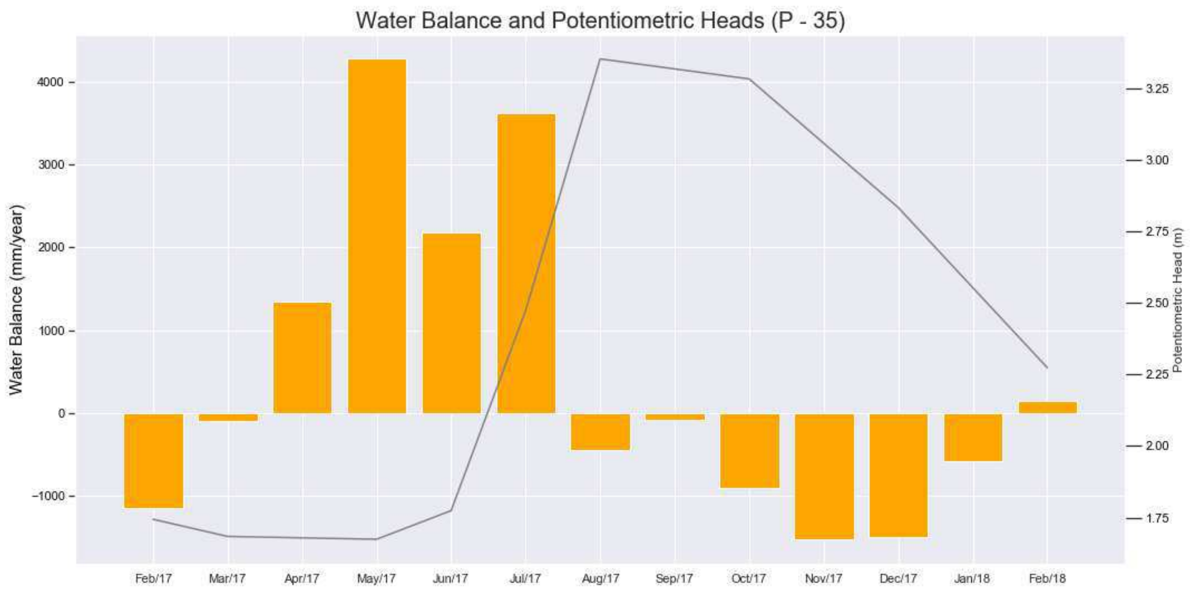
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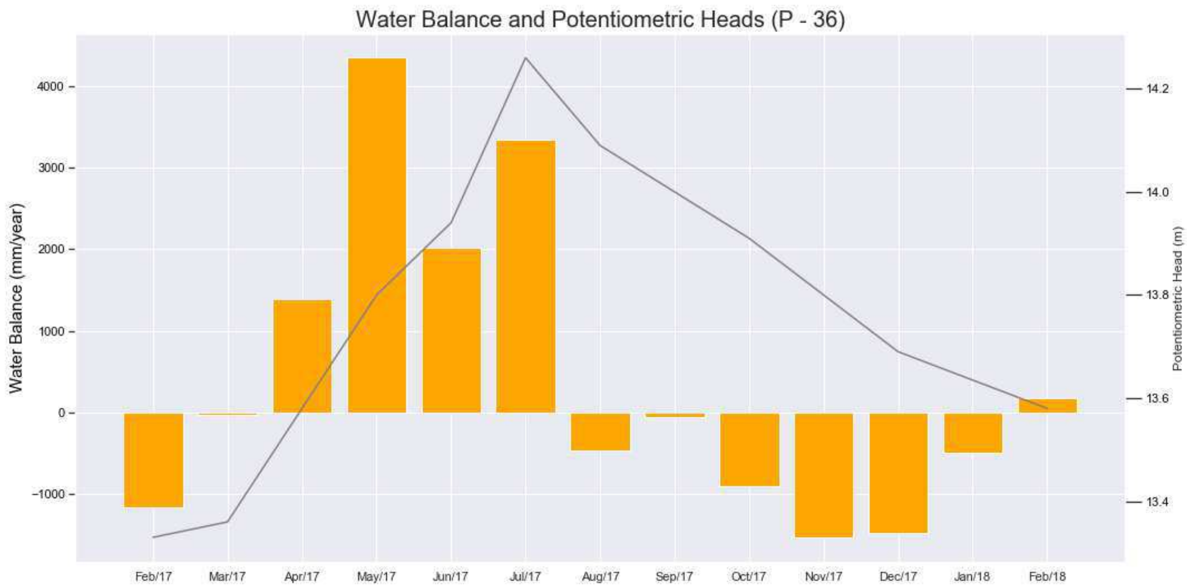
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13.4
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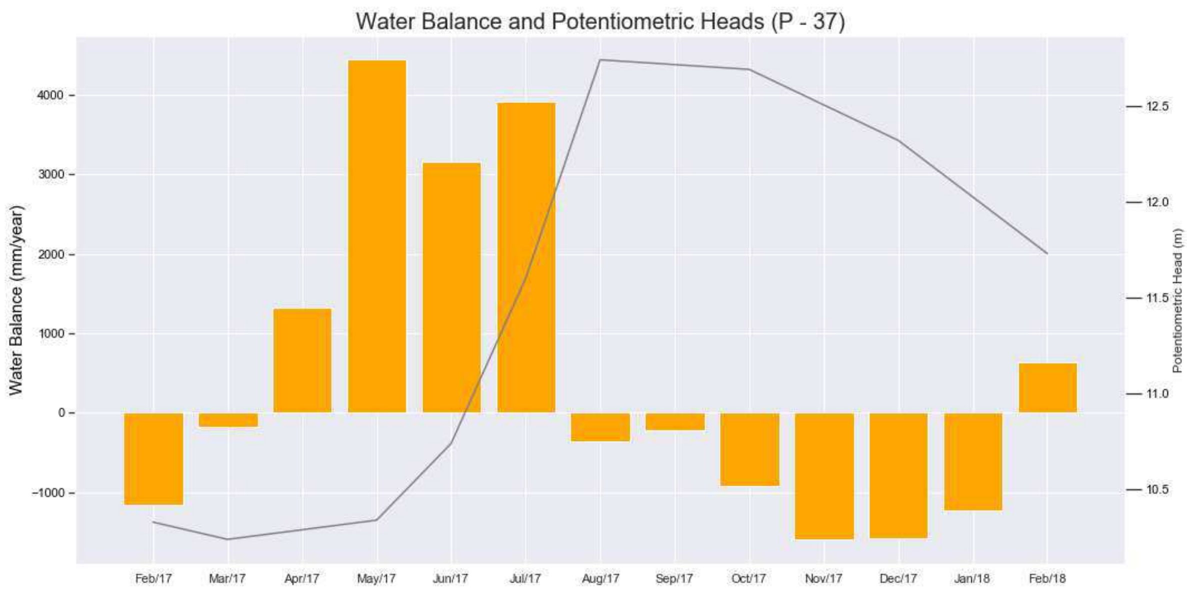
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2000
1000
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10.5
Feb/17
Mar/17
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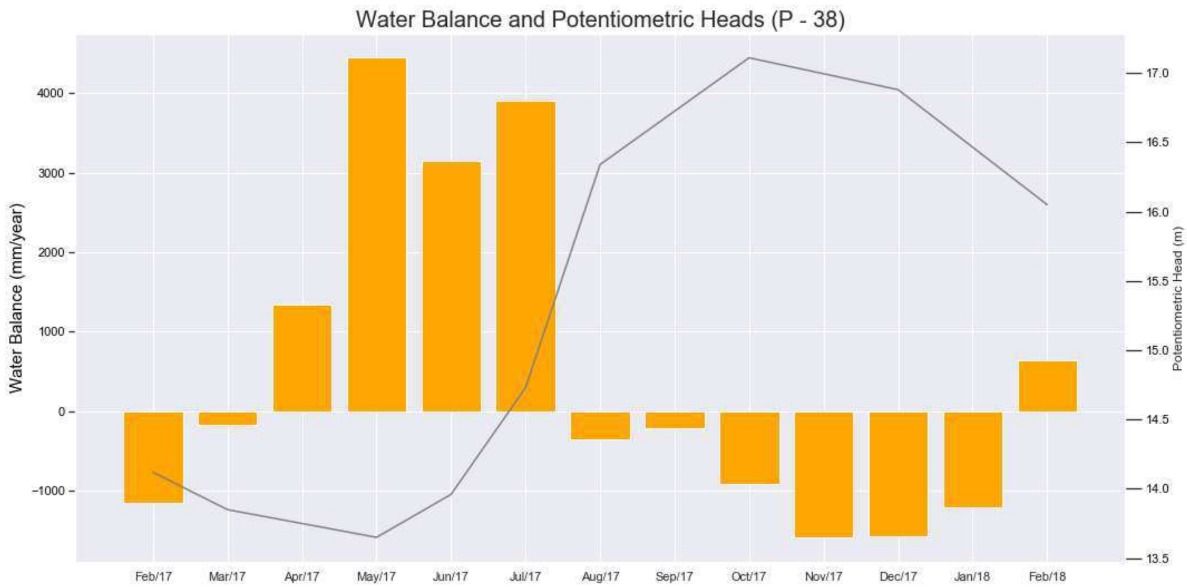
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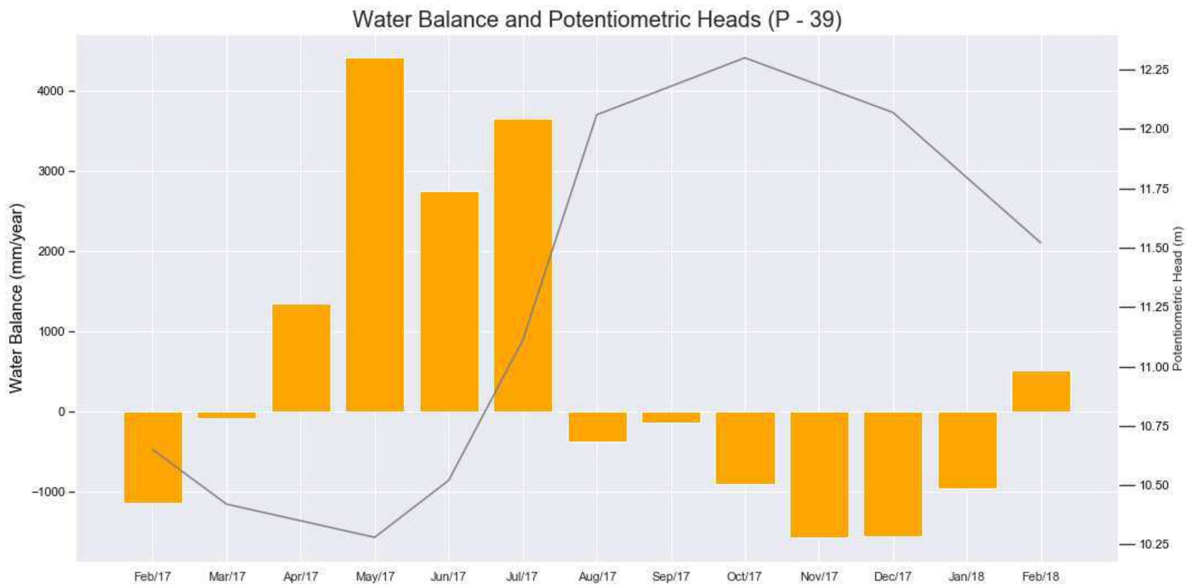
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Potentiometric Head (m)
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# Appendix C

## Gestão de Água Subterrânea através do Planejamento em Escala de Paisagem: incorporação das mudanças de uso do solo e provimento de informações[2]

Armando César Rodrigues Braga

Palavras-chave: Dinâmica especial, dinâmica temporal, recarga de água subterrânea, unidades de gestão

### 1. Introdução

As águas subterrâneas possuem um papel fundamental na manutenção dos ecossistemas e são a principal fonte de abastecimento para dois bilhões de pessoas. A crescente demanda por este recurso levou a uma depleção dos níveis freáticos ao redor do mundo. Ademais, a gestão de água subterrâneas é um trabalho desafiador. A maioria das abordagens tradicionais têm falhado na promoção de balanço sustentável a longo prazo. Desta forma, abordagens integradas que incluem os aspectos sociais, econômicos, culturais e a as configurações institucionais de cada sistema têm sido defendidas; além disso, fora da hidrogeologia, a análise das informações para gestão de água subterrânea é um desafio.

Água subterrânea e uso do solo estão intrinsicamente conectados. O uso do solo influencia a recarga e a demanda de água no aquífero, e mudanças no sistema aquífero podem impactar o uso do solo. Adicionalmente, mudanças no uso do solo atuam como um espelho do desenvolvimento socioeconômico e das mudanças ecossistêmicas. Desta forma, a gestão de águas subterrâneas precisar estar ligada ao uso do solo. O Planejamento em Escala de Paisagem (PEP) é uma abordagem para planejamento do uso do solo, definido como um arcabouço integrativo baseado em evidências e centrado nas paisagens. O PEP abrange três dimensões: a dimensão espacial, centrada no reconhecimento de unidades de paisagens distintas, a dimensão temporal, que inclui os usos passados, atuais e futuros das paisagens, e a dimensão de modificação, que envolve as alterações antropogênicas que afetaram e irão afetar a paisagens e seus recursos ao longo das dimensões espacial e temporal.

Esta abordagem oferece a oportunidade para integração entre setores de planejamento e gestão. Parte disso reside na adoção de unidades integrativas definidas como unidades de

[2] Resumo expandido da tese de doutorado intitulada “Groundwater Management through Landscape Scale Planning: incorporation of land-use changes and provision of information”, apresentada ao Programa de Pós-Graduação em Recursos Naturais da Universidade Federal de Campina Grande, em 13 de maio de 2020.

paisagem. Estas unidades são a base para coleta, análise e interpretação dos dados. A hipótese desta tese é que o arcabouço do Planejamento em Escala de Paisagem *(Landscape Scale Planning)* pode apoiar a gestão de águas subterrâneas pela incorporação das mudanças de uso do solo enquanto baseia a provisão de informações.

O objetivo desta tese é analisar a incorporação das mudanças de uso do solo na gestão das águas subterrâneas pela aplicação do arcabouço do Planejamento em Escala de Paisagem em uma paisagem cultural na costa do Nordeste do Brasil. Para este fim, i) foi investigado como o Planejamento em Escala de Paisagem e suas as conexões com as mudanças do uso do solo tem o potencial para fornecer informações para a gestão de águas subterrâneas; ii) foram sugeridas diretrizes para a aplicação do Planejamento em Escala de Paisagem como base para a gestão de água subterrâneas; iii) foi analisado em qual extensão as unidades de paisagem podem ser integradas para fornecer uma análise integrada da paisagem usando modelos numéricos de água subterrânea; iv) e analisado em qual extensão as unidades de paisagem podem ser aplicadas para a inclusão da dinâmica temporal na análise da gestão das águas subterrâneas.

## 2. Gestão das águas subterrâneas em áreas costeiras através do Planejamento em Escala de Paisagem

A análise das conexões entre a abordagem do Planejamento em Escala de Paisagem e suas conexões com as mudanças do uso do solo visando o potencial para fornecer informações para a gestão de águas subterrâneas foi realizada usando uma revisão sistemática de literatura. Este método já foi adotado em diversas pesquisas na área de gestão ambiental.

Como resultado, foram obtidos 28 artigos que trabalhavam com pelo menos uma das três dimensões do PEP e com gestão das águas subterrâneas. A partir da análise destes documentos foi encontrada uma conexão intrínseca do Planejamento em Escala de Paisagem com a gestão de águas subterrâneas. Esta conexão foi confirmada pela forma como cada dimensão do PEP se relaciona com as características do sistema de água subterrânea, incluindo as diferentes distribuições espaciais e temporais da recarga, assim como a forte influência que as modificações do uso do solo causam no aquífero. Além disso, pode ser identificado uma lacuna na inclusão das dinâmicas das mudanças de uso do solo para informar as atuais abordagens para gestão de água subterrânea.

Perspectivas para o melhoramento da gestão também puderam ser identificadas. O enquadramento da análise da gestão das águas subterrâneas no arcabouço do PEP pode levar a resultados mais integrativos. Assim, diretrizes para a gestão de água subterrânea baseadas nesta conexão foram sugeridas.

Na dimensão espacial, a adoção das unidades de paisagens pode informar melhor o processo de gestão, visto que tais unidades seriam determinadas a partir de aspectos biofísicos e socioeconômicos. Desta forma, as unidades de paisagem não precisariam obedecer aos limites político-administrativos ou coincidir com as delimitações das bacias hidrográficas.

Na dimensão temporal, a gestão deve se basear na melhor avaliação dos dados passados e atuais, sendo esses hidrogeológicos e socioeconômicos, para desenvolver possíveis narrativas tendo em vista diferentes objetivos de gestão. Além disso, é necessário analisar como a informação deve ser provida para a gestão: através de modelos transitórios ou por uma série de modelos permanentes.

Na dimensão de modificação, tanto as mudanças de uso do solo como as pressões antrópicas devem ser incluídas no processo de gestão. As forças motrizes que levaram o

Figure 11.1 - Selected Study Area for the Thesis

sistema ao estado atual, assim como as mudanças futuras ao longo do horizonte de planejamento devem ser incluídas. Um processo participativo envolvendo os usuários e atores pode ser aplicado para o desenvolvimento de perspectivas futuras que embasem a análise de cenários de forma a incorporar as pressões sociais sobre a paisagem e os sistemas de águas subterrâneas.

### 3. As águas subterrâneas em uma paisagem cultural: o caso João Pessoa

A paisagem cultural de João Pessoa representa um complexo sistema de águas subterrâneas em uma região tropical costeira que exibe diversas pressões antropogênicas, como crescimento econômico e populacional, aumento na demanda de água e rápidas mudanças de uso do solo. Esta situação não é exclusiva dessa área e representa uma configuração típica encontrada em outros locais ao redor do mundo.

A partir de uma análise do arcabouço legal relativo ao uso e cobertura do solo foi possível destacar possíveis caminhos para conectar o planejamento do uso do solo com a gestão das águas subterrâneas, baseados na legislação em vigor. Foram identificados certos instrumentos que podem ser utilizados de forma a facilitar a gestão das águas subterrâneas: áreas verdes e espaços livres de uso público, áreas de preservação permanentes e áreas de proteção da paisagem natural podem ser usadas para trazer as condições do ciclo hidrológico para mais próximas às de condições pristinas, enquanto a infraestrutura urbana básica pode ser utilizada como um sistema de recarga gerenciada de aquíferos não-estrutural.

Em relação ao arcabouço legal para a gestão das águas subterrâneas, é notado que há vários aspectos que não estão incluídos ou não são devidamente abordados pela legislação atual. Por exemplo, a legislação do estado da Paraíba que rege a concessão das outorgas possui critérios insuficientes para sua determinação. Como resultado, indicações de uma sobre-explotação tem sido detectada ao longo dos anos. Por outro lado, alternativas também foram apresentadas. O Projeto ASUB propôs diferentes níveis de gerenciamento. Estes níveis incluem um nível global (bacia hidrográfica), um nível regional (zonas de gerenciamento) e um nível local (poço). Além disso, também foram propostos critérios para outorga das águas subterrâneas baseados nos níveis propostos. Uma avaliação de parte desses critérios foi conduzida com relação a sua sustentabilidade considerando custos de implantação, conflitos de segunda ordem e impactos ambientais. Critérios em nível global apresentaram maior custo e potencial para conflitos de segunda ordem, os critérios de nível regional e local possuíram satisfatória adequabilidade aos aspectos de sustentabilidade.

## 4. Modelagem conceitual e numérica do sistema de água subterrânea

Os dados existentes sobre a paisagem cultural foram usados para a determinação de um modelo conceitual e sua posterior implementação no software computacional de modelagem numérica (aqui usado o FEFLOW). Através deste processo, foi possível caracterizar o comportamento geral do fluxo no sistema, incluindo as recargas e descargas.

O modelo conceitual para o sistema hidrogeológico analisado inclui as duas diferentes formações aquíferas, uma freática e outra confinada, sob duas diferentes bacias hidrográficas: a bacia do Rio Gramame e bacia do Rio Paraíba. A bacia do Rio Paraíba é compreendida parcialmente na área de estudo, sendo o Rio Paraíba utilizado como limite para a delimitação da área estudada (Figura 1). Além disso, nesta bacia também há a presença do reservatório Marés. Na bacia do Rio Gramame há três rios principais, o Gramame, Mamuaba e o Mumbaba. Nesta bacia há também um sistema de reservatórios conectados (o açude Gramame-Mamuaba), que abastece a cidade de João Pessoa. Assim, é possível observar que as condições de movimento das águas subterrâneas foram alteradas de suas condições pristinas.

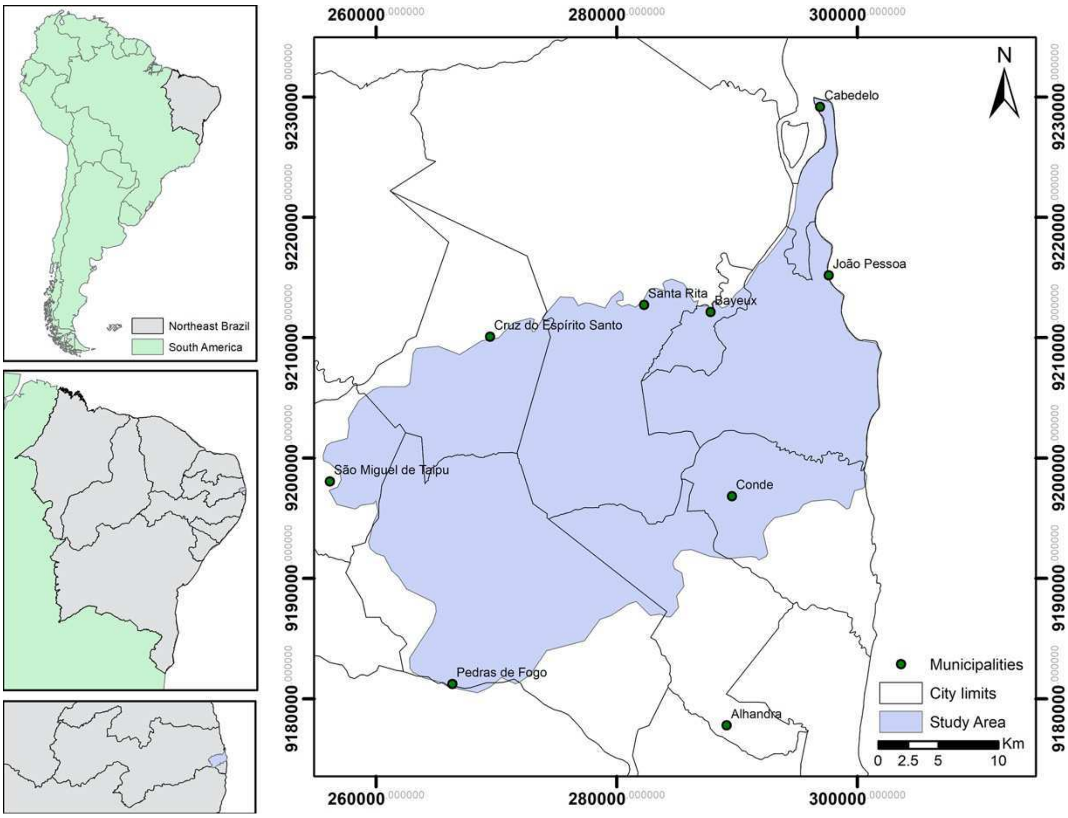


Figura 1 – Área de estudo selecionada

O sistema apresenta três diferentes formações hidrogeológicas. A superfície topográfica é considerada a camada superior do sistema. A formação hidrogeológica superficial é a camada Barreiras. De acordo com os dados obtidos, a formação Barreiras têm espessura média de 28 m. Os perfis hidrogeológicos indicam que a espessura varia entre 3 m e 174 m; este amplo intervalo de variação ocorre devido à formação ter um aumento gradual na sua espessura no sentido Oeste-Leste em direção ao mar. Este aumento ocorre devido ao processo geológico de sedimentação do aquífero. A segunda formação hidrogeológica é a formação Gramame. Essa formação não é capaz de armazenar ou transmitir o fluxo de água, agindo como uma camada confinante. Os perfis e as seções geológicas indicam que na área de estudo esta formação tem espessura média de 45 m e existe apenas a leste da falha geológica. A última formação é a formação Beberibe. Essa formação tem uma espessura média de 260 m na área de estudo. Na região confinada, a profundidade total da formação pode chegar a mais de 400 m (abaixo do nível do mar) próximo a costa; próximo à falha geológica (que define o começo do sistema confinado) a profundidade total fica em torno de 200 m. Assim, o sistema hidrogeológico pode ser dividido em dois subsistemas: um subsistema freático e um subsistema confinado.

O Rio Paraíba delimita a fronteira norte do sistema de água subterrânea. Por estar situado sobre uma formação cristalina, o sistema não apresenta conexão subterrânea entre a formação hidrogeológica Barreiras e a continuidade do sistema aquífero Paraíba – Pernambuco. Desta forma, o Rio Paraíba atua como a condição de contorno de carga constante, visto que o nível de água do rio pode ser aproximado como o nível de água do aquífero. O mar delimita a fronteira leste do sistema. A fronteira sul é delimitada pela bacia do Rio Gramame. A delimitação oeste do sistema é definida pelo limite topográfico das bacias do Rio Gramame e Rio Paraíba. Entretanto, nesta fronteira, diversos afloramentos cristalinos definem o começo da bacia sedimentar Paraíba-Pernambuco na área de estudo; assim, neste local não há entrada ou descarga de fluxo no sistema de água subterrânea.

A entrada de fluxo no sistema ocorre principalmente devido à precipitação, mas também a partir dos rios e reservatórios. A precipitação é concentrada de março a agosto, quando acontece a maior parte das chuvas. Durante os meses de abril a julho ocorre a maior parte da recarga para o aquífero. Essa recarga difusa é distribuída no aquífero e altamente influenciada pelo uso do solo existente. No período seco há alguma infiltração a partir dos rios e reservatórios. Isto ocorre mais frequentemente a montante do açude Gramame-Mamuaba, onde as descargas do açude contribuem para a perenização do rio. Entretanto, as recargas por essas outras fontes ocorrem em menor proporção do que a recarga da chuva.

As descargas dos sistemas de água subterrânea consistem na vazão de base, descarga submarina, evapotranspiração e retiradas para uso. A vazão de base mantém os principais rios e reservatórios durante o período seco. As descargas para o mar ocorrem através das formações Barreiras e Beberibe. As descargas da formação Barreiras ocorrem na linha da costa, enquanto as descargas da formação Beberibe ocorrem a 50 km da costa, dentro do mar, onde a formação Gramame deixa de existir. Algumas regiões do sistema, como as próximas aos rios e reservatórios, possuem um alto nível freático, cuja situação provoca descarga pela evapotranspiração. Finalmente, há as retiradas para agricultura, indústria e abastecimento urbano que são uma das principais fontes de descarga; é fundamental ressaltar que há um considerável número de poços irregulares e clandestinos que influenciam esta situação. A Figura 2 mostra a delimitação da área com a topografia e localização dos poços. A Figura 3 mostra a representação esquemática do modelo conceitual descrito.

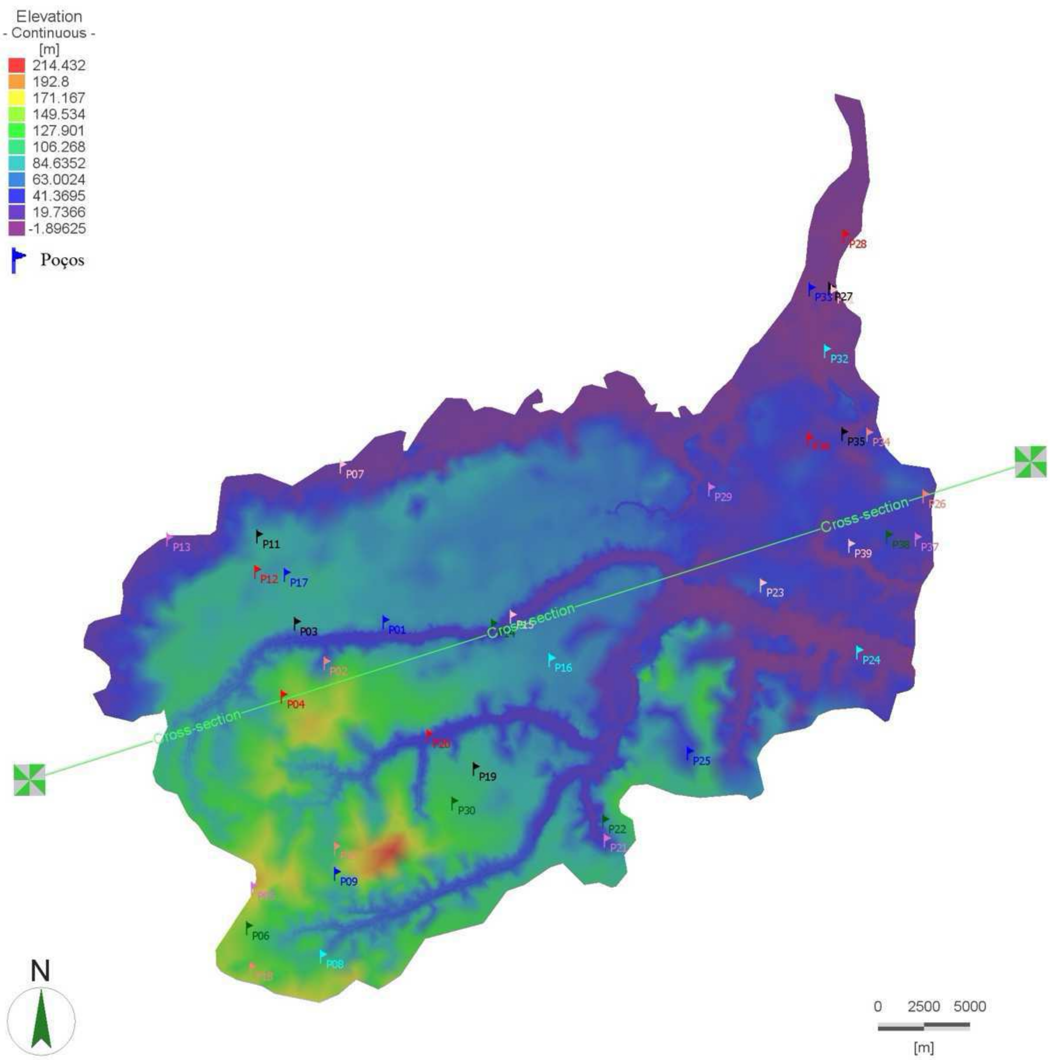


Figura 2 - Topografia da região e localização dos poços

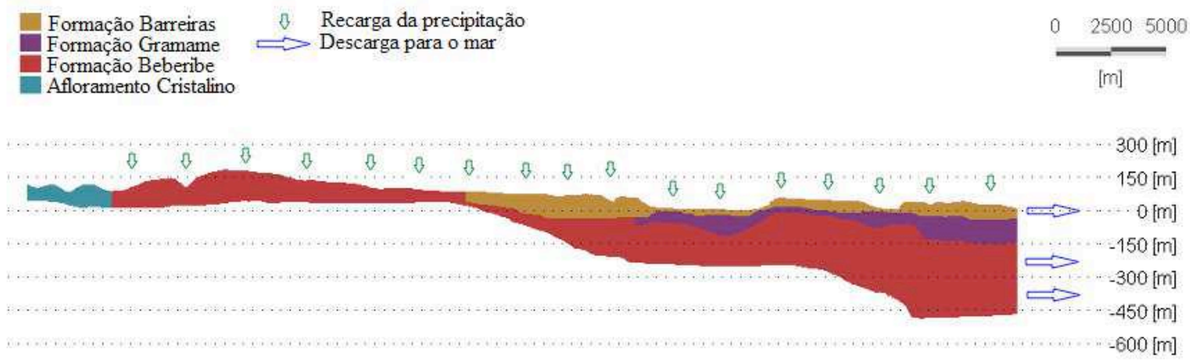


Figura 3 - Representação esquemática do modelo conceitual

O fluxo do sistema de água subterrânea ocorre em geral das regiões com maior altitude para as regiões com menor altitude, na direção do mar no sentido Oeste-Leste. Entretanto, baseado no fluxo subterrâneo, dois comportamentos específicos são identificados. Primeiro, entre os rios Gramame e Mamuaba, e os rios Mamuaba e Mumbaba, há divisores topográficos, de forma que uma considerável quantidade da recarga é transformada em vazão de base para estes rios. Segundo, a precipitação recarrega o aquífero freático e parte da quantidade recarregada é transmitida para o aquífero confinado para ser descarregada na fronteira Leste do sistema; entretanto, há uma considerável explotação próxima à área de recarga do aquífero confinado. Assim, a quantidade recarregada para o aquífero confinado pode ter sido reduzida. Além disso, esta explotação pode provocar mudanças na direção do fluxo no começo da região confinada.

O modelo numérico foi construído a partir do modelo conceitual usando o software FEFLOW. Foi efetuada a calibração para o regime permanente visando a obtenção dos parâmetros de condutividade hidráulica. A calibração foi conduzida para o mês de fevereiro de 2017. Para essa calibração foram utilizados dados de 39 poços localizados no aquífero freático, dos quais 11 foram usados como condição de contorno, e 13 poços localizados no aquífero confinados, dos quais quatro foram aplicados como condições de contorno. Devido ao elevado número de interferências nos níveis potenciométricos no aquífero confinado, resultante do bombeamento clandestino, a calibração foi conduzida em um regime de equilíbrio dinâmico. Assim, 19 poços com bombeamentos foram adicionados. Estes poços são explotados pela CAGEPA. Para o regime transitório foi efetuada a parametrização do modelo, visto que não havia dados suficientes para a calibração. Foi simulado o período durante fevereiro de 2017 a fevereiro de 2018. Os mesmos poços utilizados no regime permanente foram aplicados para parametrização no regime transitório. Além disso, a recarga foi estimada a partir do balanço hídrico na área estuada.

O modelo atingiu uma satisfatória correlação para o regime permanente. Apesar do sistema apresentar uma alta amplitude dos níveis potenciométricos, variando entre −3 m a 108 m para o aquífero freático, a maior parte das cargas foi bem representada. Uma pequena tendência de subestimativa das cargas foi notada para os valores potenciométricos acima de 80 m. Foi obtido um mapa com a zonas de condutividade hidráulica e os valores calibrados. Os valores de K variaram entre 1 m/d e 15,56 m/dia (Figura 4). As regiões próximas aos rios apresentaram maiores valores de condutividade hidráulica, o que é esperado, visto que nessas regiões há a presença de depósitos aluviais. No geral, os valores calibrados apresentaram boa concordância com as características geológicas existentes.

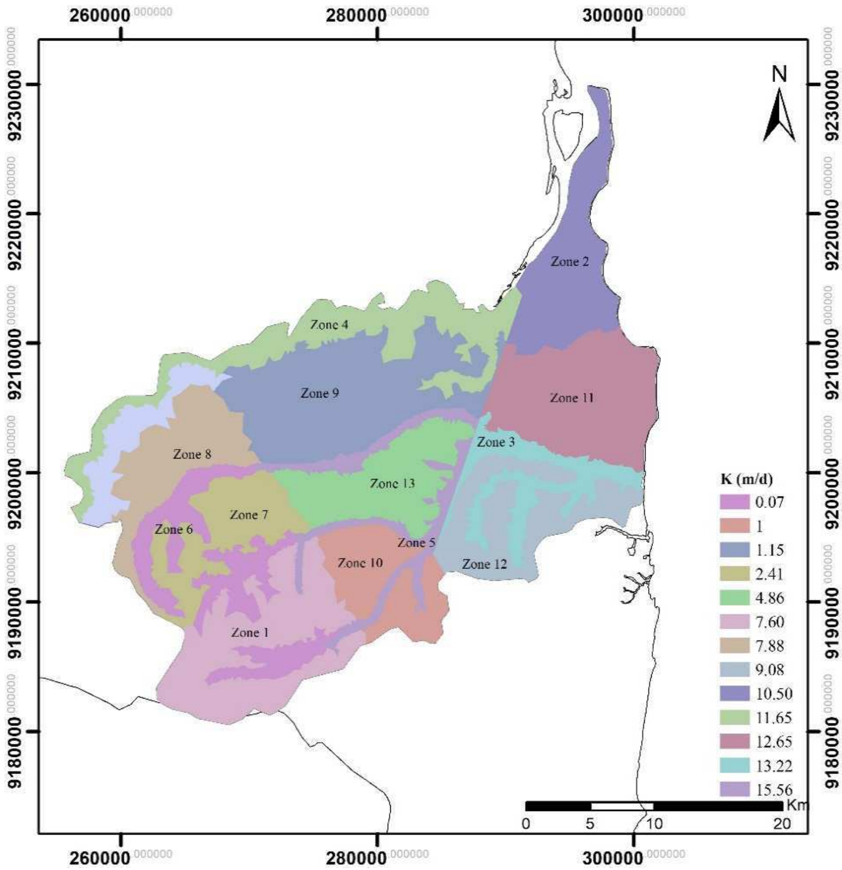


Figura 4 - Zonas de condutividade hidráulica e valores calibrados

Para o regime transitório foi estimada inicialmente a porosidade efetiva e, a partir desse valor, os parâmetros foram ajustados, de forma que a simulação representasse o comportamento do aquífero durante o período. O modelo conseguiu representar alguns comportamentos hidrogeológicos do sistema, como o aumento das cargas devido à recarga e a redução devida à evapotranspiração e à vazão de base. Uma parametrização razoável foi atingida levando em consideração os dados limitados. 23 poços foram usados para comparação entre as cargas observadas e simuladas. Em oito poços uma razoável representação do comportamento do sistema não foi atingida. Em três poços foi possível representar tanto os valores das cargas como o comportamento. Nos poços restantes só foi possível representar o comportamento das cargas, e os valores simulados apresentaram discrepâncias dos observados.

## 5. Aumento no conhecimento de um sistema de água subterrânea usando estimativas da condutividade hidráulica em uma área com escassez de dados

Um dos desafios para o processo de calibração foi a escassez de dados na área estudada. Dados são necessários para a construção do conhecimento sobre o aquífero, e a partir disso podem-se prover informações mais confiáveis para gestão de água subterrânea. Visando o aumento no conhecimento, foi realizada a estimativa da condutividade hidráulica a partir de dados de capacidade específica. Além disso, estes dados foram espacializados através de métodos geoestatísticos.

A abordagem metodológica deste trabalho seguiu os seguintes passos: inicialmente uma estimativa da relação empírica entre o índice de capacidade específica e condutividade hidráulica foi determinada através de uma regressão linear; o modelo de águas subterrâneas foi calibrado considerando dois diferentes limites para os parâmetros hidráulicos (um com dados da literatura, enquanto o outro com valores obtidos a partir das estimativas); uma análise de incerteza dos modelos calibrados foi aplicada usando o método de Monte Carlo; finalmente, a relação empírica foi espacializada aplicando o método geoestatístico da cokrigagem usando a condutividade hidráulica e o índice de capacidade específica como variáveis.

Esta estratégia de calibração foi aplicada para o subsistema aquífero confinado. Foram utilizados dados de 12 poços com medições em fevereiro de 2017. Dois diferentes conjuntos de dados de capacidade específica foram usados. O primeiro foi aplicado para determinar os valores da condutividade hidráulica na região, obtido de um banco de dados de estudos anteriores conduzidos na área e consiste em testes de bombeamento de curta duração (C1). O segundo conjunto de dados de capacidade específica foi utilizado para a estimativa da condutividade hidráulica baseando-se em relação empírica (C2). Este conjunto foi obtido nos bancos de dados da AESA e do SIAGAS.

Após a análise dos testes de bombeamento, 21 locais foram selecionados no aquífero confinado. Esses poços tiveram dados disponíveis para aplicação do método Cooper-Jacob para determinar a relação entre a condutividade hidráulica (K) e o índice de capacidade específica (SCi). Foi utilizada uma transformação logarítmica para os pares de dados de K – SC. Uma regressão linear foi utilizada para determinação da relação empírica entre K – SC. A relação linear log-K x log-SCi foi determinada com o coeficiente de correlação de 0,69, com intervalo de predição de 95% (Figura 5).

Após a determinação da relação, foi efetuada a calibração do modelo nas duas condições previamente explicadas. A primeira condição, usando dados da literatura como limites, apresentou o coeficiente de correlação de 0,91, a raiz quadrada do erro médio (RMSE) de 5,08, e erro médio absoluto (MAE) de 4,32. A calibração para a segunda condição apresentou melhores indicadores estatísticos: o coeficiente de correlação foi de 0,929, o RMSE de 4,615, e MAE de 3,86. Nas Figura 7 e Figura 6 estão apresentados os gráficos de dispersão e os mapas como as condutividades hidráulicas para as duas calibrações.

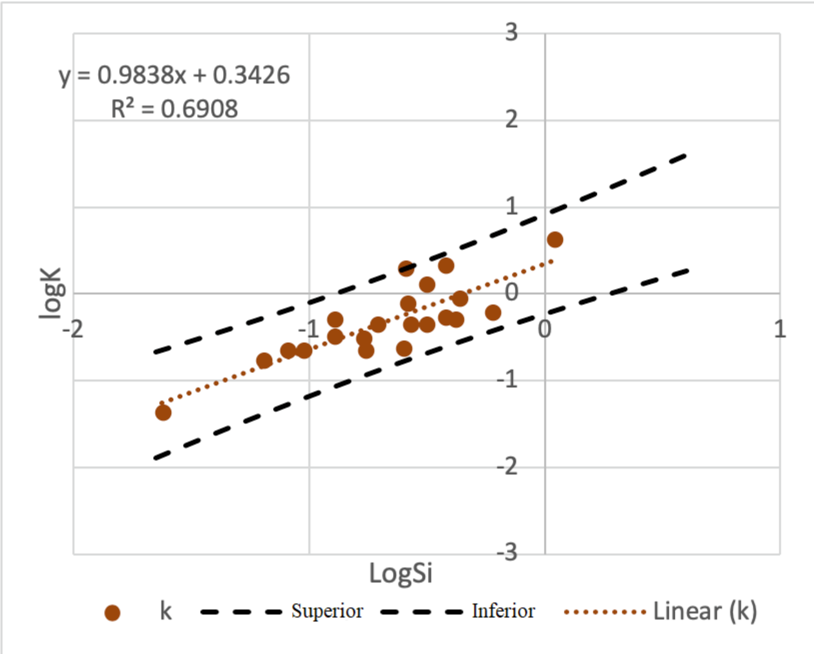


Figura 5 - Diagrama de dispersão da regressão linear

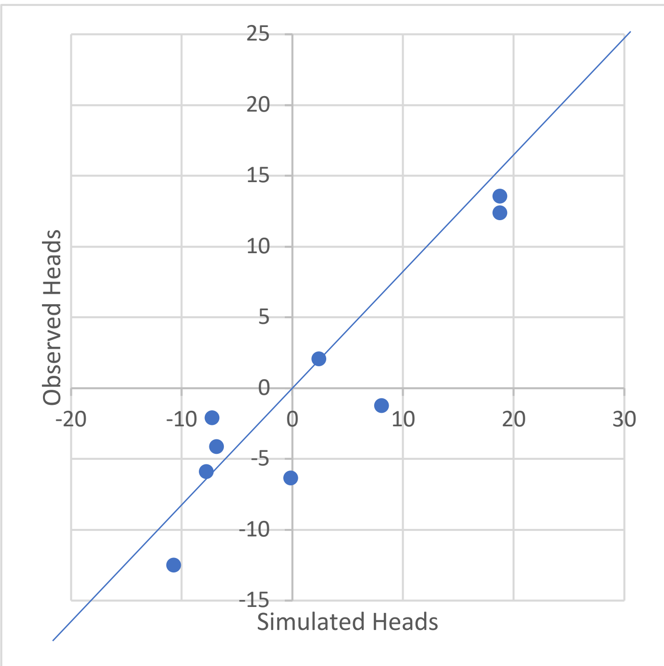


Figura 7 - Diagrama de dispersão - C1

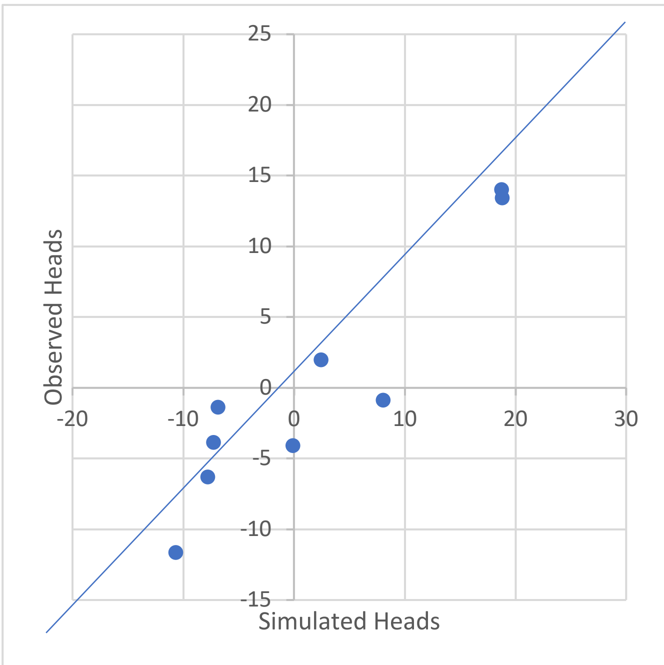


Figura 6 - Diagrama de dispersão - C2

A aplicação do método de Monte Carlo para a avaliação da incerteza preditiva indicou que o modelo calibrado usando as estimativas da condutividade hidráulica a partir do índice de capacidade específica apresentou menor nível de incerteza. Este nível de incerteza foi avaliado através do desvio padrão das cargas hidráulicas simuladas para os poços observados. Para o modelo calibrado C1, o desvio padrão variou entre 0,31 m e 2,81 m; para o modelo C2 o desvio padrão variou entre 0,07 m e 0,57 m.

Usando a regressão linear foi possível estimar dados de condutividade hidráulica para outros 121 locais a partir de dados pré-existentes do índice de capacidade específica. Em seguida, esses dados foram espacializados usando o método da cokrigagem. Os dados obtidos à parte deste método geoestatístico apresentaram um coeficiente de correlação de 0,89 com os dados estimados na regressão linear. Desta forma, o conhecimento sobre o sistema de água subterrâneo foi ampliado usando uma relação empírica entre a condutividade hidráulica e o índice de capacidade específica. Através dessa relação foi possível aumentar o número de informações sobre a condutividade hidráulica de 21 para 121. Essas informações podem ser usadas para calibração de modelos numéricos que apresentam uma menor incerteza e, por consequência, conseguem prover uma melhor informação para ser utilizada com um subsídio à gestão.

## 6. Gerenciando as águas subterrâneas na escala da paisagem: explorando as dinâmicas espaciais e temporais

A paisagem cultural adotada neste trabalho foi dividida em unidades de paisagem a partir dos conceitos da abordagem do planejamento em escala de paisagem (PEP). Essas unidades de paisagem foram delimitadas com o propósito de analisar as dinâmicas espaciais e temporais dos sistemas de água subterrânea, assim como a capacidade das unidades de paisagem de proverem informação como suporte à gestão.

As unidades de paisagens foram delimitadas usando os seguintes critérios: topográfico/hidrológico, hidrogeológico, demanda de água, uso do solo e sociais (contidas nos setores censitários). A integração das diversas camadas de informações em áreas homogêneas teve o objetivo de mostrar o potencial das unidades de paisagem como um local para análise, e sua viabilidade para aplicação como unidades de gestão. Aplicando este método, sete unidades de paisagem que incluíam características similares quanto aos aspectos ambientais culturais e socioeconômicos foram delimitadas. Assim, informações ambientais puderam ser analisadas em um mesmo espaço que as informações socioeconômicas (Figura 8).

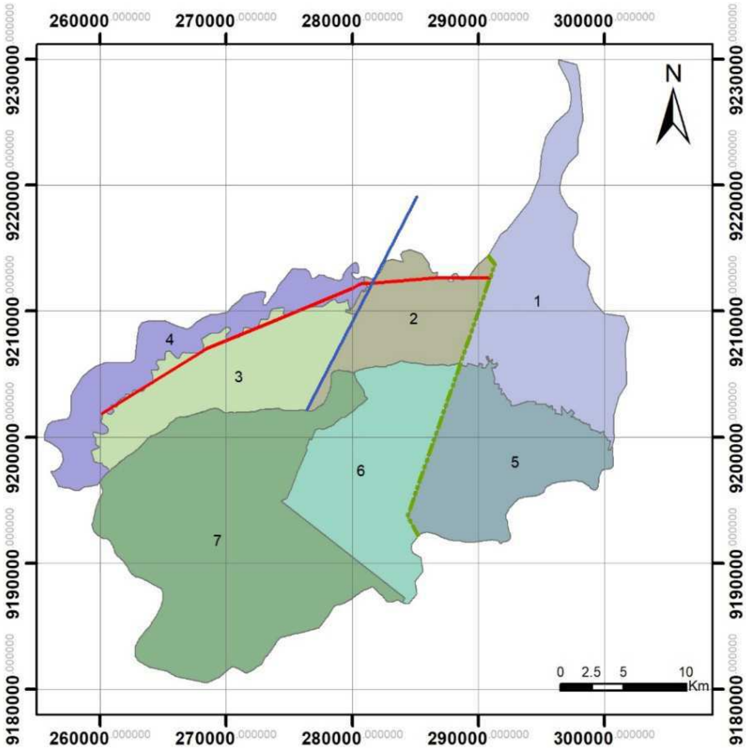


Figura 8 - Unidades de paisagem

As unidades de paisagem foram aplicadas para analisar o balanço hídrico subterrâneo e, como resultado, foi possível determinar a distribuição da recarga no sistema e em cada unidade. Neste processo foi utilizada a ferramenta de modelagem computacional numérica previamente calibrada e parametrizada. Com isso, foram identificadas algumas das dinâmicas espaciais do sistema de água subterrânea.

A recarga líquida estimada foi de 274 $hm^3$, correspondendo a 265 mm. A descarga para o mar foi calculada em 24 $hm^3$ do subsistema freático, e de 10 $hm^3$ do subsistema confinado. Além disso, foi estimada uma recarga de 32,1 $hm^3$ do subsistema freático para o subsistema confinado. O sistema aquífero foi dividido em sete unidades de paisagens. Como resultado da divisão das unidades de paisagem (LU), a recarga líquida pode ser estimada em cada LU (Figura 10), através da qual foi explicitada a diferença causada pelos diferentes padrões de uso do solo, como uma maior recarga em usos naturais e agrícolas do que nos usos urbanos. O fluxo subterrâneo entre as LUs também foi calculado a partir do modelo numérico (Figura 9). Foi possível determinar a descarga de cada LU externamente ao sistema, assim como a recarga e descarga entre os subsistemas.

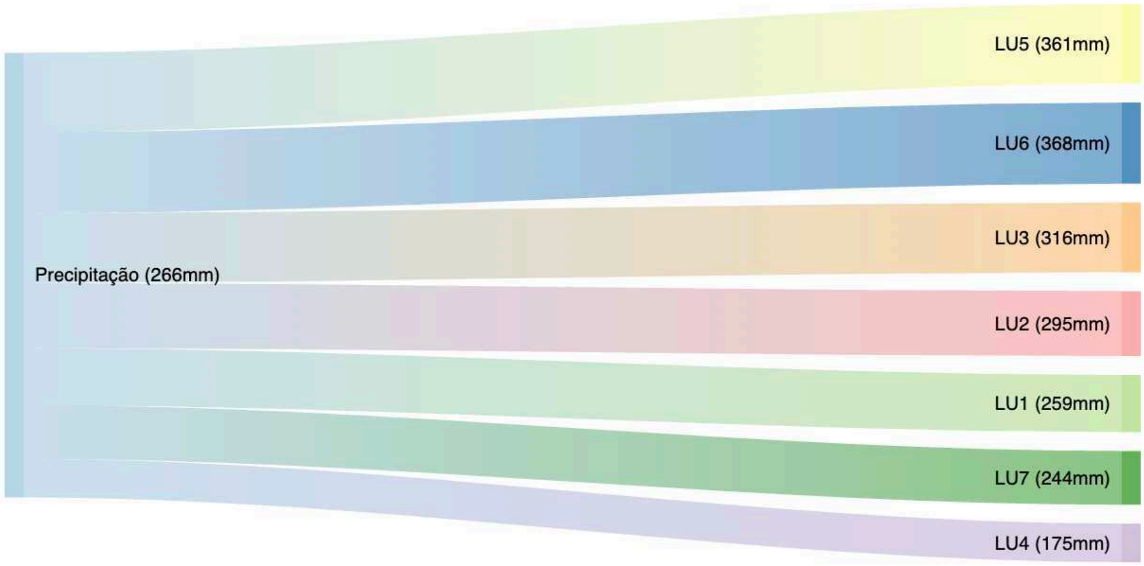


Figura 10 - Diagrama da recarga em cada unidade de paisagem

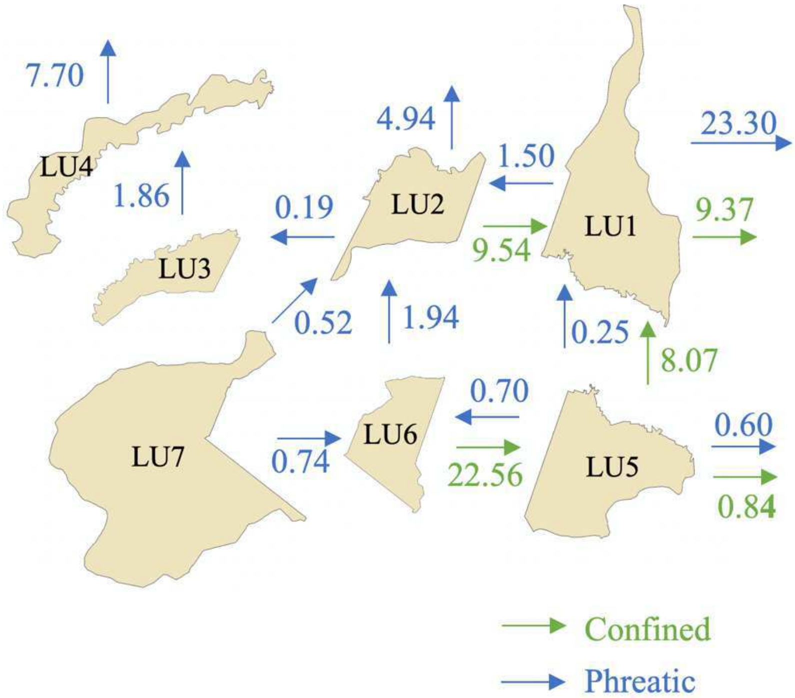


Figura 9 - Fluxo subterrâneo entre unidades de paisagem (valores em $hm^3$/ano)

A aplicação das unidades de paisagem também forneceu uma análise integrada da dinâmica espacial do sistema, levando à identificação de um fluxo subterrâneo (IGF) entre duas bacias na área de estudo. Uma possível razão para este IGF são os diferentes usos do solo e consequentes características da demanda hídrica. No subsistema freático, grande parte desse IGF nesse subsistema (70%) ocorre da LU6 para LU2. Na porção confinada do sistema aquífero, também foi identificado um IGF partindo da LU5 para a LU1.

Finalmente, a importância da dinâmica temporal para a gestão de águas subterrâneas foi avaliada baseando-se na constante temporal da bacia – definida como o tempo necessário para uma carga ou fluxo hidráulico em um sistema de água subterrânea se reajustar após uma pressão – em conjunto com a aplicação das unidades de paisagens delimitadas. Os resultados mostraram uma grande variação no cálculo da constante temporal da bacia, significando que a consequência dos distúrbios em algumas unidades de paisagem pode levar até 500 anos, enquanto para outras isto pode levar apenas 10 anos. Pressões ocasionadas por mudanças monotônicas rápidas, como alterações no uso do solo, podem afetar a descarga no aquífero no horizonte de planejamento usual. Três unidades de paisagem apresentaram constante temporal menor que 30 anos, indicando que mudanças no uso do solo podem ter 60% dos seus impactos sentidos dentro do horizonte de planejamento. Uma possível consequência disto seria a alteração da disponibilidade de água subterrânea para alocação.

## 7. Conclusões

A gestão de águas subterrâneas tem se provado uma tarefa difícil e complexa. Abordagens tradicionais baseadas apenas em aspectos técnicos têm sido incapazes de tratar dos diversos problemas existentes na gestão. Como consequência, uma abordagem integrada de gestão que inclua outros aspectos como os socioeconômicos e culturais tem sido defendida. Uma abordagem integrativa traz também uma grande gama de dados e informações que precisam ser organizados e incorporados de forma compatível para subsidiar o processo de gestão de águas subterrâneas. Apesar de tais abordagens já terem sido estudadas, um fator essencial ainda não foi adequadamente incluído: as mudanças de uso do solo. Enquanto as mudanças de uso do solo têm uma forte relação biofísica com os sistemas de água subterrânea, elas são espelhos dos desenvolvimentos socioeconômicos e culturais. Desta forma, a integração das mudanças de uso do solo com a gestão de águas subterrâneas tem importância fundamental.

Esta pesquisa levantou a hipótese que o Planejamento em Escala de Paisagem – uma abordagem para consideração do uso do solo no processo de planejamento – é capaz de subsidiar o processo de integração das mudanças de uso do solo com a gestão de águas subterrâneas, enquanto ajuda a provisão de informação para a gestão. Esta hipótese pode ser confirmada devido aos resultados a seguir: i) o Planejamento em Escala de Paisagem é intrinsecamente conectado com as mudanças de uso do solo e a gestão de água subterrânea; ii) o Planejamento em Escala de Paisagem pode guiar abordagens para a gestão de água subterrânea; iii) a análise através de unidades de paisagem pode integrar aspectos das mudanças de uso do solo para a provisão de informação como suporte à gestão de água subterrânea levando em consideração as dinâmicas espaciais e temporais.