Using remote sensing data to assess the impact of wildfires in groundwater recharge in the Vieira de Leiria-Marinha Grande aquifer – Portugal

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Thesis to obtain the Master of Science Degree in

Environmental Engineering

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“All the progress takes place
outside the comfort zone.”

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Resumo

Os incêndios florestais afetam a vida de milhares de pessoas no mundo todo, não apenas por seus impactos ambientais, mas também pelas perdas sociais, culturais e económicas inerentes a estes eventos. Embora as consequências dos incêndios florestais tenham sido documentadas por vários autores ao longo dos anos, os seus impactos nas águas subterrâneas têm sido pouco investigados e ainda são mal compreendidos. A região de Marinha Grande, Portugal é essencialmente abastecida por águas subterrâneas provenientes do Aquífero de Vieira de Leiria-Marinha Grande para consumo humano, irrigação e indústria. No entanto, muito pouco se sabe sobre os efeitos do grande incêndio ocorrido no Pinhal de Leiria em outubro de 2017 e responsável pela devastação de cerca de 86% da floresta. Compreender as consequências dos incêndios florestais na qualidade e quantidade das águas subterrâneas é fundamental para propor medidas eficazes de gestão e adaptação e garantir o abastecimento futuro.

No presente estudo, dados de NDVI recolhidos da base de dados do satélite MODIS e dados climáticos da base de dados E-OBS para o período de 2001-2020 foram usados para estimar a evapotranspiração potencial ajustada (PET_{CA}) na área de estudo. Dados climáticos e de propriedades do solo foram inseridos no software Easybal, desenvolvido pelo grupo de Hidrologia Subterrânea da Universidade Politécnica da Catalunya (Barcelona, Spain), para simular a recarga de água subterrânea nas áreas ardida e não ardida do Pinhal de Leiria. Os resultados mostram uma diminuição da PET_{CA} devido à remoção da vegetação pelo incêndio e um aumento da recarga da água subterrânea no aquífero de cerca de 15% no primeiro ano, 7% no segundo e 3% no terceiro ano quando comparados com os valores esperados. Este aumento provavelmente não está relacionado exclusivamente com a diminuição da evapotranspiração, mas também é condicionado pelas características geológicas e pedológicas da área, gradiente topográfico suave, escoamento superficial insignificante, altas taxas de infiltração, além de condições climáticas específicas.

No futuro, e com o objetivo de otimizar as estimativas de recarga subterrânea e a avaliação dos impactos dos incêndios florestais nesta ou em outras áreas de estudo recomenda-se: (1) a instalação de uma rede de monitorização do nível de água subterrânea do aquífero superficial; (2) monitorização de parametros meteorológicos para comparação, correção e validação dos dados de satélite; (3) recolha de amostras de água da chuva e água subterrânea para análise de cloretos; e, (4) devenvolvimento de investigações focadas nos fatores climatológicos e propriedades do solo que influenciam a recarga ao aquífero na região de Leiria.

Palavras Chave: recarga subterrânea; deteção remota; ETP; incêndios florestais

Abstract

Wildfires impact lives of many people worldwide not only for their environmental implications, but also the social, cultural, and economical losses inherent to these events. Although the consequences of wildfires have been documented by several authors over the years, the impacts in groundwater are still poorly understood.

The region of Marinha Grande - Portugal is essentially supplied by groundwater from the Vieira da Leiria-Marinha Grande Aquifer for domestic and industrial uses, however, very little is known about the hydrological consequences of the fire occurred in Leiria Pine Forest in October 2017, responsible for the devastation of about 86% of the forest. Understanding the impacts of wildfires in groundwater quality and quantity is imperative to propose effective management and adaptation measures and guarantee future supply.

In the present study, remote sensing NDVI data from MODIS satellite database and climate data from the E-OBS database for the period of 2001-2020 were used to estimate crop adjusted potential evapotranspiration (PET_{CA}) in the study area. Climate and Soil property data were inserted in the Easybal software, developed by the Hydrology Group of the Universitat Politècnica de Catalunya (UPC, Barcelona, Espanha), to simulate groundwater recharge in both burnt and unburnt areas of the Leiria Pine Forest. The results show a decrease in PET_{CA} due to the removal of the vegetation by the fire and an increase groundwater recharge in the aquifer of about 15% in the first year, 7% in the second and 3% in the third year when compared to the expected values. The increase is probably not exclusive related to the decrease in evapotranspiration, it is also conditioned by geological and pedological characteristics of the area, smooth topographic gradient, negligible runoff, high infiltration rates and specific climate conditions.

For the future and with the objective of optimizing recharge estimates and the assessment of the impacts of wildfires in this and other areas, there are some recommendations: (1) the installation of a monitoring network for the shallow aquifer; (2) monitoring meteorological parameters to compare, correct and validate satellite data; (3) collection of rainwater and groundwater samples for chloride analysis, and (4) development of studies focused on understanding the climatological factors and soil properties influencing groundwater recharge in the region.

Keywords: groundwater recharge; remote sensing; PET; wildfires

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1) Introduction

This chapter presents a brief introduction about the topic of the research and its background considering the factors, events and particular conditions that make the study relevant for a better management of the water resources in the study area, as well as a brief description of the objectives to be achieved and the methods applied.

1.1) Background

Wildfires impact lives of many people worldwide and cost billions of euros in direct and indirect damage, not only due to their environmental implications, but also the social, cultural, and economical losses they may cause (Turco *et al.,* 2019).

The effects of wildfires on the environment include removal of the soil-protection vegetation, ash deposition, changes in the physical properties of rocks and soil and, impacts in the water cycle leading to changes in water quantity and quality (Figure 1). These effects have been documented in several studies worldwide (Greenbaum *et al.,* 2021; Balocchi *et al.,* 2020; Loiselle *et al.,* 2020; Robinne *et al.,* 2020; Carvalho-Santos *et al.,* 2019; Hawtree *et al.,* 2015; Morán-Tejeda *et al.,* 2015; Smith *et al.,* 2011; Shakesby and Doerr, 2006, Neary *et al.,* 2005).

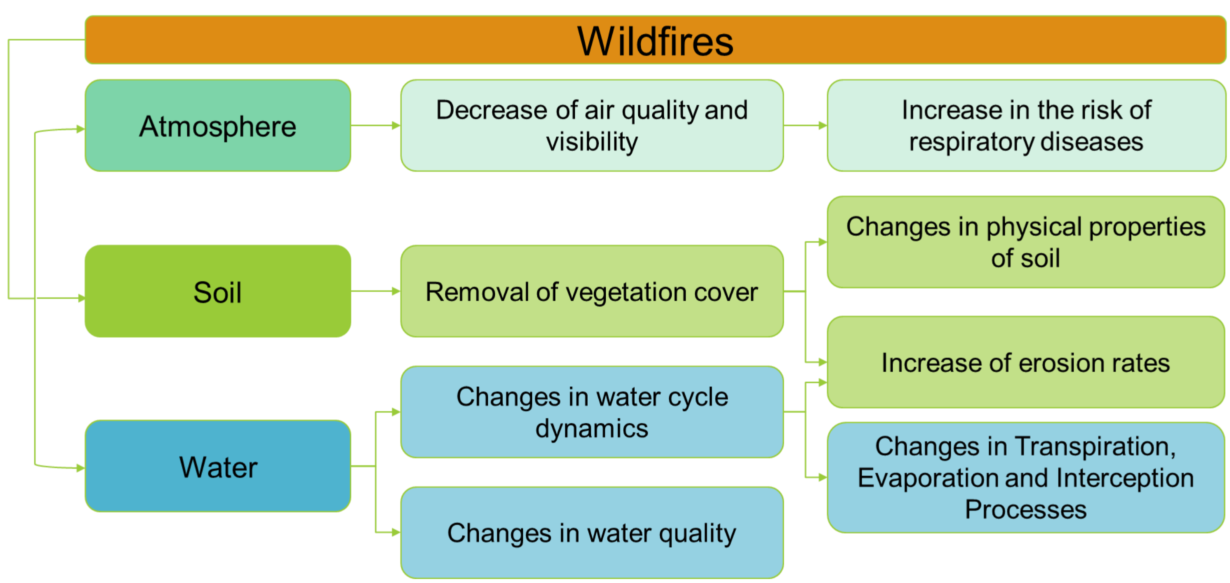


Figure 1 - Effects of wildfires in the environment (atmosphere, soil and water resources).

The removal of the vegetation cover affects not only the soil properties but also changes the rates of important hydrological processes such as evaporation, transpiration, and interception (Poon and Kinoshita, 2018; Nolan *et al.,* 2015). The unprotected soil is more susceptible to erosion and the changes in infiltration rates might pose a threat to the water supply in groundwater dependent regions affected by wildfires (Balocchi *et al*., 2020; Kinoshita and Hogue, 2011).

According to the US National Interagency Fire Center (NIFC, 2021) the number of wildfires worldwide was 58,950 in 2020 compared with 50,477 in 2019, representing an increase of about 15%. Besides the increase in the number of events, the extensive biodiversity loss and property destruction raised the discussion and promoted awareness about the worrying consequences of these events (Rhoades *et al.,* 2019; Jergler, 2019).

Several studies report that climate and land-use changes are the main drivers for the increasing number of wildfires, as well as their intensity and extent, making them reach areas nowadays where they rarely occurred in the past (Feyen *et al.,* 2020; EU, 2018; Santos *et al.,* 2019; Seidl *et al.,* 2017; Bladon *et al.,* 2014; Chaplot, 2007; Costa *et al.,* 2003). The Mediterranean region has always suffered from wildfires but according to Calheiros et al. (2020) the fire regime in Portugal has changed in the last 40 years.

The increase in the occurrence of climatic extreme events such as droughts and heatwaves can escalate the risk of these events, especially during summer (Turco *et al.,* 2019). For instance, in 2019, intense fires were reported in Australia, Canada, Siberia, the Mediterranean and the Amazon Forest (Brazil).

According to data from the Portuguese Institute of Conservation of Nature and Forests (ICNF), the probability risk of occurrence of a fire with a burned area higher that 1000 km^2 in Portugal increased from 30 to 61% between 2000 and 2017, respectively. Although the Mediterranean region is known by its high susceptibility to fires and Portugal is one of the countries most affected by fires in Europe (Beighley & Hyde, 2018), these events as well as their effects in the atmosphere, soil and water resources are still poorly evaluated, especially when it comes to understanding the qualitative and quantitative impacts on groundwater resources.

Understanding its causes and assessing impacts is the best way to learn how to better adapt to wildfires, preventing not only economic, social and environmental damages, but also ensuring a better management of resources under stressful climate conditions (Rhoades *et al.,* 2019; Hallema *et al.,* 2016).

1.2) Problem statement

The boost in the number of wildfires in Portugal by the increase in extreme climate events may enhance even more the unbalance in the hydrological cycle and affect the security of water supply for local people in some areas of the country. After a prolonged dry summer with extreme temperatures, a fire occurred in October 2017 in the Marinha Grande region, Portugal burning about 86% (11 hectares) of the Leiria Pine Forest ('Pinhal de Leiria') also called the King's Pine Forest (Pinhal do Rei) (Figure 2).

Figure 2 - Images from the fire occurred in the Leiria Pine Forest on October 17th, 2017.

The Leiria Pine Forest is a unique woodland area that started to be planted more than 700 years ago, in the 13th century, during the reign of King Afonso III and later by his son, King D. Dinis to break the development and restraint the degradation of the dune system close to the city of Leiria. The forest was very important to the development of the region, and besides its environmental value, it has cultural and historical meaning not only to the region, but to the country.

In 2020, after 3 years from the fire, the impacts of the fire in the soil and the hydrological cycle are still poorly understood and, although 11% of the area was reforested, the inexistence of a reforestation plan with defined goals combined with demographic, political and climatic factors increase the risk for water supply in the region (Figure 3).

Figure 3 - Images from before and after the fire in the Leiria Pine Forest (Source: https://muitobom.com/o-pinhal-de-leiria-era-assim-mas-ve-como-esta-agora-e-desolador)

The region of Marinha Grande is essentially supplied by groundwater for drinking purposes, food preparation and irrigation. The occurrence of wildfires constitutes a great environmental concern and its impacts in groundwater are still poorly understood.

According to Smith et al. (2016) about 50% of all drinking water worldwide comes from groundwater and although there are several studies reporting the impacts of wildfires in soil, surface water and in the water cycle in general, very little is known about the effects of these events specifically in groundwater (Nunes *et al.,* 2018). The lack of knowledge about these impacts increases uncertainty and vulnerability of the communities that completely rely on groundwater resources, threatening sustainability of water supply.

Wildfires may potentially contaminate irrigation water with sediment and ash, and increase fluxes of carbon, nutrients, and other water quality constituents into the water table (Smith *et al.,* 2011). Besides the water quality changes, these events have a huge potential of changing the processes in the hydrological cycle, altering water availability as well.

Following wildfire, removing vegetation leads to an increase in the temperature of the soil, which affects hydrological processes such as evaporation, transpiration and interception. The absence of protection vegetation often increases erosion rates, altering the balance between runoff generation and infiltration rates, and potentially changing groundwater recharge (Carvalho-Santos *et al.,* 2019; Poon & Kinoshita, 2018; Morán-Tejeda *et al.,* 2015; Nolan *et al.,* 2015).

Therefore, understanding and quantifying the impacts of these events can be decisive to guarantee the water availability by ensuring not only the best water management strategies, but also the future of the forests over the country, including the Leiria Pine Forest.

1.3) Objectives

Considering the increasing in the magnitude and distribution of wildfires worldwide, it is crucial to understand the hydrological effects of these events in order to develop better management and mitigation strategies.

The main objective of this study is to evaluate the effects of wildfires in groundwater recharge in the shallow portion of the Vieira de Leiria-Marinha Grande aquifer. To achieve it the following specific research objectives are defined:

1. Prepare a database containing the fountains, springs, wells and boreholes known in the area;

2. Identify recharge and discharge zones and elaborate a conceptual model of the region;

3. Use satellite data in order to get spatial distributed information on vegetation indices and climate variables;

4. Determine Crop Adjusted Evapotranspiration for the burnt and unburnt areas of the Leiria Pine Forest;

5. Estimate groundwater recharge for the area for the period of 2001-2020 using EasyBal, a programme designed to calculate water balances of water in soil (Serrano-Juan & Vazquez-Suñe, 2015).

1.4) Overview of the method

Aquifer and water points information were collected using data from SNIRH, the Portuguese National Information System of the Water Resources; and, organized together with remote sensing data in a database that was set up to manage all the available data. Field work was proceeded to get geophysical and borehole information on the groundwater level depth and help elaborating a conceptual model of the study area.

Climate variables and vegetation indices data from 2001-2020 were taken from the E-OBS platform and MODIS satellite, respectively. The use of remote sensing data was chosen in order to overcome the absence of spatial and temporal distributed monitoring data in the area. These data were processed using Python and R codes in order to get monthly values that could be applied in the recharge simulations. Limitations in this type of dataset were considered and discussed in the following chapters.

Recharge simulations were done using Easybal v10.9 (Serrano-Juan & Vazquez-Suñe, 2015) developed by the Hydrogeology Group of the Polytechnic University of Catalonia to provide a better understanding on how the changes over time my affect the shallow portion of the Vieira de Leiria-Marinha Grande aquifer. A schematic overview of the method is given in Figure 4.

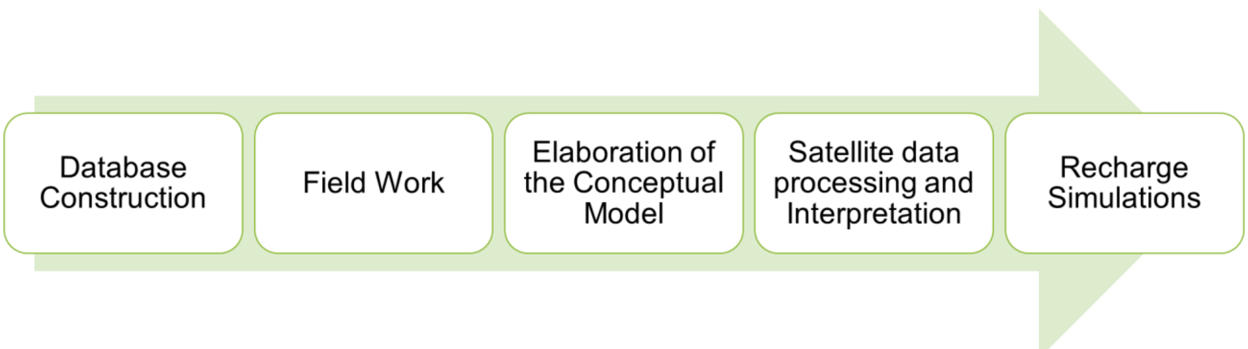


Figure 4 - Summary of the methodology followed in the present study.

2) Literature Review

Chapter 2 presents the literature review on the topic with the main advances and studies about the evolution of wildfires worldwide in the last decade, followed by the social, economic and environmental impacts of these events.

2.1) Wildfires worldwide

Wildfire has been a growing concern worldwide due to their huge social, cultural, economic, and environmental impacts in fire-prone forest areas. Yearly, large areas of forest land are burned, particularly in western North America, Mediterranean and in the south-eastern Australia (FAO, 2001).

For instance, in the summer of 2018, a series of wildfires were reported around the globe affecting Australia, Greece, Scandinavia, Indonesia, Artic Circle, Canada, United States, Brazil and United Kingdom (Turco et al. 2019). According to the Canadian Council of Forest Ministers (2019), in 2018 there were more than 7000 wildfires in Canada, corresponding to a burned area of 2.3 million hectares of forest.

In the United States, 58,083 wildfires were reported in 2018, totalizing an area of about 3.5 million hectares of burned forest, from which 800.000 hectares were in California (National Interagency Fire Center, 2020). The wildfires occurred in Australia (2019-2020) were one of the most severe from the last decades, compromising 10.7 million hectares (Boer *et al.,* 2020), destroying 3,000 houses and killing 33 people.

In 2019, South America also experience a series of wildfire events affecting Peru, Venezuela, Bolivia and Brazil. The Brazilian Amazon was the most affected area with 74,155 spots of fires and about 1 million hectares of burned forest from January to August. Considering all South America, the rate of increasing in the number of fire outbreaks in 2019 was 33% higher than the same period of 2018 (INPE, 2021).

In Siberia, the fire season of 2019-2020 reached the record amount of carbon dioxide released in the atmosphere. It happens because about half of the wildfires occurred on carbon-rich peatland, that can burn longer than forests and release much more carbon. According to an interview given by Mark Parrington, a senior scientist at the Copernicus Atmosphere Monitoring Service (CAMS) of the European Centre for Medium-Range Weather Forecasts to the Earth Observatory (NASA), the increasing temperatures make easier for the frozen peatlands to melt, making them highly flammable.

In Europe, the situation is also alarming. According to a report elaborated by a research group from the European Union (EU), in 2019 over 400,000 hectares of natural lands including protected areas were affected by wildfires inside the EU (San-Miguel-Ayanz *et al.,* 2020).

In 2018, more EU countries were hit by wildfires than ever, and in 2019, the number of fires recorded in the EU was three times the average over the past decade (Feyen *et al.,* 2020). However, Figure 5 shows a slight decrease in the burned areas for the Mediterranean Europe (EUMED5 – Southern France, Greece, Italy, Portugal and Spain), with exception of Portugal (Turco *et al.,* 2016).

Back in the day, fires were a concern only for EUMED5, but nowadays, they have spread across the whole EU territory. In Germany, the number of fires reached 1523 in 2019, with a burnt area of 2,711 ha which corresponds to an increase of 528% compared to the average from 2009-2018 (514 ha). Despite the case in Germany represents a tremendous increase compared to the previous decade, other European countries have much bigger areas compromised by wildfires in 2019 such as Italy (36,034 ha), Portugal (42,084) and Spain (83,963 ha).

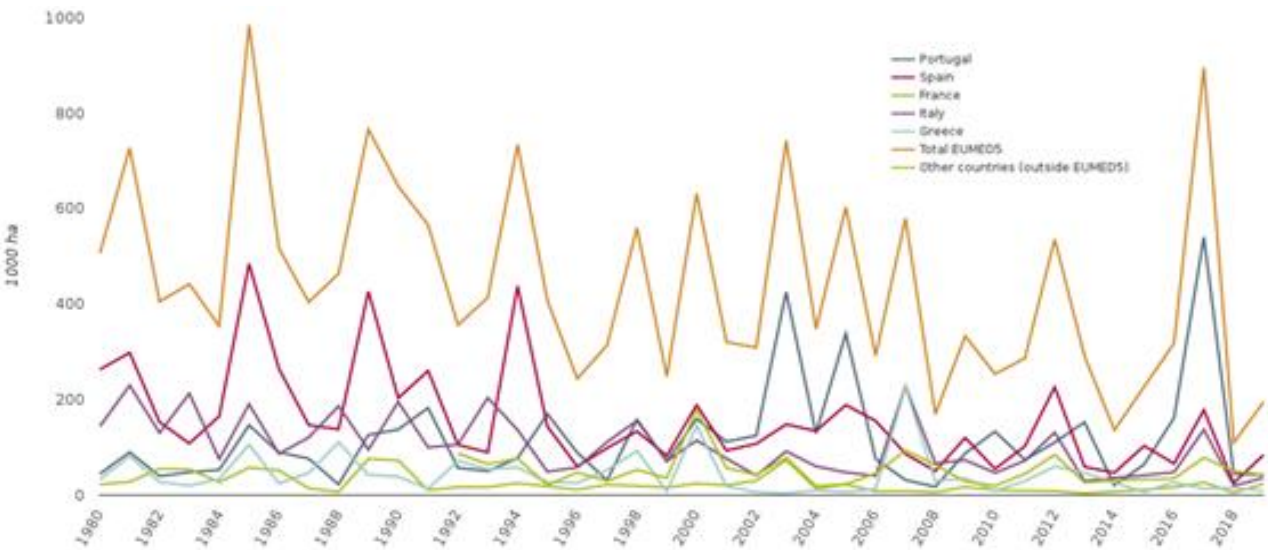


Figure 5 – Burned areas in European Countries (1980-2019). The data are supplied by the countries for the annual reports "Forest Fires in Europe, Middle East and North Africa 20nn" series (San-Miguel-Ayanz *et al.,* 2020).

In Portugal, 10,832 were registered in 2019 corresponding to a burnt area of 42,084 ha, about 49% occurred in the summer period (July-September), which consumed around 31,505 ha (75% of the total burnt area in the year) (San-Miguel-Ayanz *et al.,* 2020). From the total burnt area, about 50% (21,432 ha) corresponds to forests.

Data gathered by the European Union and presented by the European Commission in the JRC Technical Report: Forest Fires in Europe, Middle East and North Africa 2019 shows that despite there us a decrease in the number of fires in Portugal, we can point an increasing trend in the average fire size and in the burnt area since 2006.

In terms of total burnt area in the mainland, 2017 was the worst year since 1980, when the monitoring started in Portugal (Figure 5), with 539,921 hectares of burnt area. This area corresponds to an increase of 498% when compared to the average of the previous 10 years (90,269 ha).

2.2) Impacts of wildfires

Wildfires may have social, economic and environmental impacts. These events can disrupt essential services to the affected communities (transportation, communications, power, gas services, and even water supply), lead to loss material goods (property, resources and crops) as well as animal and people lives, and deterioration of the environment (air, soil and water quality) (Candela *et al.,* 2005, Inbar *et al.,* 1998).

There is a growing concern about the effects of the wildfires worldwide, especially the environmental risks associated to them and the persistent effects on stream flow volume, seasonal timing of flow, water quality, aquatic ecosystem health, and downstream community drinking water treatment (Robinne *et al.,* 2021; Flint *et al.,* 2019; Bart, 2016; Shakesby, 2011).

During and after a fire, the smoke can travel thousands of kilometers, affecting air quality, visibility and increasing the risk of respiratory diseases in the population. Besides, due to the removal of the vegetation and deposition of the ashes, alterations in the physical properties of rocks and soil, the enhancement, formation, or destruction of the water repellent layer, changes in sediment yield and soil moisture content as well as an increase in the potential for mass movement are commonly described in the literature (Greenbaum *et al.,* 2021; Horel *et al.,* 2015; Inbar *et al.,* 2014; Chen *et al.,* 2013; Shakesby and Doerr, 2006; DeBano, 2000)

Wildfires are agents of land cover change able to disturb the vegetation and destabilize the processes associated with water production (Rodrigues *et al.,* 2019; Costa *et al.,* 2003). These events alter the carbon cycle in watersheds and affect not only the response of important hydrological processes (Balfour *et al.,* 2014), but also water quality properties (Hohner *et al.,* 2019).

Among the impacts of fire in the water cycle reported by the scientific community we may find the disruption of important ecohydrological processes like interception, evaporation, transpiration, infiltration and storage that will lead to alterations in peak streamflow, base flow and water yield; increase in erosion rates and surface runoff; and, decrease in groundwater recharge (Greenbaum *et al.,* 2021; Poon and Kinoshita, 2018; Bart, 2016; Nolan *et al.,* 2015; Ebel *et al.,* 2012; Shakesby, 2011; Shakesby & Doerr, 2006; Martin & Moody, 2001).

The magnitude and response time of the hydrological processes of forests to rainfall events after wildfires may vary according to the size of the catchment, geography, climate, and pre-fire conditions and post-fire climate and it will be determined by a combination of several factors including pre-fire land attributes (vegetation type and land use), fire characteristics (intensity, duration, magnitude), geology and previous soil composition (organic matter content, aggregate stability and soil water), among others (Greenbaum *et al.,* 2021; Loiselle *et al.,* 2020; Shakesby & Doerr, 2006; Neary *et al.,* 2005).

Besides the quantitative effects, wildfires also have impacts in water quality (Rhoades *et al.,* 2019; Feikema *et al.,* 2011). They may potentially contaminate irrigation water with sediment and ash, and increase fluxes of carbon, nutrients, and other water quality constituents into the water table (Hohner *et al.,* 2019; Balfour *et al.,* 2014; Smith *et al.,* 2011).

The deposition of the ashes combined with a potential rise in the amount of sediments delivered to the streams due to the increase in the erosion rates may lead to deteriorated physical and chemical water quality, with potentially substantial and long-lasting effects that can affect the water supply for the communities (Balocchi *et al.,* 2020; Hohner *et al.,* 2019; Bladon *et al.,* 2014).

The factors that affect the type and severity of post-forest fire water quality issues vary substantially from place to place depending on the amount and intensity of precipitation, soil and vegetation cover characteristics, geology, topography and the intensity of the fire. The main water quality problems associated with fire events include turbidity, flooding, increased temperature, change in soil properties, rise in the production of nutrients and biological demand and decrease in dissolved oxygen (Tecle & Neary, 2015).

Forest fires may also increase the store of already existing contaminants, allowing their mobilization into the streams and even enhance the concentration of additional pollutants that are normally present in the natural environment in negligible concentrations (Smith *et al.,* 2011). Organic carbon is an important indicator of water quality after fire events due to its ability to transport heavy metals and organic contaminants, as well as support bacteria and biofilm (Loiselle *et al.,* 2020; Laudon *et al.,* 2012).

The effects of wildfires in different compartments of the environment have been extensively documented over the years, although direct effects of these events on groundwater quality are still poorly investigated and understood, representing a threat, especially to communities which supply completely relies on these resources (Rodrigues *et al.,* 2019).

2.3) Wildfires and climate change

Wildfires have always been natural disturbances in the environment, but nowadays, many studies reported the possible relationship between the changing wildfire activity in many regions of the planet and climate change (Calheiros *et al.,* 2021; Azari *et al.,* 2017; Chaplot, 2007).

According to Parente *et al.* (2019), most of the recent extreme fire events occurred in Europe were driven by the occurrence of extreme weather events such as severe droughts and heatwaves in the last couple decades.

Climate change has increased the frequency, size and severity of natural disturbance events such as wildfires, droughts and storms in different regions of the world (IPCC, 2013; Seidl *et al.,* 2017; Loiselle *et al.,* 2020) and according to the IPCC report (2013), the increment of CO_2 in the atmosphere will increase the temperature and change precipitation patterns in the Earth until 2100. The increasing of extreme climate events with warmer and drier weather, longer fire seasons, higher frequency of droughts and thunderstorms, changes the burning conditions and leads to more frequent and intense wildfires (Turco *et al.,* 2019; Robinne *et al.,* 2020).

In Europe, the years from 2015 – 2019 were the hottest recorded since 1850 (beginning of global temperature tracking). It is estimated to be 1.1°C above pre-industrial times (1850-1900) and 0.2°C warmer than 2011-2015. A study using the PESETA IV model to assess the consequences of climate change on several categories (human mortality from heat and cold waves, windstorms, water resources, droughts, river flooding, coastal flooding, wildfires, habitat loss, forest ecosystems, agriculture and energy supply) reported that using the period from 1981-2010 as a reference, global average temperature was already 0.8°C higher on average when compared to pre-industrial times. Regarding precipitation, the higher the increase in the temperature, the more significant changes are projected with increases up until 15% in northern Europe and decreases until -15% in the Mediterranean countries (Feyen *et al.,* 2020).

Changing weather conditions associated with global warming as a result of higher temperatures and dryer conditions, could increase even more the fire danger in most of the European countries (Figure 6) but this increase will probably be sharp in Southern Europe (Portugal, Spain. Southern France, Italy and Greece) (Calheiros *et al.,* 2021; Feyen *et al.,* 2020).

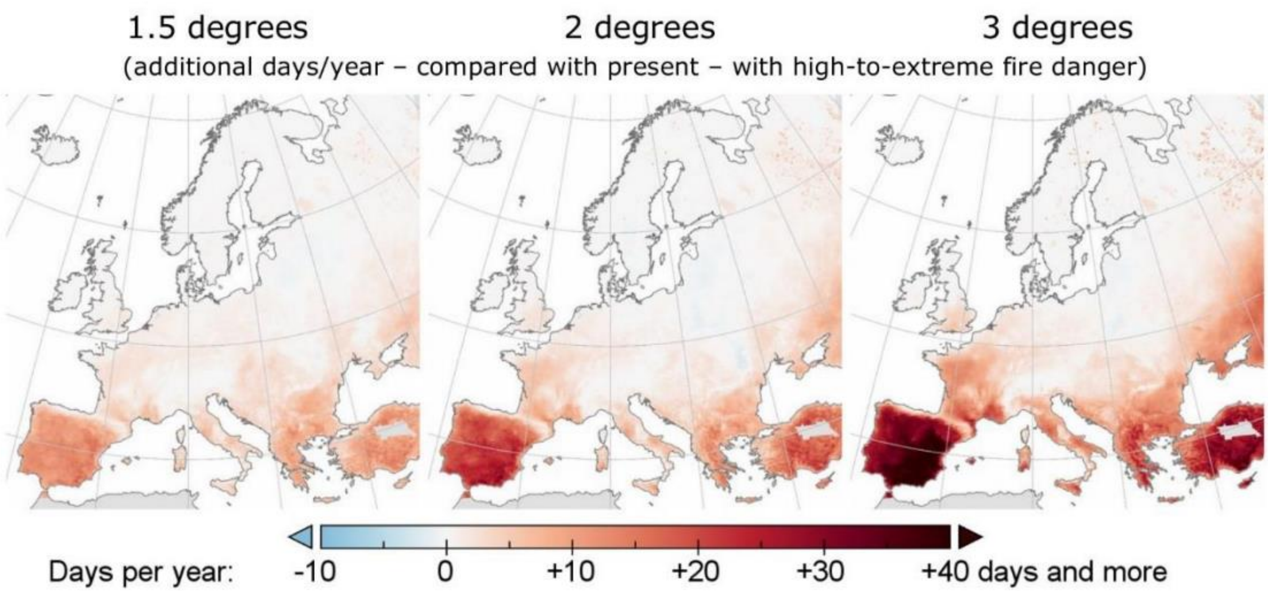


Figure 6 - Additional number of days per year with high-to-extreme fire danger (daily Fire Weather Index ≥ 30) for different levels of global warming compared to present (1981-2010) (Feyen *et al.,* 2020).

Climate variability (at monthly and longer timescales) and weather conditions (especially precipitation, wind, air temperature and humidity) are determinant factors in global fire activity, because these actors have a very strong influence on fuel availability, flammability, fire spread (Calheiros *et al.,* 2020). Although, in some regions, human activities can also disturb the relationship between climate and fire occurrence (Abatzoglou *et al.,*2018). In Portugal, socioeconomic and landscape changes favor the occurrence of extensive wildfires and can increase or decrease the number of ignitions, fire size and change the fire regime (Calheiros *et al.,* 2021; González-Ávila *et al.,* 2020; Parente *et al.,* 2018a, 2018b).

Several studies suggest a relationship between climate conditions and wildfires (e.g. Vieira *et al.,* 2020), and according to Pérez-Sánchez et al. (2019) there is an increase in fire risk for future climate conditions in the Iberian Peninsula.

3) Regional settings

The geographical, climatological, geological, and hydrogeological settings regarding the study area are descripted in this chapter.

The study area consists of the Vieira de Leiria-Marinha Grande aquifer with an area of about 320 km². It is located in the district of Leiria (Central Portugal) and comprises four municipalities: Alcobaça, Marinha Grande, Leiria and Nazaré (Figure 7).

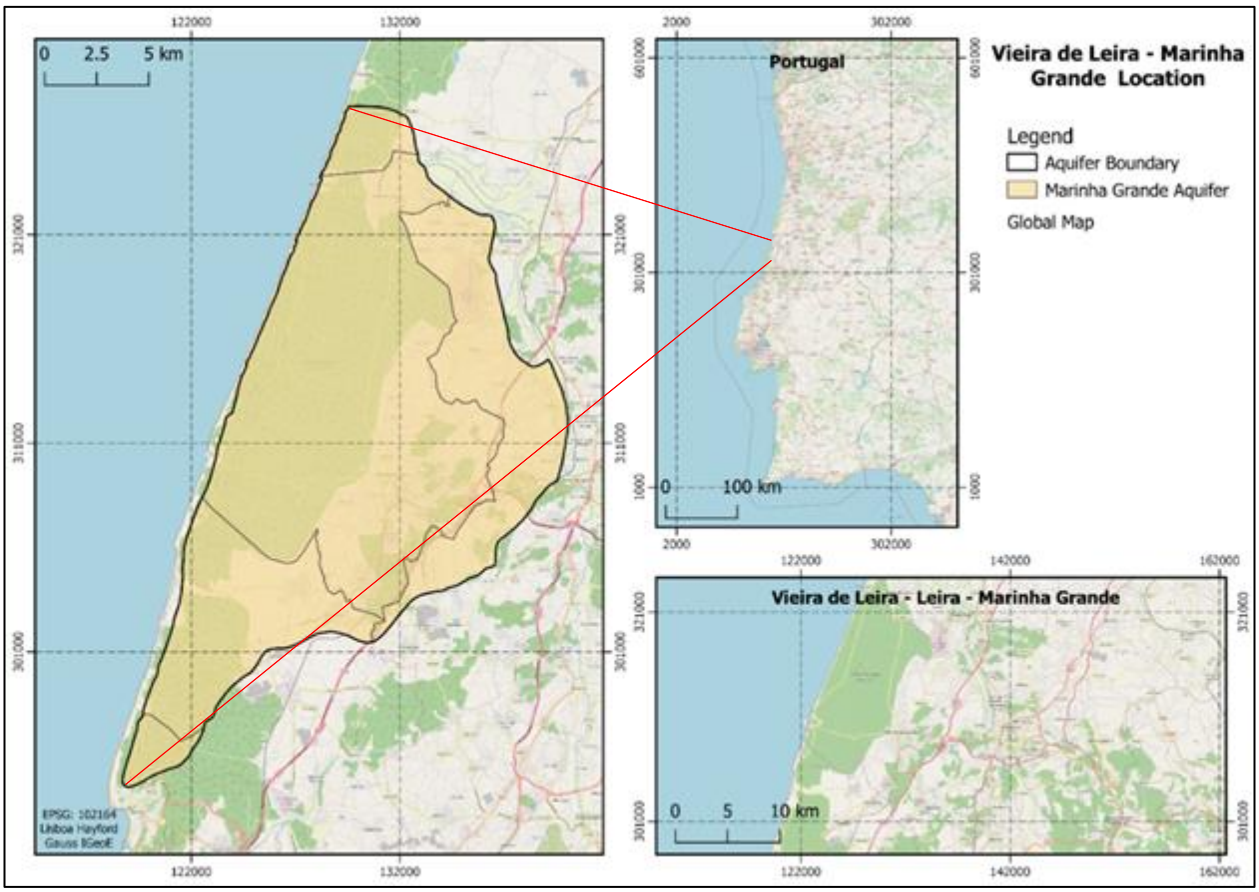


Figure 7 - Location of study area (Vieira de Leiria - Marinha Grande aquifer) in the Leiria District – Central Portugal.

3.1) Climate

The climate in the area is classified as Mediterranean Temperate with average annual rainfall of about 750 mm (based on the data from E-OBS for the period of 2001-2020). It presents strong seasonality in precipitation distribution due to latitude, orography, and continental-oceanic influences. June, July and August are the driest months, and November, December and January are the wettest ones, with higher precipitation values. The average temperature is approximately 16°C with relatively low and high values in winter and summer, respectively (Figure 8).

Besides the high seasonal variability, the region also presents strong interannual climatic variability, with precipitation values ranging from 400mm in a very dry year to 1100mm in a wet year according to the time series obtained in the E-OBS database.

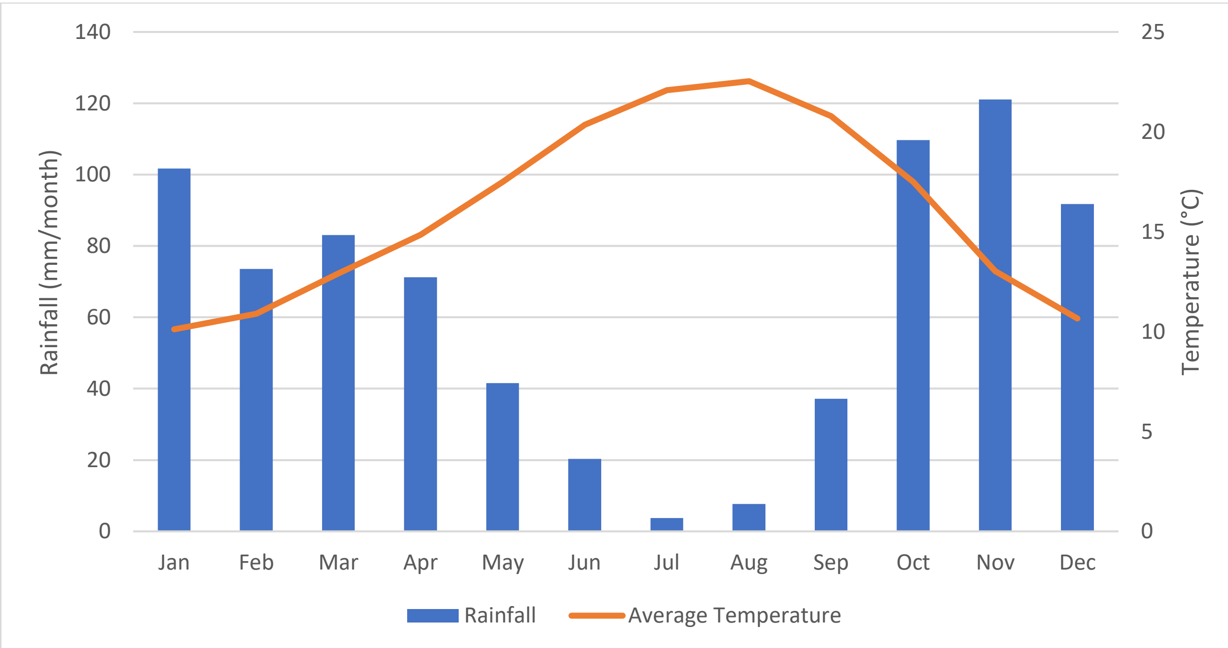


Figure 8 - Climatological data for the study area takes from the E-OBS database for the period of 2001-2020. The precipitation data was corrected using the meteorological stations (Batalha, Caranguejeira, Crespos, Leiria, Maceira, Mata da Bidoeira, Monte Real, Porto de Mós e Santa Catarina da Serra) of the Portuguese Water Resources System (SNRH) in the area.

3.2) Topography

The topographic gradient in the study area is relatively low and the altitudes range between over 170 m above sea level in the southeast to around 0 m (shore with Atlantic Sea) in the northwest part (Figure 9).

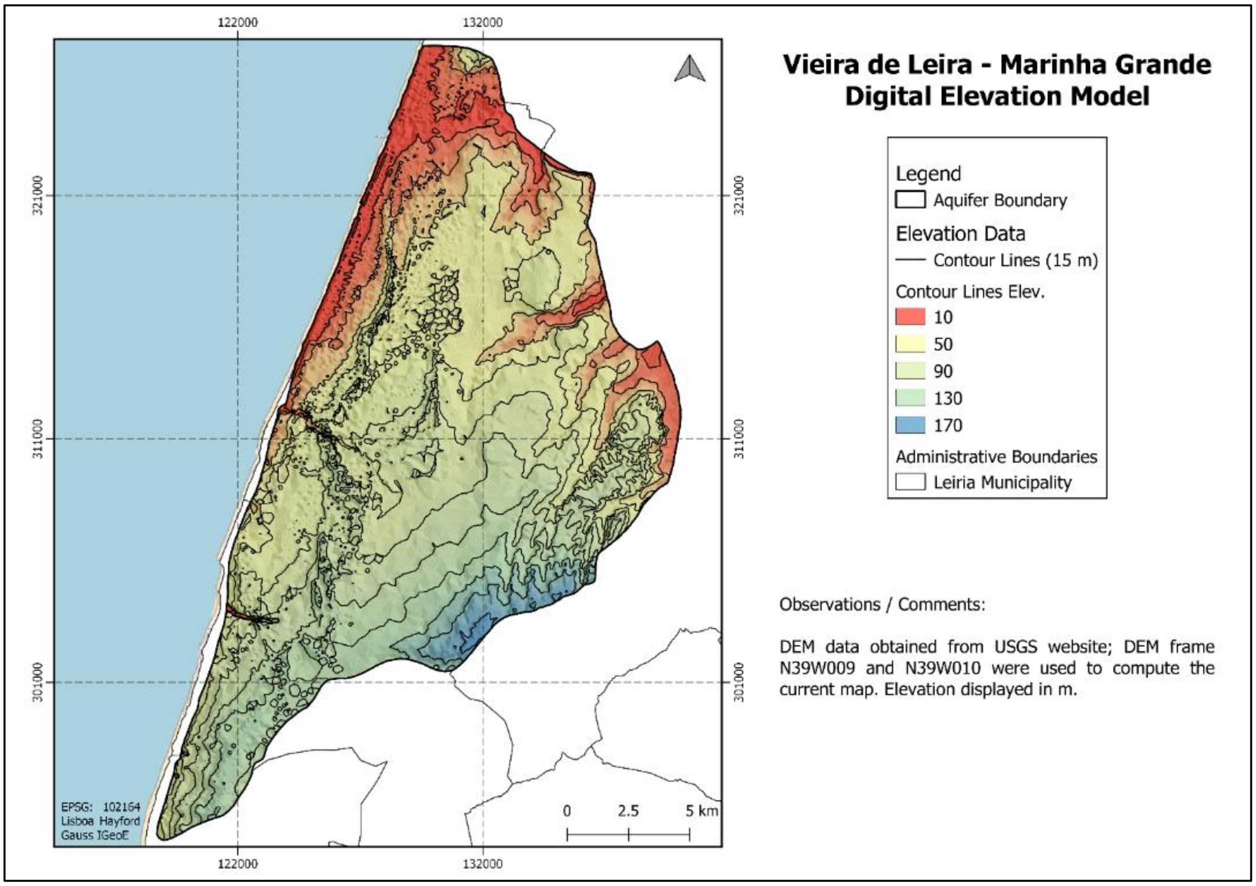


Figure 9 - Topography map of the study area elaborated using the Digital Elevation Model (DEM).

A simplified cross section based on GPR survey conducted in the study area is given in Figure 10.

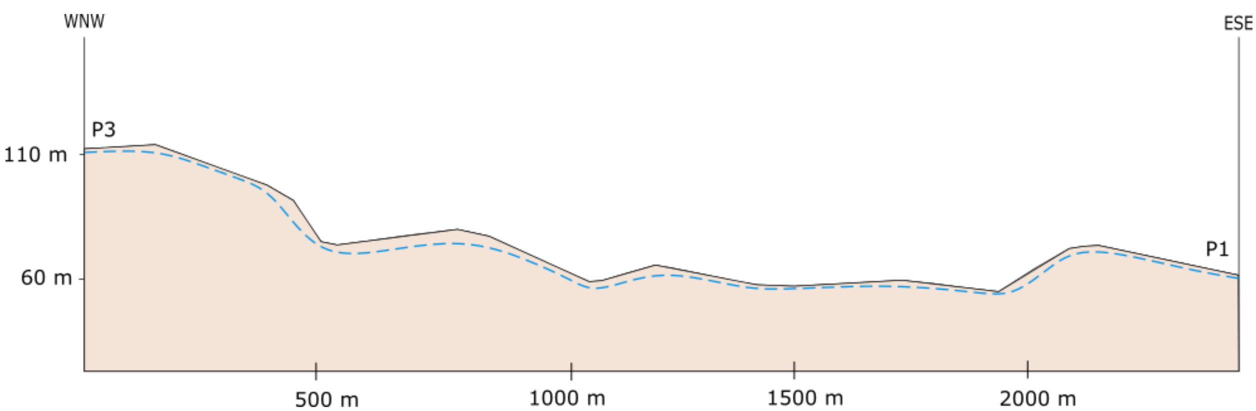


Figure 10 – Simplified cross section of the study area based on the GPR profile (MG1) (Figure 22) (The dashed blue line represents the water level in the area).

Although the topographic gradient is smooth if we consider the whole extent of the study area, the dune system causes local abrupt topographic differences that may interfere in the hydrological conditions of the shallow aquifer.

3.3) Landuse

According to the data from the Corinne Land Cover Project from 2018, developed by the Copernicus Land Monitoring Service, more than 70% of the study area is covered by forest, non-agricultural vegetation areas, scrub, herbaceous vegetation, and pastures. Besides, there are agricultural activities and several factories that produce products like glass operate in the area (Figure 11).

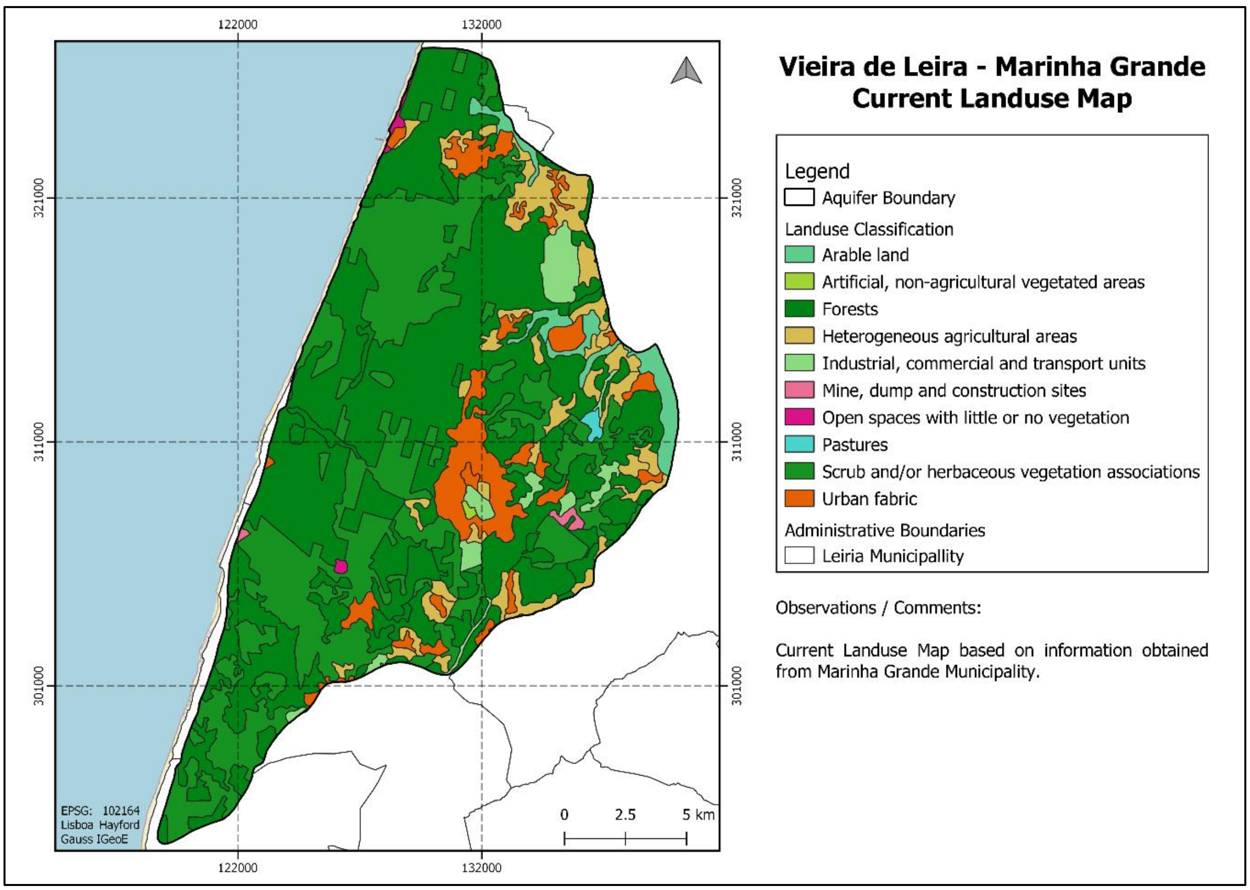


Figure 11 - Land use map of the study area obtained using the data from the Corinne Land Cover 2018 project.

The forested area known as Pinhal de Leiria or Pinhal do Rei covers approximately 111 km² of the study area and consists of a maritime pine tree plantation from the XIII century with the aim to contain the invasion of sand dunes inland close to the city of Leiria.

The first reports of wildfires in the region are from 1806 (Silva & Batalha, 1859) and since then, several others have been reported in the area throughout history. The most recent occurred in October 2017 and burnt about 86% of the Pinhal de Leiria forest, changing completely the characteristics of the land cover and land use.

Nowadays, more than three years after the fire, only 11% of the area was reforested and specialists are worried about the development of eucalyptus trees and the presence of invasive species like acacias that may make the recovery and development of the new pine trees even more difficult.

3.4) Hydrology

The study area is located in the Lis river basin and part of the Vouga, Mondego, Lis e Ribeiras do Oeste river management plan (APA, 2016). The dominant orientation of the rivers and streams is NE-SW, but there are also rivers with E-W direction region, which coincides with the direction of the main geological faults (Figure 12).

The main river in the study area is the River Lis and its affluents. The river source is located in Fontes (Leiria District), at about 400 m of altitude, in a predominantly limestone region and flows downstream for about 40 km until it reaches the Atlantic Ocean at Vieira de Leiria.

The Lis river basin is mostly constituted by sandstones, gravel, limestones and mudstones. It presents a high seasonal variation, which led to the construction of several weirs to raise the water level slightly on the upstream side of the main river.

Besides the Lis River, another important surface water body in the region is the Ribeira de São Pedro, a stream 6 km long which is located in the centre of the studied region and that is the only permanent river stream in the Leiria Pine Forest area, since its affluents normally get dry during summer.

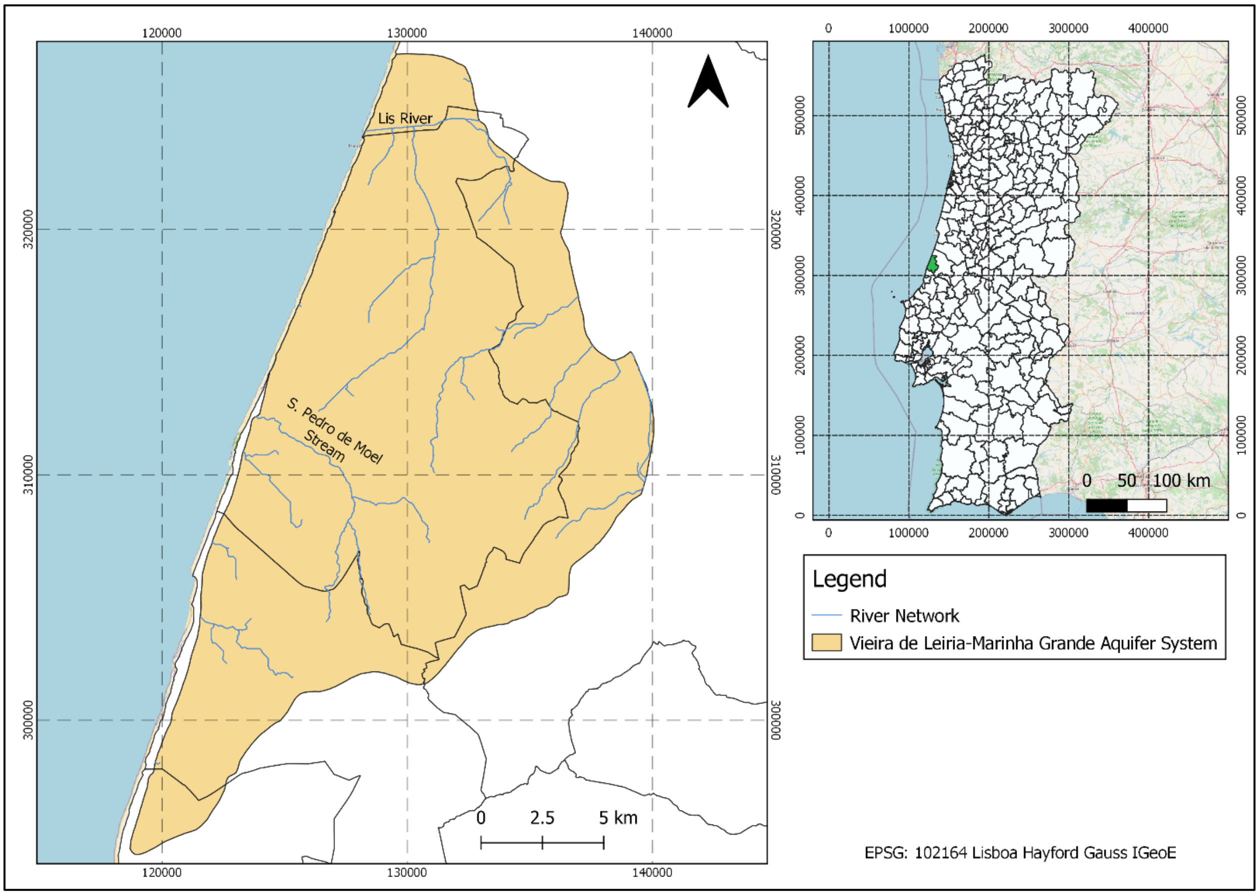


Figure 12 - River network with the Lis River in the North and East border and the São Pedro de Moel stream in the central part of the study area.

3.5) Geology

The study area is located in the Lusitanian sedimentary basin (LSB) (Figure 13) which formation is related to the opening of the North Atlantic Ocean during the breakup of Pangea supercontinent during the Mesozoic.

The oldest geological units from the Lusitanian basin are from Late Triassic age (approximately 237 Ma) and are above Paleozoic formations. The sedimentation develops until the Cenomanian (100.5 Ma), although the majority of the sediments in the basin are Jurassic (201 to 145 Ma).

Above this set of rocks, there are Cenozoic deposits related to the inversion movements due to Alpine collision (Kullberg *et al.,* 2013; Rey *et al.,* 2006).

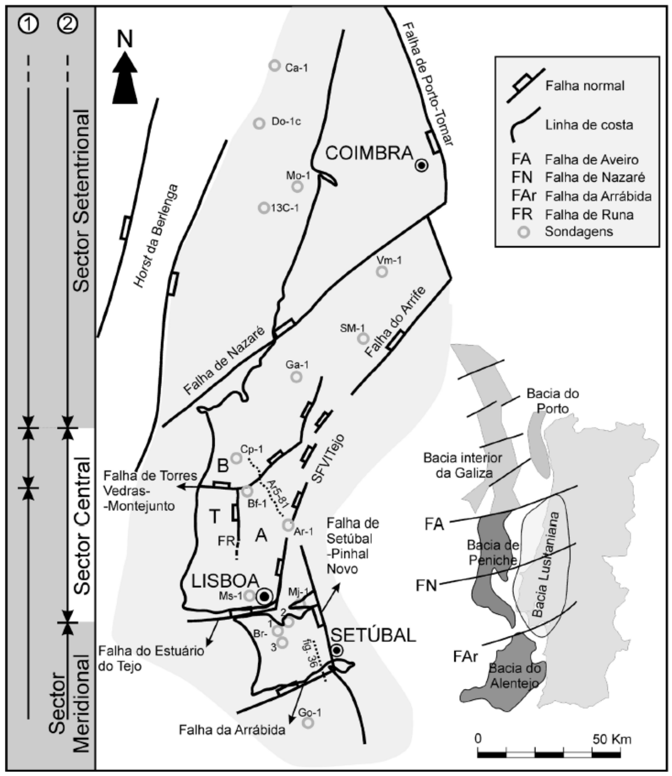


Figure 13 - Geological setting of the Lusitanian sedimentary basin (Kullberg *et al.,* 2013).

The Lusitanian Sedimentary Basin consists in a rift basin (Kullberg *et al.,* 2013) developed as a result of four Mesozoic rifting phases followed by Cenozoic inversion movements due to Alpine collision (Rasmussen *et al.,* 1998; Wilson *et al.,* 1989).

According to Kullberg (2000) the first rifting phase (Late Triassic – Early Jurassic) was concentrated in the central part of the basin and would be related to the early tension phases in Pangea that would lead to the opening of the Atlantic Ocean. A second rifting phase (Early to Middle Jurassic) occurred as a results of small salt movements along the bigger faults, provoking a big transformation in the geometry and kinematics of the basin. The third rift phase (Late Jurassic) is characterized by an increase in the extension forces that reactivate some existing faults, activated new ones, and promoted some diapiric related activity (Mateus *et al.,* 2017). This period is pointed by Dias et al. (2013) as an important one for development of the sub-basins tectonics. The final phase (Late Jurassic – Early Cretaceous) triggered salt movement and the development of salt structures in the basin (Rasmussen *et al.,* 1998). After the end of the extensional forces in the Late Cretaceous, the inversion of the basin started, and continued during the Cenozoic.

The several sets of geological faults in the LSB promote a strong compartmentalization thus, the structure of the basin is influenced not only by the geometry of its borders, but also by the orientation of the faults (mainly between NE-SW e E-W) that will perform an important role in the delimitation of sectors with distinct tectonic-sedimentary evolutions and in the development of the sedimentary fill inside the basin (Kullberg *et al.,* 2013).

Salt tectonics is very important in the LSB development. The salt formations evolved above active faults and its appearance was triggered by the inversion tectonics occurred in the Cenozoic (Kullberg, 2000). Most of the diapires are located along tectonic faults following to main directions NNE-SSW to N-S and ENE-WSW to E-W.

The Dagorda Formation is the responsible for the diapiric movements in the LSB and according to Rasmussen (1998), the migration of the evaporitic units, mainly composed by evaporitic mudstones led to the appearance of several anticlines, of asymmetrical profile, and the creation of sub-basins with subsidence which are located in the deeper parts of Lusitanian Basin. Besides, the diapiric tectonics also plays a decisive role in the distribution of the thickness of the geological layers and in their spatial organization in the subsoil.

3.5.1) Lithostratigraphy

The study area is located in the northern section of the Lusitanian basin with the Nazaré fault representing the border to the central section (Dias *et al.,* 2013) and its sedimentary deposits range from the Upper Cretaceous to Holocene (Figure 14).

The Holocene sedimentary deposits (approximately 11700 years) are mainly consisted by beach sands and sand dunes extended along the coast. The beach sands consist of small width but continuous deposits on the littoral. The sand dunes represent the majority of the Quaternary deposits in the area and extend up until 7.5 km inland between São Pedro de Moel and Marinha Grande (Zbyszewski and Torre de Assunção, 1965). The sand dunes are composed by well-sorted very fine to fine sands and can develop structures able to reach more 50 m height creating preferential infiltration zones.

Results from the Ground Penetrating Radar (GPR) investigations presented in chapter 3.6 suggest the presence of clay (possibly iron enriched) layers in some regions in the first 10 meters underground. These layers could represent low conductivity zones, locally reducing infiltration.

Plio- and Plistocenic sedimentary deposits (11700 years to 5.3 My) consist in fine to medium sands with conglomerates and yellowish gray clay intercalations and can be found within a large region in the eastern part of the study area. The Pliocene sedimentary outcrops in the area are likely a result of the steep topographic gradient, as more recent Quaternary dune deposits have overlaid the Pliocene units in less steep coastal areas. In the East portion of the study area, sand dunes and pliocenic sand mix among themselves making very hard to differentiate them.

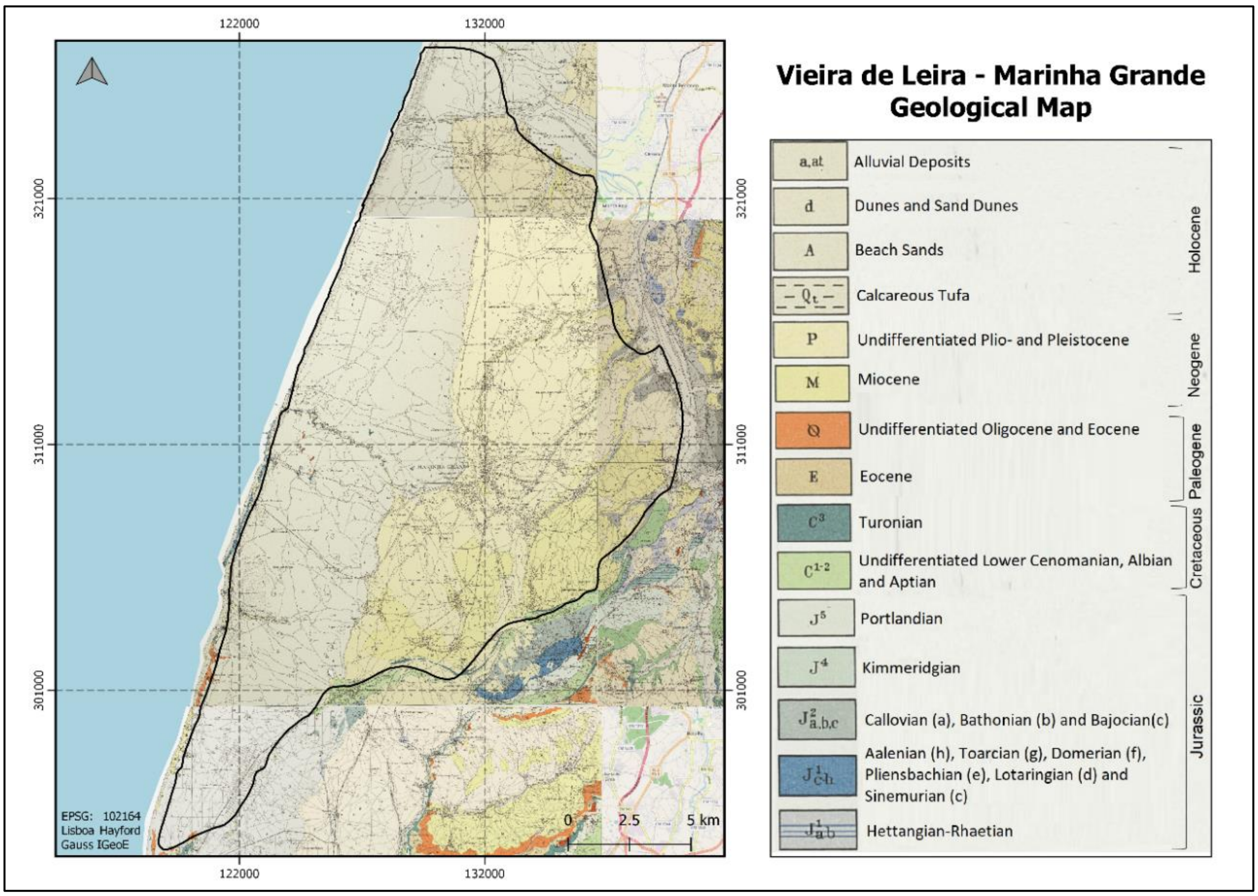


Figure 14 - Geological map of the study area obtained using the data from Zbyszewski et al., 1965.

Below these units, it is possible to find Miocene deposits (5.3 to 23 My) consisting of clayey medium to coarse sandstones of a yellow/brown color, with conglomeratic levels, clay layers and yellowish to brownish carbonate material (Almeida *et al.,* 2000) and can be found in the southeastern bord of the study area.

The Oligocene (23 to 34 My) deposits positioned stratigraphically below the Miocene unit consist in interbedded marls, mudstones, sandstones, and limestones, with some sedimentary units including coarse grained material such as angular pebbles. The alternating sandstones and mudstone deposits suggest a transitional paleo-environment.

The oldest sediments present in the region are from the Upper Cretaceous and consist in a carbonate complex represented by white to pinkish limestones that can be identified at depth in some boreholes (Zbyszewski and Torre de Assunção, 1965). These formations cover the Portlandian sandstone and marly formations from the Late Jurassic.

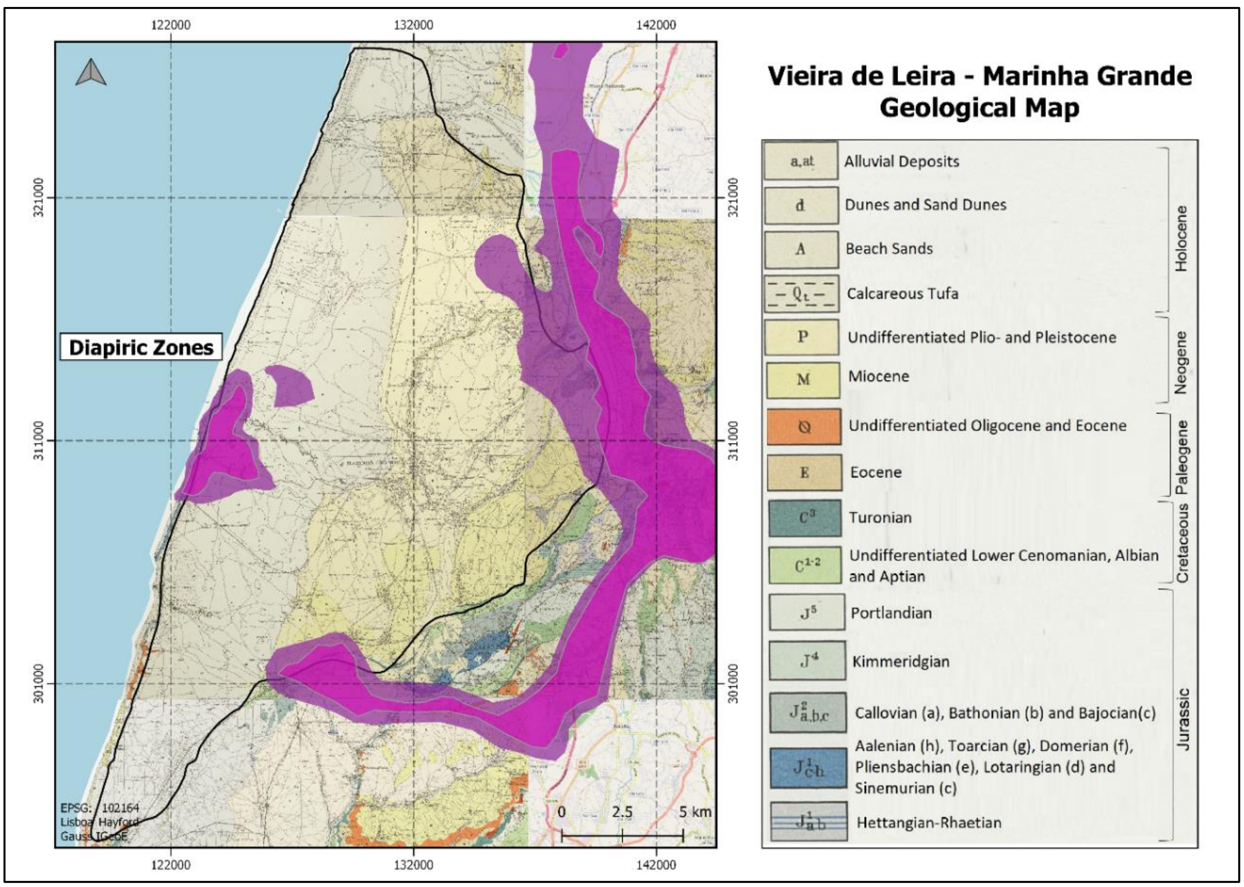


Figure 15 - Geology map of the study area with diapiric structures (shown in purple) (Adapted from Dias, 2017).

Several diapiric formations have been mapped over the years in the LSB in Caldas da Rainha, Porto de Mós - Rio Maior, Bolhos, Santa Cruz, Matacães, Sesimbra, Soure and Ervedeira as well as in the study area, namely in São Pedro de Moel, Leiria-Parceiros and Monte Real (Dias, 2017) (Figure 15).

3.6) Geophysical survey using Ground Penetrating Radar

A geophysical investigation was conducted in the area aiming to identify existing geological features and the potential presence of groundwater in the subsoil, determining is depth, extent and spatial distribution.

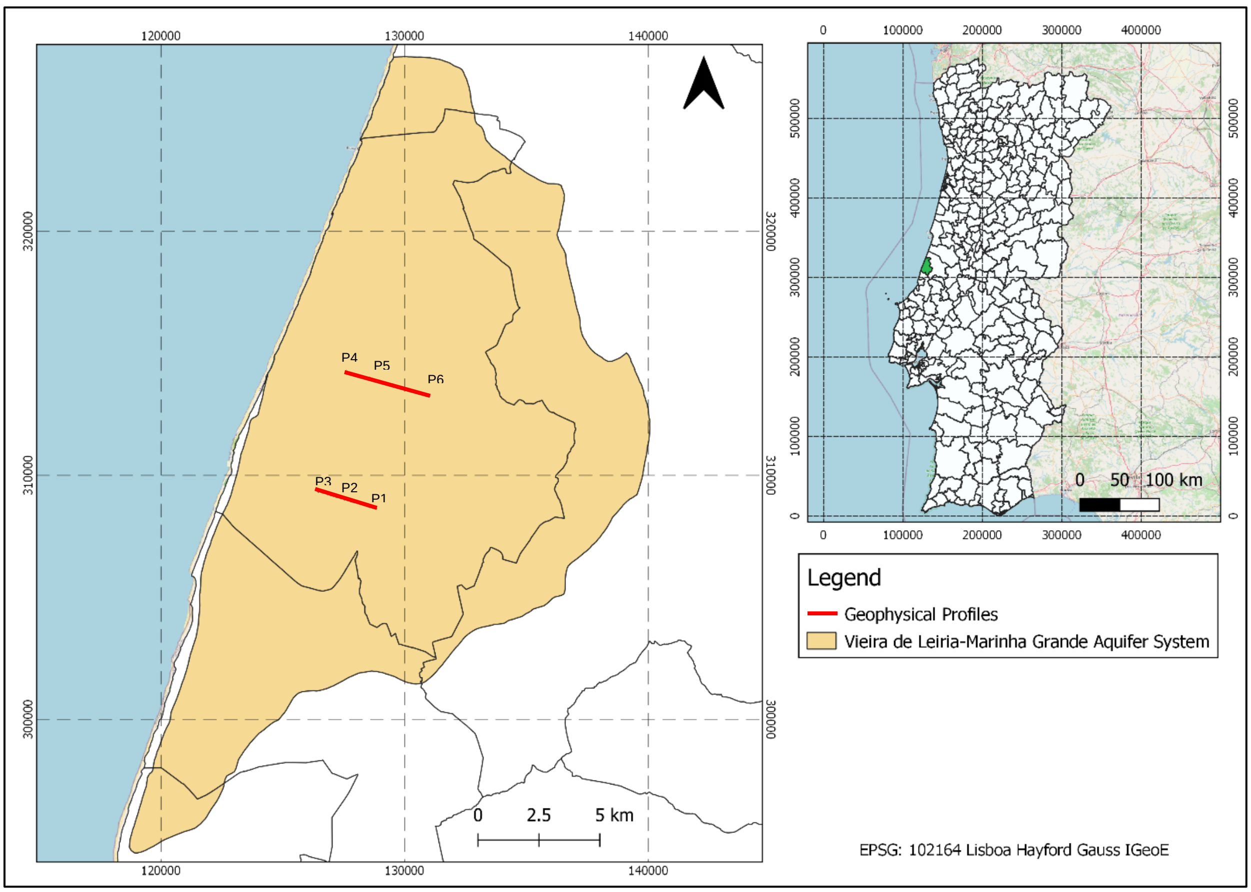


Figure 16 – Location of the Geophysical Profiles' elaborated during the field work.

In order to achieve it, six geophysical profiles (Profile 1, Profile 2, Profile 3, Profile 4, Profile 5, Profile 6) grouped in 2 main profiles (MG1 – Profiles 1,2 and 3 and MG2 – Profiles 4,5,and 6) with total length of about total of about 6,032 meters were selected (Figure 16) and investigated using

GPR technology (pulseEKKO) using an antenna with frequency of 100MHz. The main interferences able to affect the obtained results were: (1) presence of metallic structures, (2) presence of electric transmission lines, and (3) dunes.

The GPR consists of an active geophysical method based on the emission and reception of electromagnetic impulses and is used to acquire and register information related to the shallow subsoil in several scientific areas like geology, engineering, archaeology, among others.

In geological studies, it is normally applied to detect of pores filled with air underground, to identify geological features of interest, to map the subsoil stratigraphy, and to detect the interface of materials with contrasting electromagnetic properties (Electrical conductivity, dielectric permittivity and magnetic permeability). These properties will determine the velocity and attenuation of the electromagnetic impulses.

The data acquisition is done by moving the antenna equipped with a transmitter and a receptor along the desired profile. The electromagnetic waves are sent by the emission antenna and received in the receptor after being partially reflected in the subsoil. The antenna is linked toa control unit able to control the acquisition parameters and store the data (Figure 17).

Figure 17 – General aspects of the study area and GPR equipment and preparation during fieldwork conducted on the 24/06/2021.

The data is registered and visualized as 2D profiles called (radargrams) that may be interpreted with or without image visualization techniques and/or processing.

The equipment used in the geophysical survey in the study area is described in (Table 1).

Table 1 - Equipment and acquisition methods used in the geophysical survey in the Marinha Grande region on the 24/06/2021.

Equipment and Data Acquisition	
GPR	pulseEKKO Sensors & Software System with an 100MHz antenna with bistatic antennas, non-blinded
Geolocation	EMLID Reach RS+ differential GPS
Data Treatment	Sensors & Software EKKO_Project 5
Data Processing	Reflex-Win v9.0.
Method	Reflection
Acquisition method and trigger	Continuous, Odometer
Antenna	Bistatic, Non-blinded
Frequency	100MHz
Distance between the antennas	Common Offset: 40 cm
Temporal interval	350,4 ns
Velocity	0.100 m/ns
Distance between the pulses	30 cm

The data processing was conducted according to the methodology presented in Daniels (2000) and Annan et al. (1992) that consists in: (1) Time zero adjustment; (2) Dewow; (3) Background removal, (4) Bandpass butterworth, (5) Determination of the signal propagation velocity; (6) Data migration and, (7) Volumetric Interpolation (Figure 18).

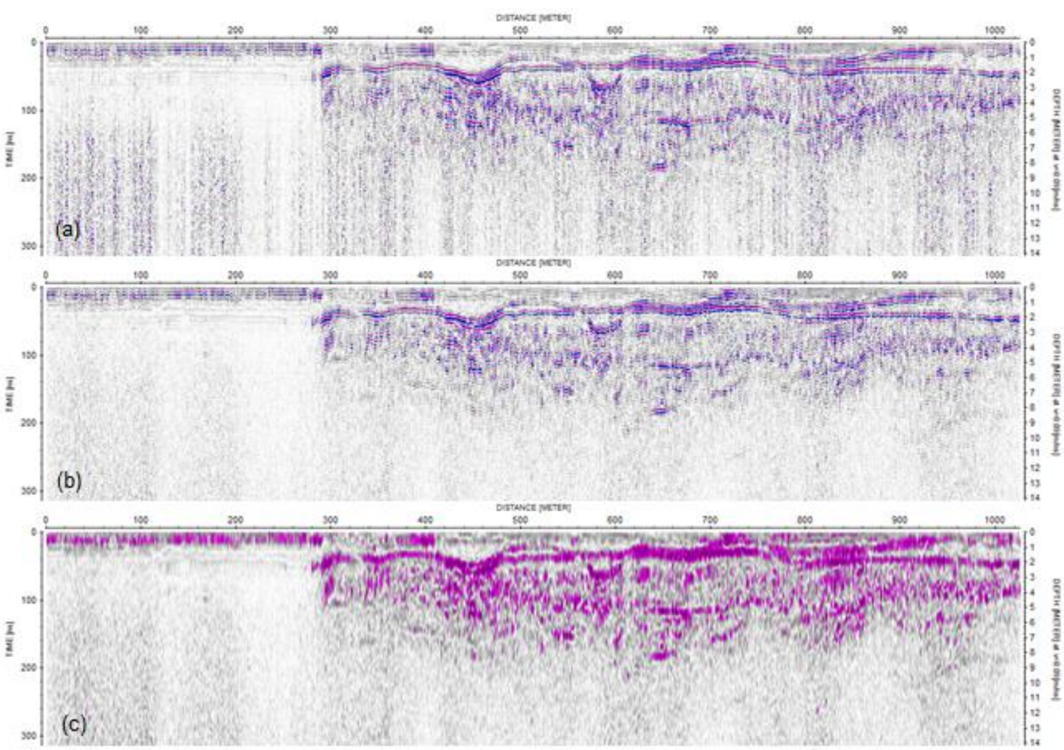


Figure 18 - Radargram Profile 1. (a) Bandpass Butterworth; (b) Data Migration and (c) Volumetric Interpolation.

The identification of the structures in the subsoil of the Marinha Grande region were investigated and the identification of these geological features are discussed below. The Figures 19 to 21 represent the radargrams of the 6 geophysical profiles investigated in the area and each color of arrow represent different types of structures of interest in the profile (Table 2).

Table 2 - Geological Structures detected in the radargrams of the geophysicial profiles in Marinha Grande.

	Reflectance Amplitude	**Geological Structures**
Yellow	High	Potential water levels
Red	Low	Soil/Sand
Green	Low	Cavities
Blue	High	Presence of Paleochannels
Orange	High	Clay Levels, possibly enriched in iron.
Brown	Low	Asphalt

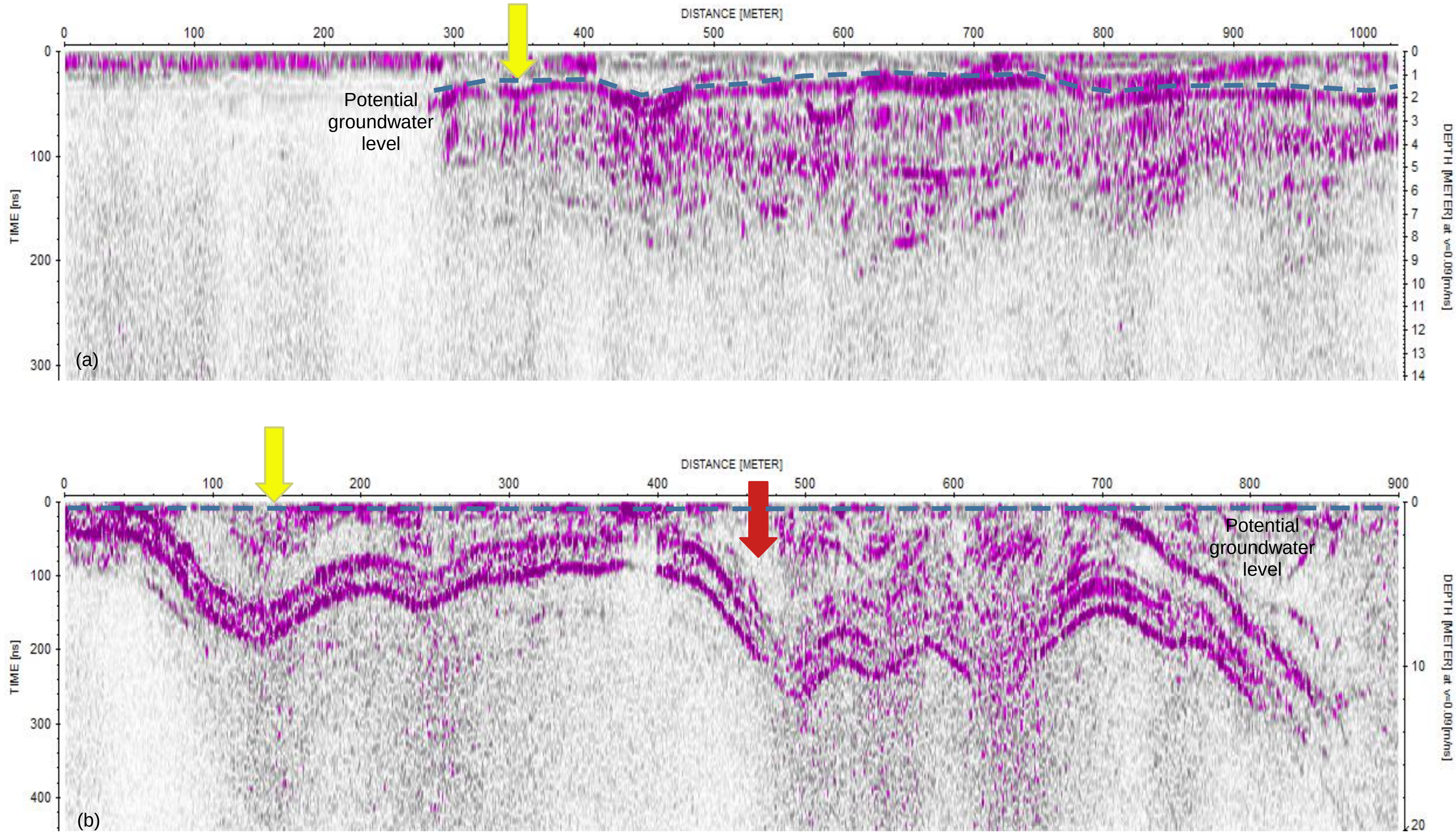


Figure 19 - Volumetric Interpolation. (a) Radargram Profile 1 and (b) Radargram Profile 2.

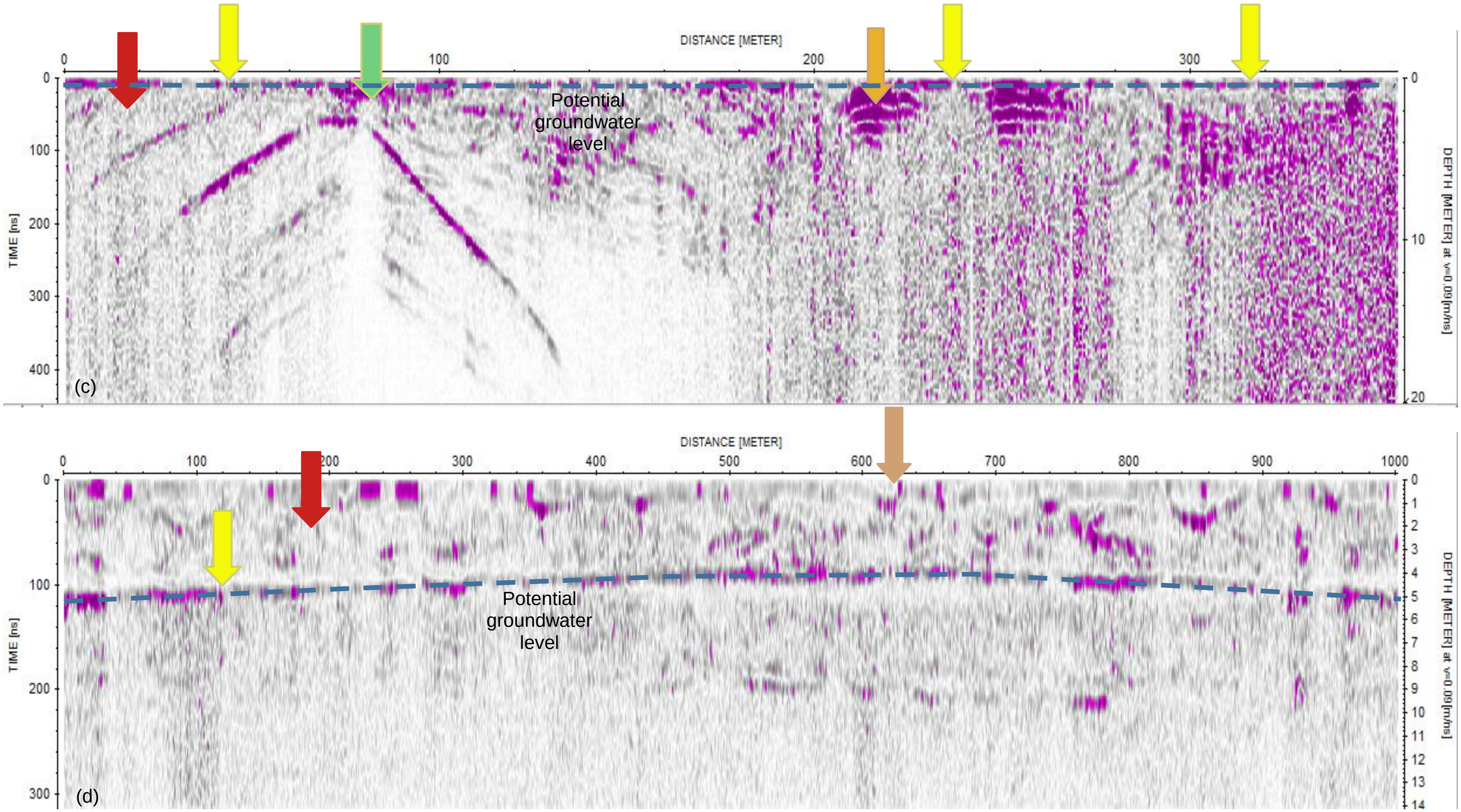


Figure 20 - Volumetric Interpolation. (c) Radargram Profile 3 and (d) Radargram Profile 4.

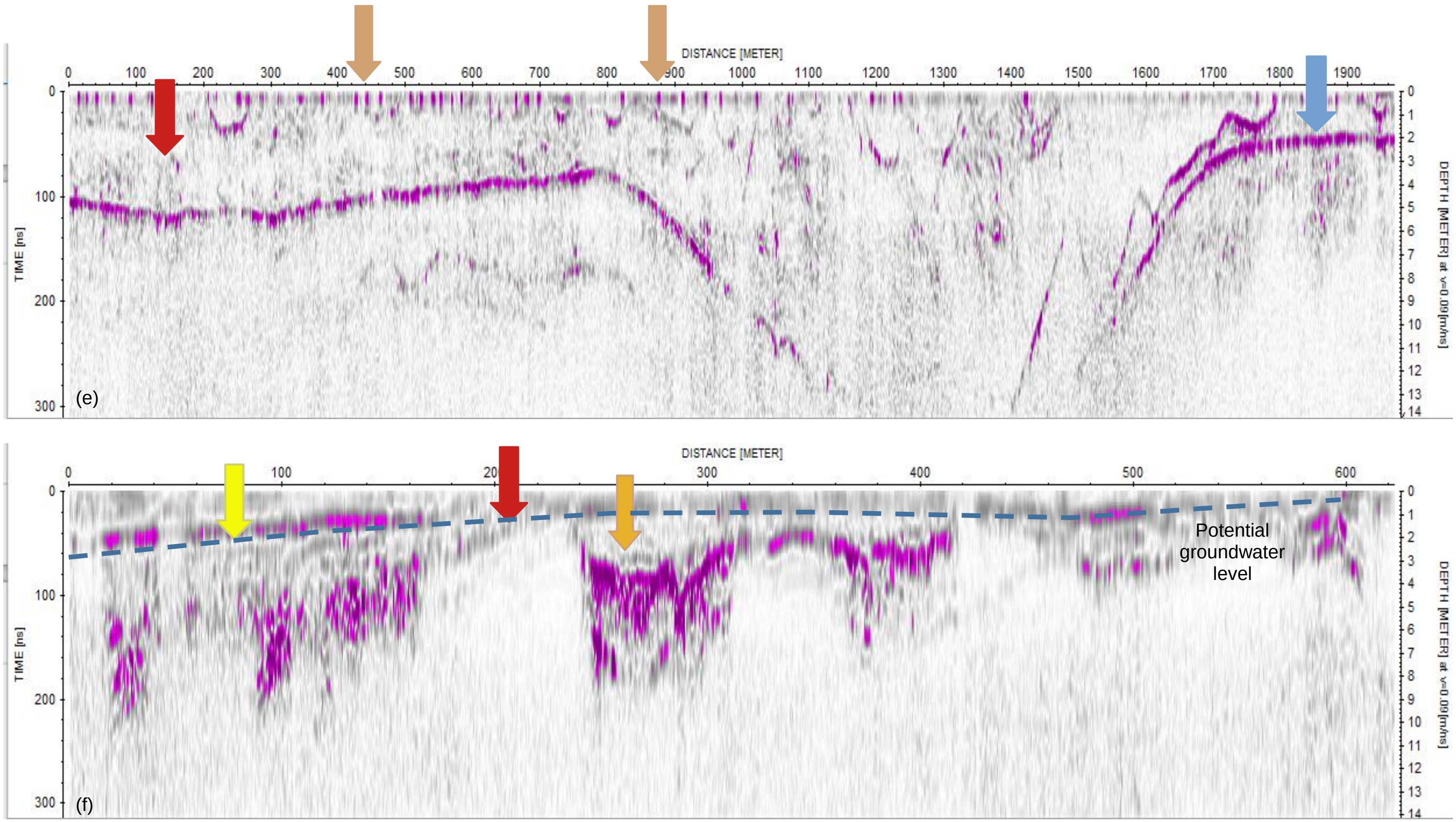


Figure 21 - Volumetric Interpolation. (e) Radargram Profile 5 and (f) Radargram Profile 6.

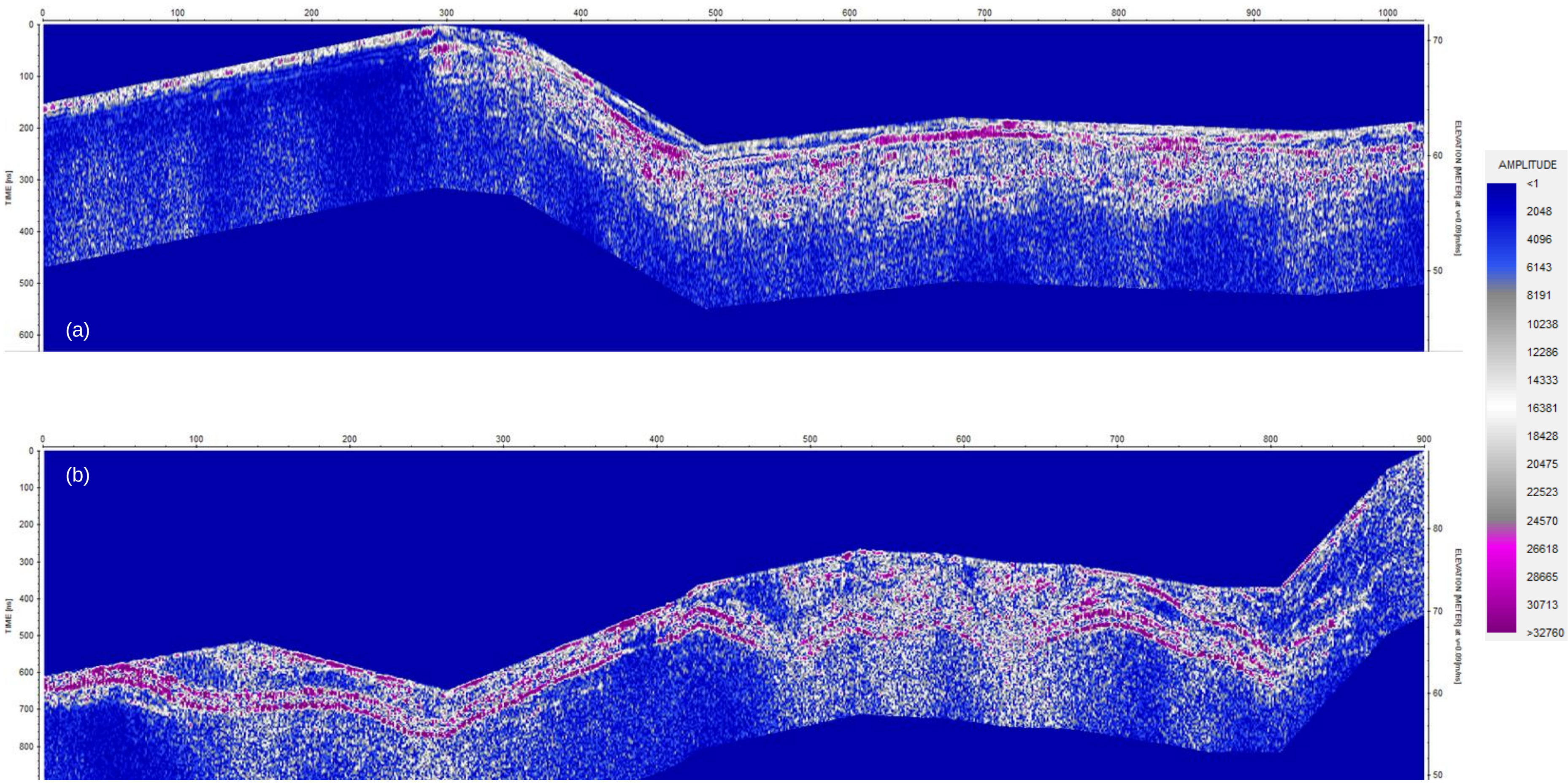


Figure 22 - Volumetric Interpolation with topographic correction. (a) Profile 1 and (b) Profile 2.

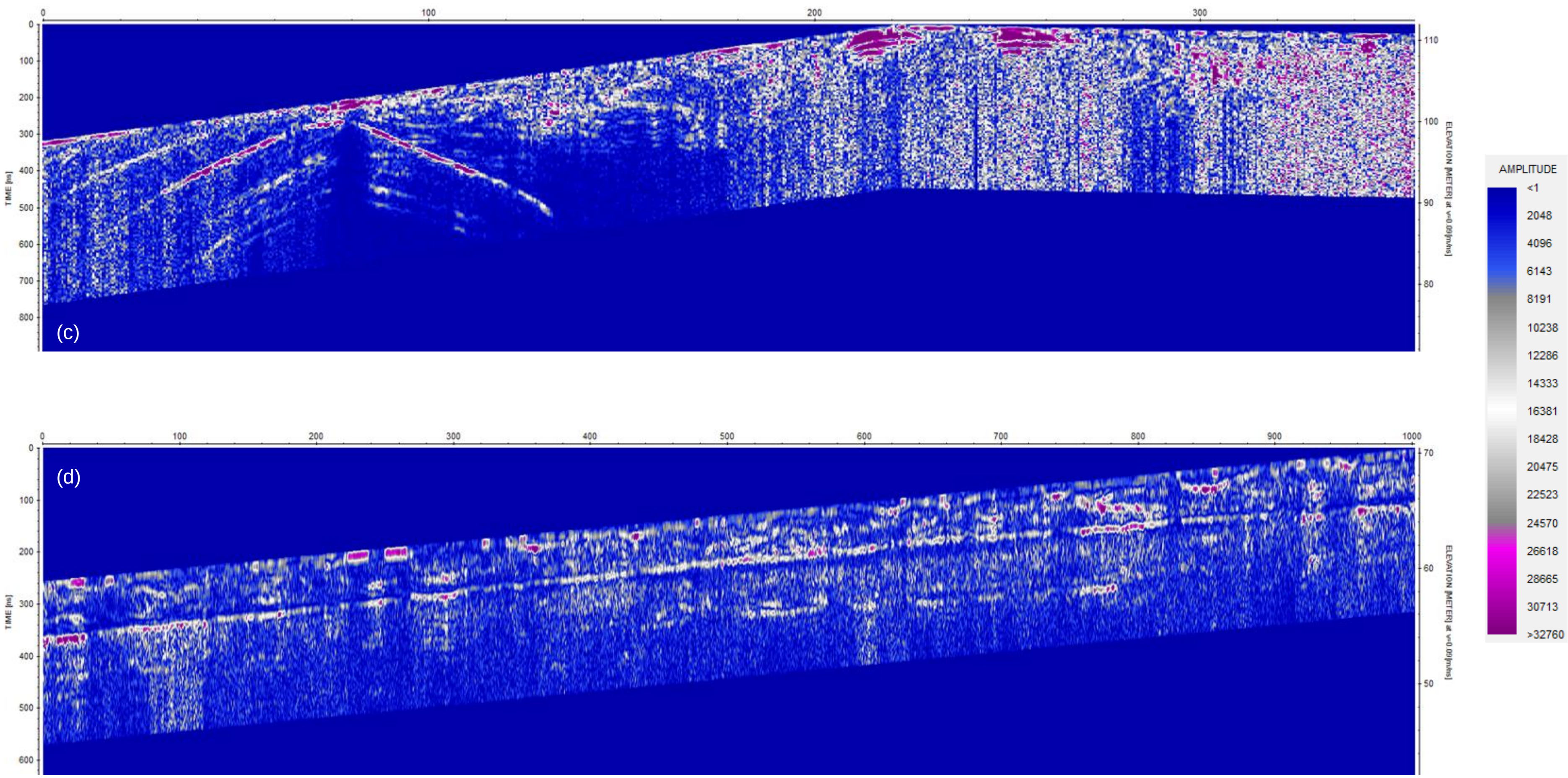


Figure 23 - Volumetric Interpolation with topographic correction. (c) Profile 3 and (d) Profile 4.

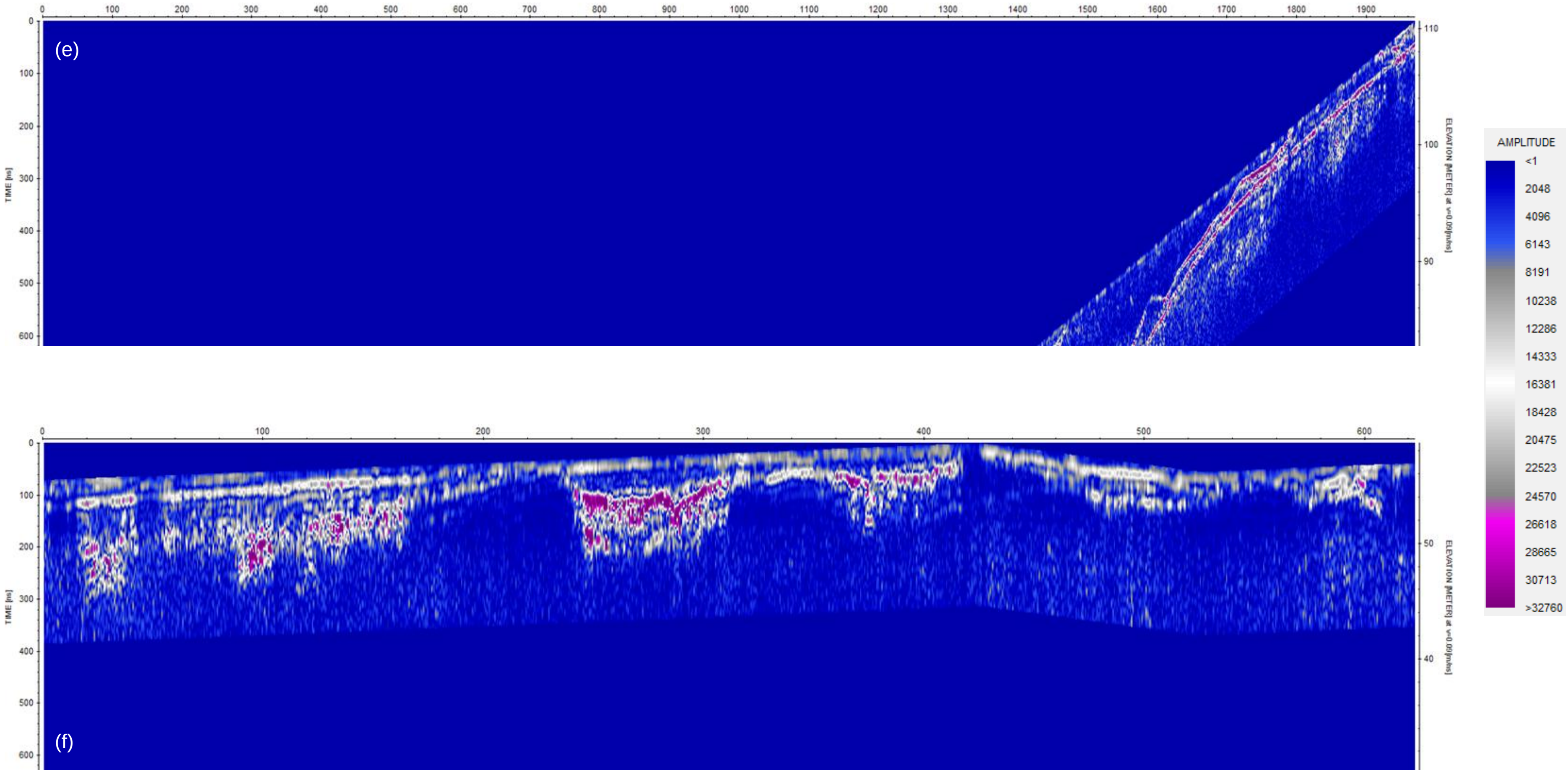


Figure 24 - Volumetric Interpolation with topographic correction. (e) Profile 5 and (f) Profile 6.

Figures 22 to 24 show the radargrams of the geophysical profiles topographically corrected. The depth of investigation varies between 15 to 20 meters and is conditioned by the subsoil constitution. These results show that the water level is located about 2 meters below the surface, reaching up to 10 meters in specific areas and that despite the predominance of sand dunes there are several small clay layers intercalated that may be able to affect the hydrological regime locally by reducing infiltration.

3.7) Hydrogeology

The Vieira de Leiria- Marinha Grande aquifer system (O12) located in the municipalities of Alcobaça, Leiria, Marinha Grande and Nazaré. It has an area of about 320 km² and is limited to the west by the Atlantic Ocean and to the southeast by the limb of an anticlinal fold of the Cretaceous sedimentary units (Wallström, 2019). The aquifer is composed of a multilayer system consisting mainly of Pleistocene, Pliocene and Miocene sands, overlaying Lower Cretaceous sandstones (Almeida et al., 2000; Dias et al., 2013).

The O12 aquifer is part of the Lis River basin and was classified in the River Management Plan elaborated by APA (2016) as a porous, moderately productive groundwater body, with an average recharge of about 95 hm³/year. According to Almeida et al. (2000) the average and median transmissivity, estimated using 47 discharge values are 230 m²/day and 80 m²/day, respectively.

Figure 25 – Shallow wells sampled in the study area during field work conducted on 29-30/07/2021 (a: W32; b: W34).

The Quaternary sand dunes and Pliocene sandstone units in the O12 aquifer usually present good hydrogeological conditions and feed several springs in the region (Zbyszewski & Torre de Assunção, 1965). The water table in the shallow unconfined aquifer is around 2 meters and since the population in the study area is highly dependent on groundwater, the use of hand dug wells for irrigation purposes is quite common (Figure 25).

There are historic cases of abandoned deep wells in the region due to poor quality of the water (Zbyszewski and Torre de Assunção, 1965) which can be quite problematic because many people rely on this water resources not only for irrigation, but also for basic needs. Nowadays, the municipality of Marinha Grande is concerned about the water quality in some of the deep boreholes in view of high concentrations of As, Fe and Mn reported in the monitoring campaigns.

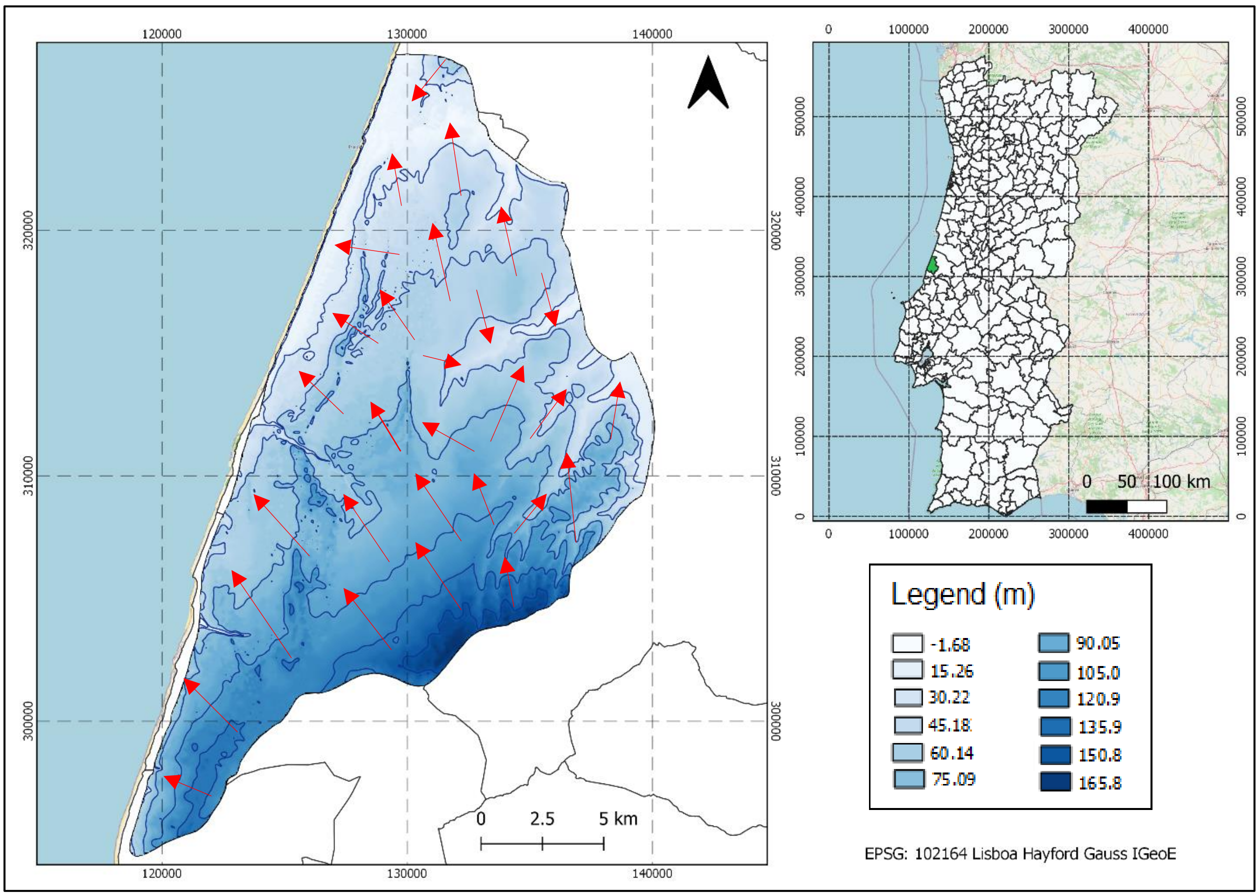


Figure 26 - Piezometric Map of the shallow portion of the Vieira de Leiria-Marinha Grande Aquifer obtained using field data.

The average annual precipitation in the area is 750 mm year-1 and the groundwater recharge is estimated to be about 300 mm year-1 (Almeida et al., 2000) which corresponds to approximately 40% of the precipitation. This high percentage of recharge can be explained by the geological characteristics of soils and the shallow aquifer, that consist of well-sorted high hydraulic conductivity sands that promote the infiltration of the water in the subsoil and therefore, increase the recharge. More than 60% of the area is covered by forest and non-agricultural vegetation.

The regional groundwater flow direction in the aquifer that is mainly from SE-NW, turning into almost E-W in the southern portion of the aquifer (Wallström, 2019). The piezometric map of the shallow portion of the Vieira de Leiria-Marinha Grande is shown in Figure 26, and although the main discharge occurs along the western border of the aquifer to the Atlantic Ocean, local discharge zones mainly related to the main rivers exist in the area and are verified in the field by the presence of coastal and inland springs.

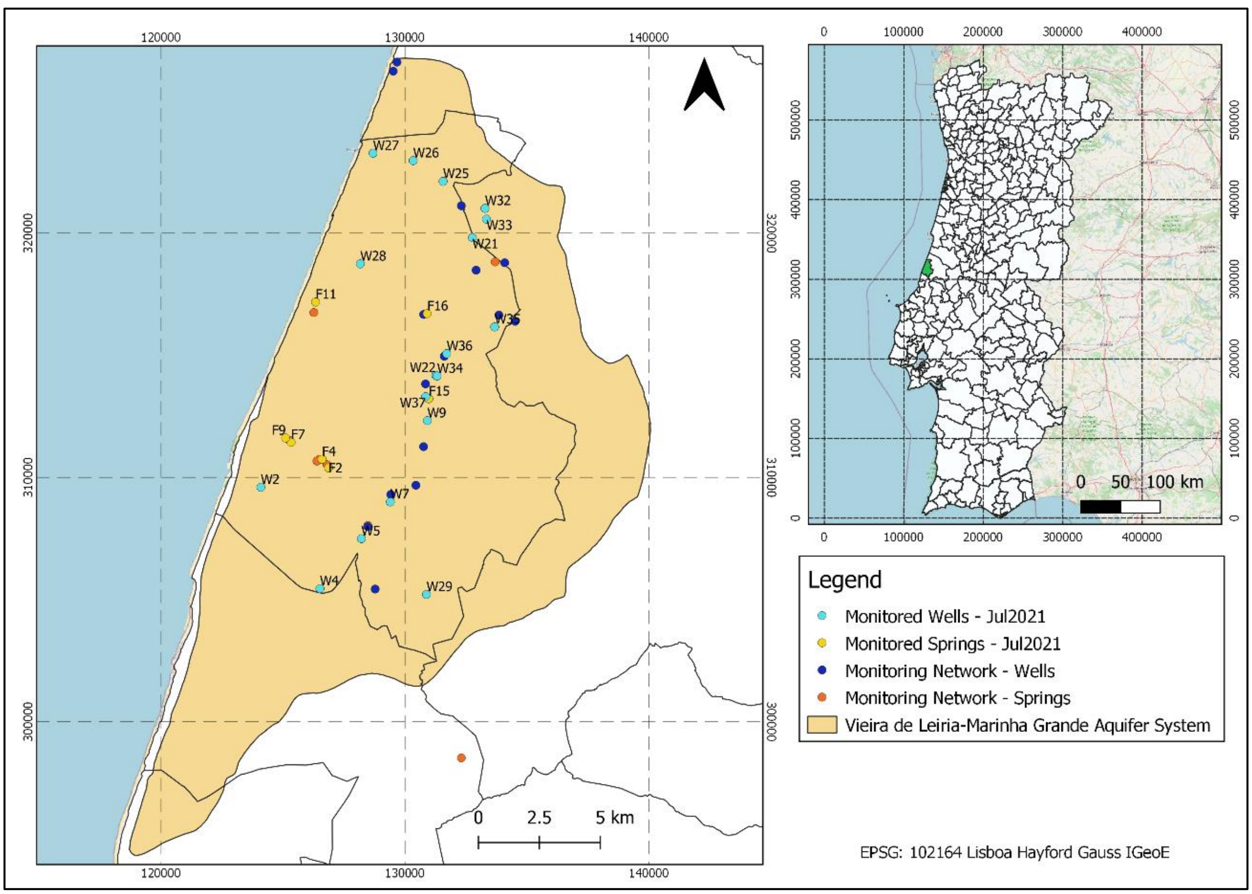


Figure 27 – Water points inventory containing the known wells and springs in the study area as well as the points sampled during field work on 29-30/07/2021.

The total discharge of the aquifer is unknown, although the total groundwater resource available on the system is estimated to 85 hm³ year^{-1} (APA, 2016). Private abstraction of groundwater for domestic, industrial and irrigation is not officially recorded, while groundwater extraction for public supply in 2019 was in the order of 3.7 hm³ year^{-1} according to the data made available by the municipality.

Since there is no monitoring network set up in the shallow portion of the Vieira de Leiria-Marinha Grande aquifer, a database was prepared with all the information available about the location of shallow wells and springs, as well as information provided by ICNF and also the information collected in the field work in order to get monitor the groundwater level in the area (Figure 27).

The presence of diapiric formations favors the development of lowered areas which were filled by the sediments and are responsible for holding aquifers in the area (Dias, 2017). These evaporitic materials in relatively lower depths can influence the composition of the groundwater originating strongly chlorinated or sulfated waters and locally, the occurrence of waters that cannot be used for drinking purposes and domestic uses.

3.8) Groundwater recharge (Chloride Mass Balance method)

Chloride mass balance method was used to estimate long term groundwater recharge to the aquifer. This method is based on the conservative character of the chloride ion and uses it as a chemical tracer to estimate groundwater recharge. Chloride is a conservative ion, so usually is not affected by chemical reactions during infiltration processes, therefore it is reasonable to assume that the atmosphere is the only source of chloride in groundwater (Healy, 2010; Condesso de Melo, 2002). This assumption can only be made because the Vieira de Leiria-Marinha Grande aquifer system is a fresh coastal aquifer with no salinization processes, otherwise, chloride could not be used as a tracer.

In order to estimate groundwater recharge using the chloride mass balance, three parameters must be considered: annual average precipitation in the study area, average chloride concentration in precipitation and dry deposition and, the average concentration of chloride in groundwater in the study area (Equation 1) (Allison & Hughes, 1978).

$$R_{Cl} = \frac{P\left(Cl_{Rainfall} + Cl_{Dry\ Dep.}\right)}{Cl_{Groundwater}} \tag{1}$$

Being, R_{Cl} = Groundwater recharge, P = annual average precipitation (mm/year), $Cl_{Rainfall}$ = average chloride concentration in precipitation (mg/L), Cl_{DryDep}. = average chloride concentration in dry deposition (mg/L) and, $Cl_{Groundwater}$ = average concentration of chloride in groundwater.

The chloride mass balance is a very simple and cheap methodology. The results' precision is directly dependent on the number of samples as well as their temporal and spatial distribution.

This methodology to estimate groundwater recharge has been widely applied by researchers worldwide (e.g. Gebru & Tesfahunegn, 2019; Healy, 2010; Marei *et al.,* 2009; Condesso de Melo, 2002; Bazuhair & Wood, 1996) and according to Custódio e Llamas (1983) presents very good results in sandy zones with small runoff rates.

The samples collected during fieldwork occurred on the 29th July 2021 were analyzed for chloride in the Water Analysis Laboratory of the Instituto Superior Técnico and used to estimate average annual recharge in the aquifer. There was a total of 9 samples, being 8 from springs and 1 rainfall sample (combined sample from Dec/2020, Jan/2021 and Feb/2021) (Figure 28).

Figure 28 - Groundwater samples collected in the study area during the fieldwork from 29-30 July 2021.

Poor conditions during sampling (stagnated water) led to non-representative Cl results and the elimination of two groundwater samples (SF9 and SF16) from the analysis. The median Cl concentration in groundwater was 32.5 mg/L and since there is only one available data of Cl concentration in rainfall (7.9 mg/L), it was extrapolated to all the points. The precipitation used in the recharge calculations was the annual average precipitation in the study area obtained from E-OBS database for the period of 2001-2020 (750 mm/year) and the average annual recharge was approximately 150 mm/year, which corresponds to 25% of the average precipitation.

The application of this methodology in the case is very limited due to the lack of a continuous monitoring of at least five years, the small amount of samples without temporal resolution, and especially the absence of more rainfall samples. However, considering that there are no other references on groundwater recharge for the shallow portion of the Vieira de Leiria-Marinha Grande aquifer system for the study area, the results obtained can be used as an approximation.

The establishment of a monitoring network in the area and continuous evaluation of the chloride concentrations could be an efficient resource to increase accuracy in recharge estimations and provide valuable data to improve groundwater management in the Vieira de Leiria-Marinha Grande Aquifer.

3.9) Conceptual Model of the study area

The hydrogeological conceptual model of the aquifer system was elaborated based on the geological maps, river network and information in the bibliographic references, as well as GPR and groundwater level data obtained during fieldwork (Figure 29).

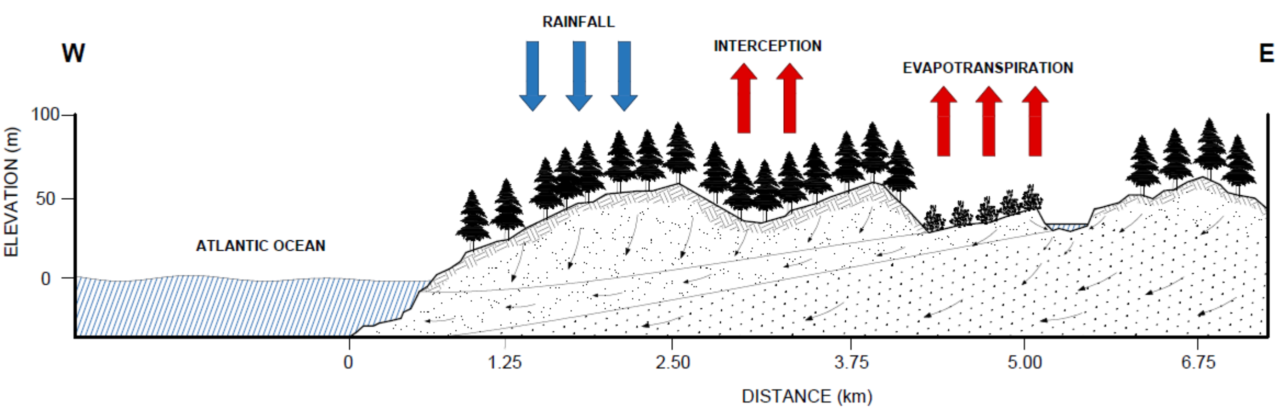


Figure 29 – 2D Hydrogeological Conceptual Model of the study area.

The shallow portion of the Vieira de Leiria-Marinha Grande aquifer consists of beach sands and sand dunes composed by well-sorted very fine to fine sand with high hydraulic conductivity and infiltration rates. Due to the characteristics presented before, the dunes can increase the effective precipitation and consequently, groundwater recharge.

GPR studies conducted suggest the presence of layers with small hydraulic conductivity (possibly clay) that may influence the infiltration rates in some regions of the aquifer.

The main discharge zone is the Atlantic Ocean in the Western boarder of the aquifer but the Lis River and others smaller streams like the São Pedro de Moel Stream act like local discharge zones, influencing the hydrological regime. The water level is normally found two to four meters depth, reaching surface in some areas.

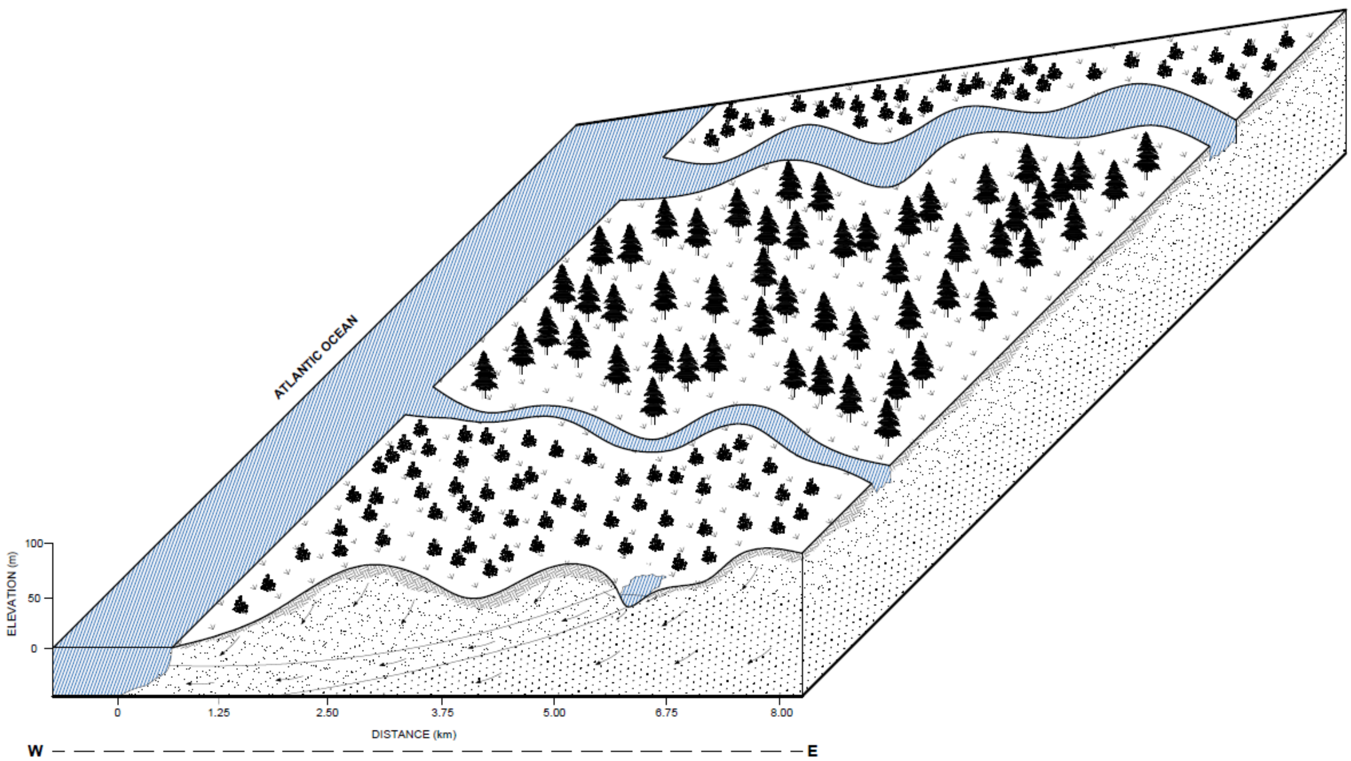


Figure 30 - 3D Hydrogeological Conceptual model developed for the study area.

The 3D hydrogeological conceptual model is presented in Figure 30. Before the forest fire occurred in October 2017, the vegetation in the area was composed essentially by pine trees, with average root depth of about four meters (Appelo & Postma, 2005). After the fire, in the burnt area, the vegetation that was completely removed initially, started to grow, but instead of the original pine trees, it was substituted by bushes and shrubs with an estimated average root depth of about half meter.

This change in vegetation may alter the hydrological regime, not only due to the decrease in evapotranspiration and interception rates, but also due to changes in soil characteristics (soil water repellency, formation of macropores, etc.).

4) Estimation of evapotranspiration based on MODIS products

In this chapter, it is explained the origin of the data used to calculate potential evapotranspiration, as well as the concepts, relationships and limitations involving the estimations conducted.

4.1) Climate Data

The daily observational dataset for the climate variables used in this study was obtained from the EU-FP6 project UERRA and the Copernicus Climate Change Service, and the data providers in the ECA&D project, named E-OBS. It consists on a daily gridded land-only observational dataset available for all the European territory available at 0.10 and 0.25 degree grid and comes as an ensemble for: daily mean temperature (TG), daily minimum temperature (TN), daily maximum temperature (TX), daily precipitation sum (RR), daily average sea level pressure (PP) and global radiation (QQ) (Cornes *et al.,* 2018).

The dataset is constructed through interpolations of meteorological observations sourced from the European National Meteorological and Hydrological Services (NMHSs) or other data holding institutions. In this study, version 23.1 was used for the variables TG, TN, TX and RR with a 0.1-degree grid ensemble for the period of 1995 to 2020 (Table 3).

Table 3 - Descriptions of the main variables taken from E-OBS dataset (Source:https://cds.climate.copernicus.eu)

Name	Code	Unit	Description
Maximum temperature	TX	°C	Daily maximum air temperature measured near the surface, usually at height of 2 meters.
Mean temperature	TG	°C	Daily mean air temperature measured near the surface, usually at height of 2 meters.
Minimum temperature	TN	°C	Daily minimum air temperature measured near the surface, usually at height of 2 meters.
Daily Precipitation	RR	mm	Total daily amount of rain, snow and hail measured as the height of the equivalent liquid water in a square meter The data sources for the precipitation are rain gauge data which do not have a uniform way of defining the 24-hour period over which precipitation measurements are made. Therefore, there is no uniform time period (for instance, 06 UTC previous day to 06 UTC today) which could be attached to the daily precipitation.

In order to avoid uncertainties that may arise from the interpolation and homogenization techniques from E-OBS (Hamouda & Pasquero, 2021) a correction factor was calculated for the precipitation data, using observational data from eight meteorological stations from the Portuguese System of Water Resources (SNRH) close to the study area (Figure 31).

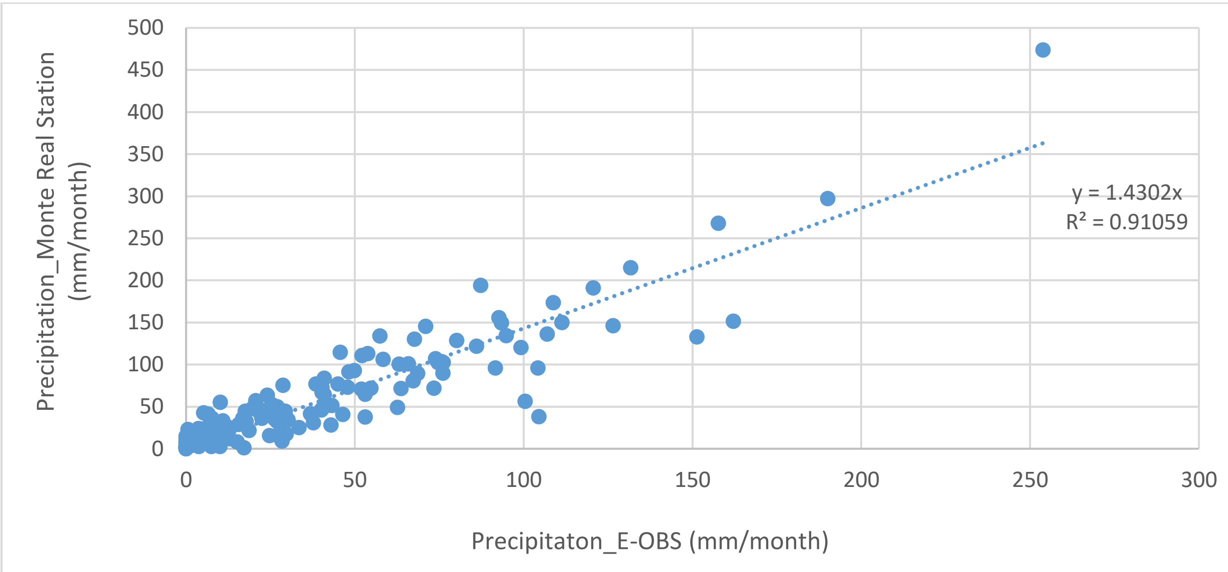


Figure 31 - Linear regression of the correction factor of the precipitation values obtained on E-OBS using data from SNRH.

The data from SNRH was not used directly in the study due to the absence of a full time series for the needed parameters in the proposed temporal interval. The corrected values are about to 50% higher than the ones obtained directly from E-OBS (Figure 32), this is probably related to the fact that E-OBS considers points outside the study area in its interpolations.

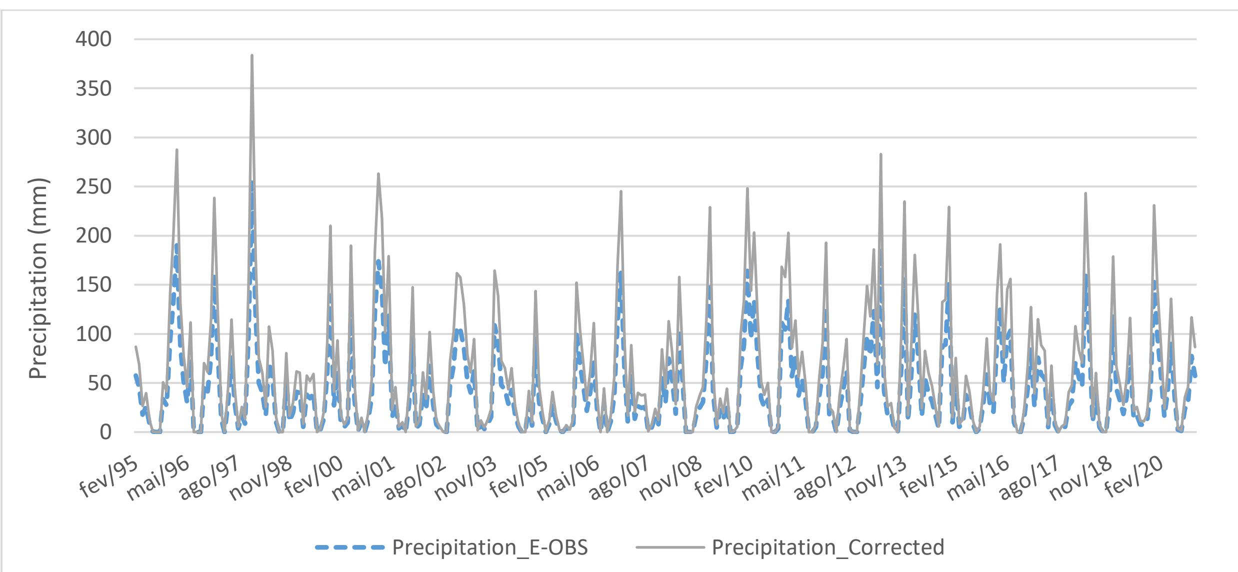


Figure 32 - Precipitation data from E-OBS corrected using data from eight meteorological stations from SNRH.

4.2) Moderate Resolution Imaging Spectroradiometer sensor (MODIS)

The use of remote sensing data, named satellite data, to estimate environmental variables has been widely applied in the last decades due to the increase in the quality of the data, as well as in the temporal and spatial resolutions these datasets allow (Jiang *et al.,* 2015; Zhang *et al.,* 2018; Lacouture *et al.,* 2020).

In the present study, the Moderate Resolution Imaging Spectroradiometer sensor (MODIS) was used in order to obtain the Normalized Difference Vegetation Index (NDVI) and the Real (ET) and Potential Evapotranspiration (PET).

MODIS data was obtained from the Application for Extracting and Exploring Analysis Ready Samples (AρρEEARS) for the National Aeronautics and Space Administration (NASA) according to the defined location. The final datasets comprehend NDVI, ET and PET data from 2001-2020.

4.3) Normalized Difference Vegetation Index (NDVI)

The Normalized Difference Vegetation Index (NDVI) is a can be defined as a measure of surface reflectance that can be used to perform analysis of vegetation using remote sensing data (Nunes *et al.,* 2016; Carlson and Ripley, 1997). It has been widely applied to monitor vegetation density, heath, and growth, as well as to monitor droughts and predict wildfires risk zones.

Plants use solar radiation as energy source during photosynthesis. According to the spectral range absorbed by a surface, it is possible to distinguish clouds, snow, and different types of vegetation cover. For example, the chlorophyll present in the plant leaves absorbs visible light and reflects the near-infrared, so if the reflected radiation in the infrared is much higher than in the visible wavelengths, we probably have a vegetation with many leaves, likely a forest (van der Slik, 2014, Sellers, 1985).

The NDVI is calculated according to Equation 2, meaning that the vegetation density in a pixel of the image will correspond to the difference between the intensity of the light reflected in the infrared and red portions of the electromagnetic spectrum divided by the sum of the same bands.

$$NDVI = \frac{IR-RED}{IR+RED} \quad (2)$$

This index returns values from -1,0 to 1,0 that represents different kinds of land cover. For instance, negative values normally correspond to clouds, water and snow cover, values close to zero are related to rock and bare soil. Values between 0.1 and 0.4 represents scrubs, bushes and grassland while high values between 0.6 and 1 indicate a denser vegetation such as tropical and temperate forest vegetation (Weier and Herring, 2013).

The NDVI index from the MOD13Q1.006 (Didan, 2015) products were used from the Terra satellite with spatial resolution of 250 meters and an interval of 16 days. A total of 922 images were acquired, being 461 for the area burnt in the fire in October 2017 and the same number for the unburnt areas of the Leiria Pine Forest. The monthly values obtained for the burnt and unburnt areas varied between 0.753 – 0.299, and 0.725 – 0.471, respectively.

Figure 33 shows the monthly average of the NDVI index for the burnt and unburnt areas in the Leiria Pine Forest.

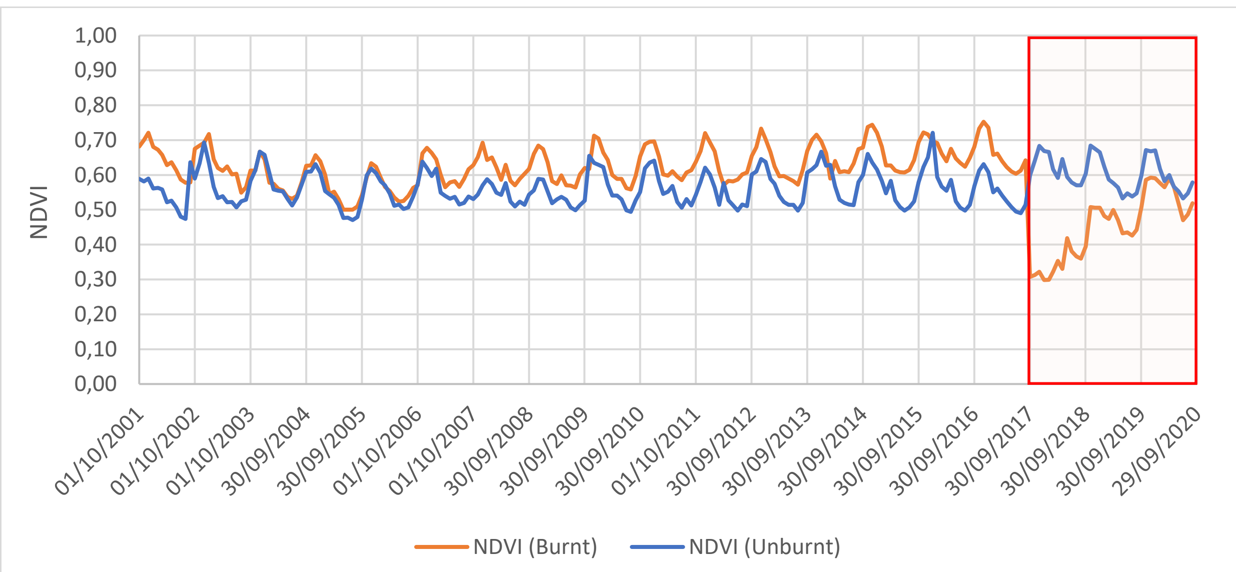


Figure 33 - NDVI indices for the burnt and unburnt areas in the Leiria Pine Forest from 2001 -2020. The red rectangle marks the period after the fire.

The NDVI index for the area burnt on the fire of October 2017 has been bigger than the index for the unburnt area over all the time series. It could be related to the fact that although the unburnt area corresponds to approximately twice the size of the burnt one, the land cover in the burnt area has always been essentially the pine forest, while in the unburnt area, the land cover distribution is much more diverse, including cities, factories, agricultural areas, pastures and scrubs, that will likely reduce the overall average of the NDVI index (Figure 34).

The abrupt decrease observed in the NDVI series of the burnt area (Figure 33) reflects the change in vegetation just after the fire, while in the unburnt area, the values keep the trend observed in the previous years. This sharp decrease is related to the loss of biomass and the reduction of leaves (reduction in chlorophyll absorption).

Besides the sharp decrease related to the fire in October 2017, we can also observe a decrease, especially in the unburnt area between August 2003 and October 2006. This could be related with the fire occurred in August 2003. In this case, the decreased in the unburnt area instead of in the burnt one can be related to the fact that in this work, burnt and unburnt areas were defined based on the fire of 2017, and they probably don't match completely with the area burnt in 2003. Besides, since NDVI is impacted by climatic variability, it could also be associated with the severe drought occurred in 2003-2005.

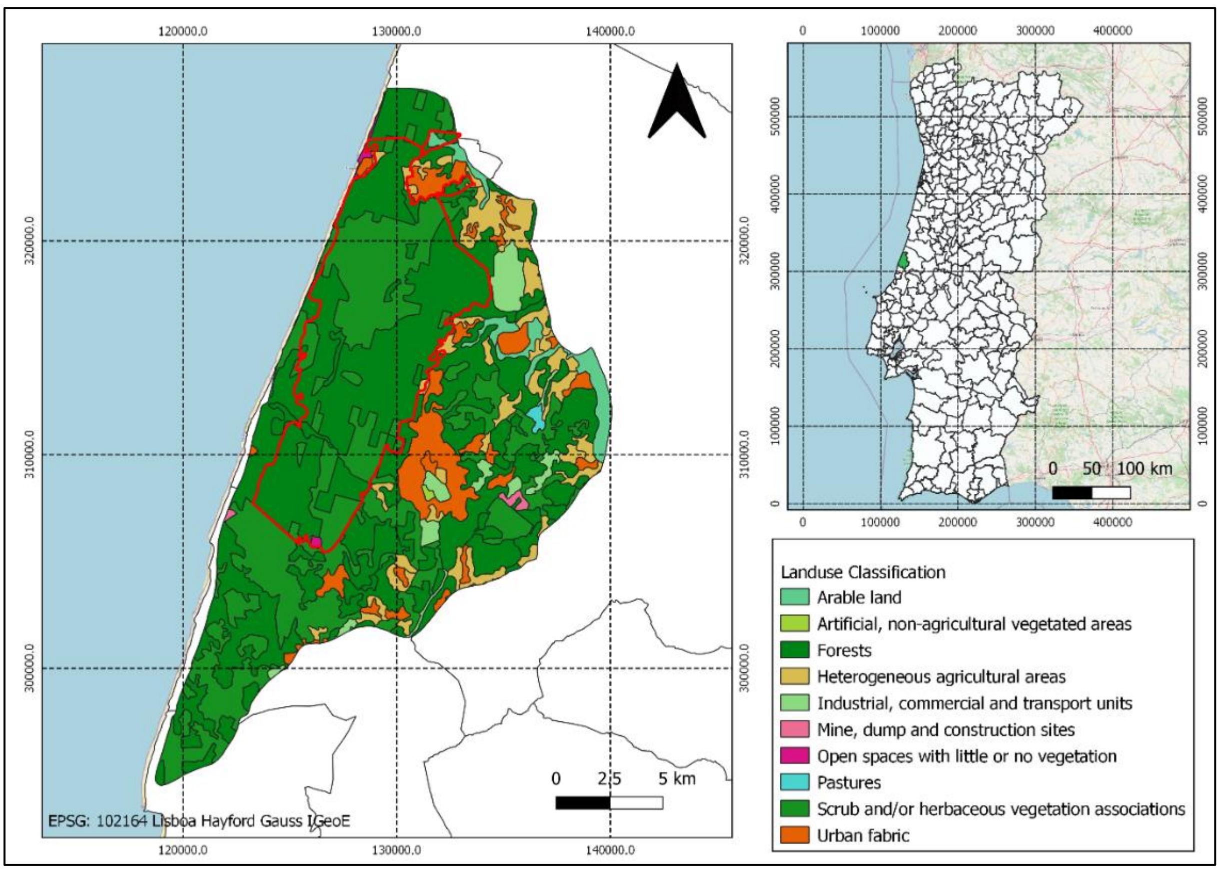


Figure 34 - Land Use map of the study area with the area burnt in October 2017 limited by the red line.

The boxplot in Figure 35 shows the distribution of the NDVI values before and after the fire in the burnt area. The maximum values observed in the burnt area before the fire were around 0.85 with an average of about 0.62. Looking into the data from after the fire, we can see that the minimum values observed continue in the same range, but the maximum value was reduced to about 0.60, which also causes a reduction in the average NDVI for the area. This is also an effect related to the reduction of the vegetation density in the area. The values from the 'Burnt_PosFire' series consist in the data from the 3 years after the fire, and as we can observe in Figure 36, these values are increasing over time, probably in the future, reaching the pre-fire observed values.

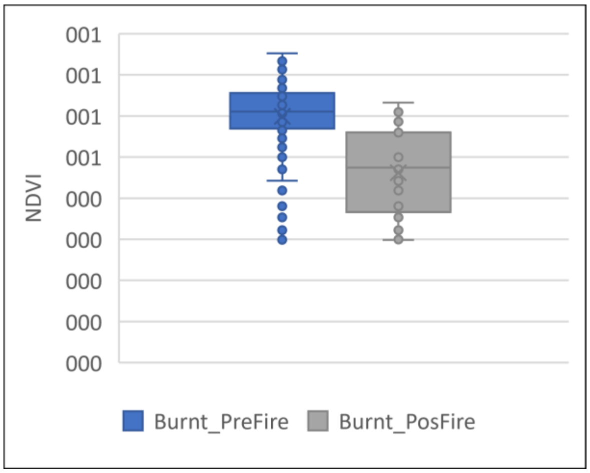


Figure 35 - NDVI distribution before and after the fire.

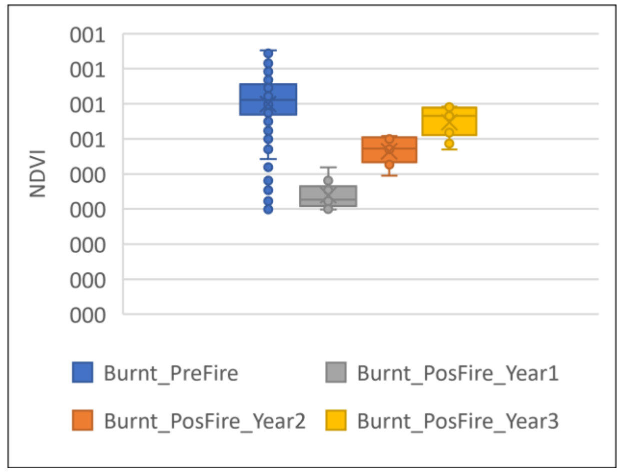


Figure 36 - NDVI Distribution in the period before the fire and yearly after the fire.

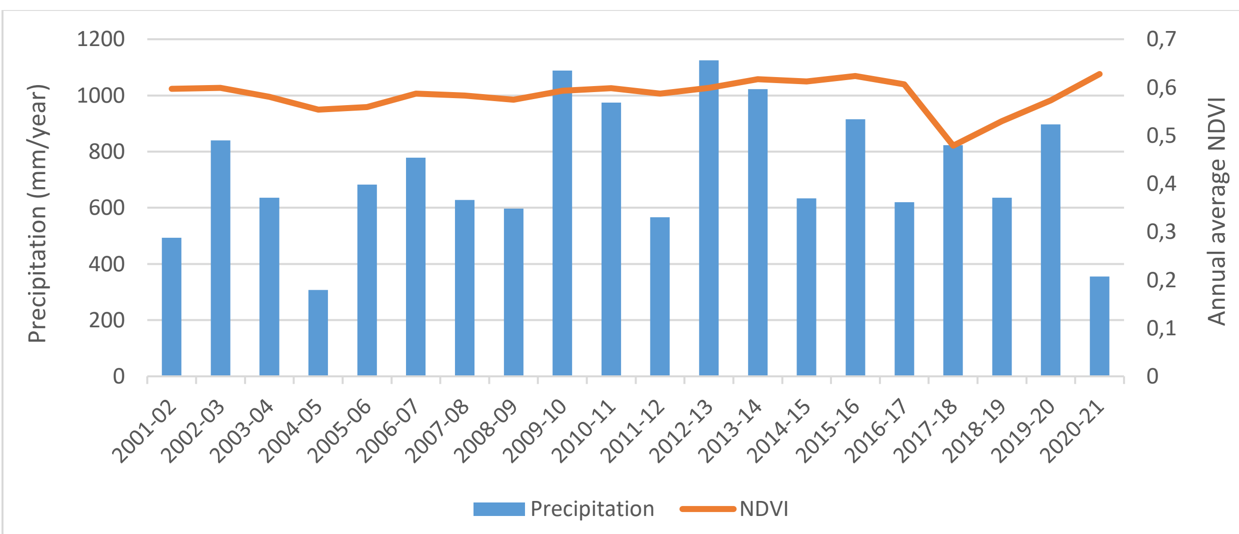


Figure 37 - Annual Precipitation x annual average NDVI in the Leiria Pine Forest for the period between 2001-2020.

Annual precipitation data (hydrological years) was calculated using the data extracted from the E-OBS database for the period of 2001-2020 and corrected using data from Portuguese system of Water resources (SNRH) was compared to the annual average NDVI for the same period for the whole (Figure 37).

The results of an annual analysis show that NDVI does not vary much with precipitation in the study area in an annual time scale. The sharp decrease in the NDVI can be seen in the 2017-18 hydrological year and is related to the wildfire and consequent elimination of approximately 86% of the vegetation. In addition, even though in a smaller scale, another decrease in NDVI can be observed between 2003 and 2006 and although the severe drought experienced in the area during this period may have contributed to it, most likely the decrease in NDVI is related to another forest fire occurred in August 2003 in the region. It suggests that climate variations do not affect directly NDVI trends, and although they may influence the fire risk, NDVI changes are mostly related to meaningful land cover changes.

In order to determine the recovery time of the vegetation in the study area, the annual average NDVI index for the period before the fire (jan-2001 to sep-2017) in the burnt area was used as a baseline and compared to the annual average NDVI indices after the fire. Linear regression was applied to determine the recovery time of 3.7 years.

It is important to understand that the recovery of the NDVI index values do not mean that the area and the vegetation is returning to what it was before the fire. The Leiria Pine Forest was a 11ha forest of about 750 years consisted of mainly Pinheiro-bravo tress, planted to slow down the progress of the dune system in the area. The Pinheiro-bravo species is considered to have a slow growth compared to eucalyptus and take about 8 years to flourish for the first time. These factors indicate that it would be quite difficult to recover the forest in such a small amount of time, even if the reforestation program started just after the fire, which did not happen.

So, the increase in the NDVI in this case, cannot be used to determine the recovery time, since it is probably related to the growth of other vegetation species, meaning that although there is vegetation growing in the area, it does not mean that the Leiria Pine Forest is coming back to its pre-fire conditions (Figure 38).

Figure 38 – Differences in land use in the Leiria Pine Forest area before (a) and after (b) the 2017 fire taken in July 2021.

In fact, the development of invasive species slows down the recovery of the Pine Forest, because this species starts to compete with the pine trees, hampering their natural development. Besides the environmental and biodiversity impacts of these vegetation cover changes, the historical, cultural, and social impacts are also to be considered since the forest had an important role in the community routine for decades as a source of wood and resin, for example.

So, in order to get a better understanding about the recovery time of the forest, other indicators may be applied such as the Normalized Burn Ratio (NBR) and Integrated Forest Index (IFI) (Bright *et al.,* 2019; Shvetsov *et al.,* 2019; Chen *et al.,* 2011), for example.

4.4) Potential Evapotranspiration (PET) and Evapotranspiration (ET)

The definition of Potential Evapotranspiration (PET) according to Rosenberg (1974) is "the evaporation from an extended surface of [a] short green crop which fully shades the ground, exerts little or negligible resistance to the flow of water, and is always well supplied with water" (Kirkham, 2014).

Evapotranspiration is the combined loss of water by transpiration, release of water through plants leaves, and the evaporation from the soil surface, in a given area and during a specific period of time (Porkony, 2019; Kirkham, 2014).

In most cases, PET and ET are different because PET is established under specific (ideal) conditions while ET is the value obtained in the real environment considering all the interactions and processes occurring and all its complexity.

Several methods can be applied to determine PET, from the simpler ones, purely empiric, to the more complex ones. These methods can be direct like lysimeters or indirect such as the methodologies developed by Thornthwaite (1948), Blaney-Criddle (Doorenbos e Pruitt, 1977), Hargreaves (Hargreaves, 1994) e de Penman-Monteith FAO (PM-FAO) (Allen *et al.,*1998).

This variety of methods arises from the high complexity of the processes involved in the water movement among the environment's compartments, the differences between climate variables over the world and the difficulty in acquire reliable data to apply the methods (Carvalho *et al.,* 2011).

Remote sensing data from MODIS has been applied in several regions to estimate ET presenting good correlations between NDVI and ET (Cherif *et al.,* 2013; Nagler *et al.,* 2005).

ET and PET values were taken from the MODIS database (MOD16A2GF.006 Terra Net Evapotranspiration) with an interval of 8 days and a resolution of 500m for the period of 2001 - 2020 for the burnt and unburnt areas in Leiria Pine Forest (Running, 2017). A total of 3680 images were acquired being 1820 (ET and PET) for the area burnt in the fire in October 2017 and the same number for the unburnt area.

The monthly average values obtained for the burnt and unburnt areas varied between 21.37 – 101.19, and 22.58 – 98.2, respectively. The lower values are observed in winter between December-January and the higher ones in the summer from Jun-July. The time series for the burnt and unburnt areas of the Leiria Pine Forest can be seen at Figure 39.

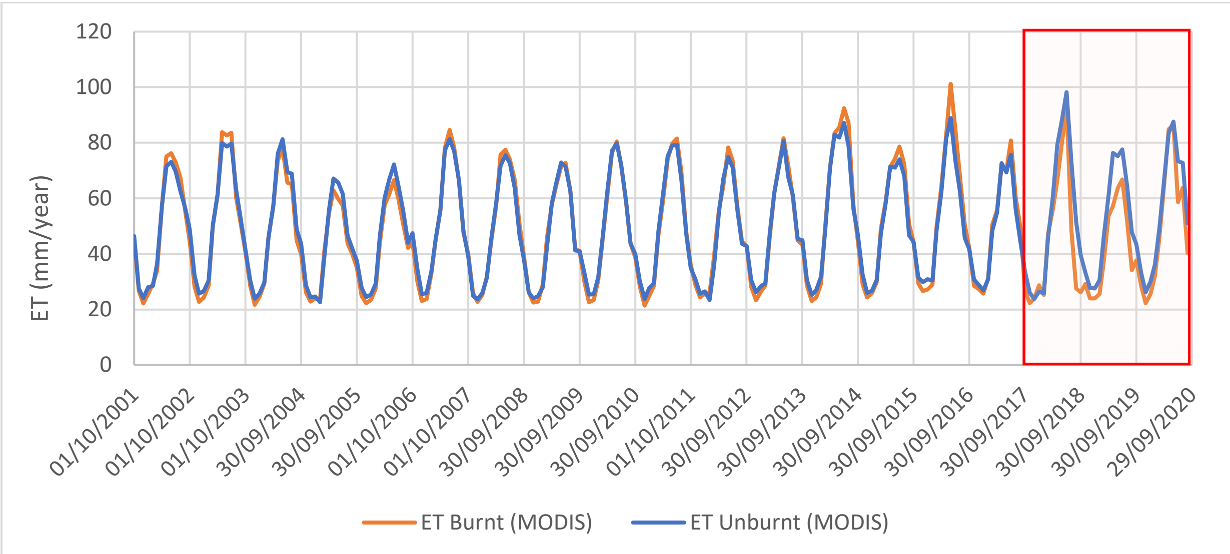


Figure 39 - MODIS Evapotranspiration data for the Burnt and Unburnt areas in the Leiria Pine Forest.

According to the data obtained from MODIS, the time series for evapotranspiration in the burnt and unburnt areas have a high correlation in the analyzed period (r^2= 0.953). This correlation is a little bit higher if we consider the period before the fire (r^2= 0.984) and decreases for the period after the fire (r^2= 0.912) these variations are expected, since before the fire, the vegetation was more homogeneous than after, which changes in the evapotranspiration rates. Although, it is clear that even after the fire, the ET in the area keeps a relatively high correlation.

The relation between the ET in the burnt and unburnt areas was used to estimate the expected ET values for the area if the fire did not happen. Since ET is highly variable from year to year, this analysis allows us to compare the ET values expected if the fire did not happen with the observed (real) values.

According to the data of MODIS satellite, the occurrence of the fire reduced the ET in 14% for the first year, 19% in the second and 10% in the third year after the fire when comparing the data from burnt and unburnt areas. So, by comparing the ET values from before and after the fire, we can see that the values after the fire are lower than the ones before, which was expected if we considered similar meteorological conditions, due to the reduction of the amount of plants in the area and consequently, the decrease in the overall transpiration by them (Figure 40). The data from MODIS satellite was not used in groundwater recharge simulations due to its poor representation of the reality in the Leiria region, it will be discussed in the next chapters.

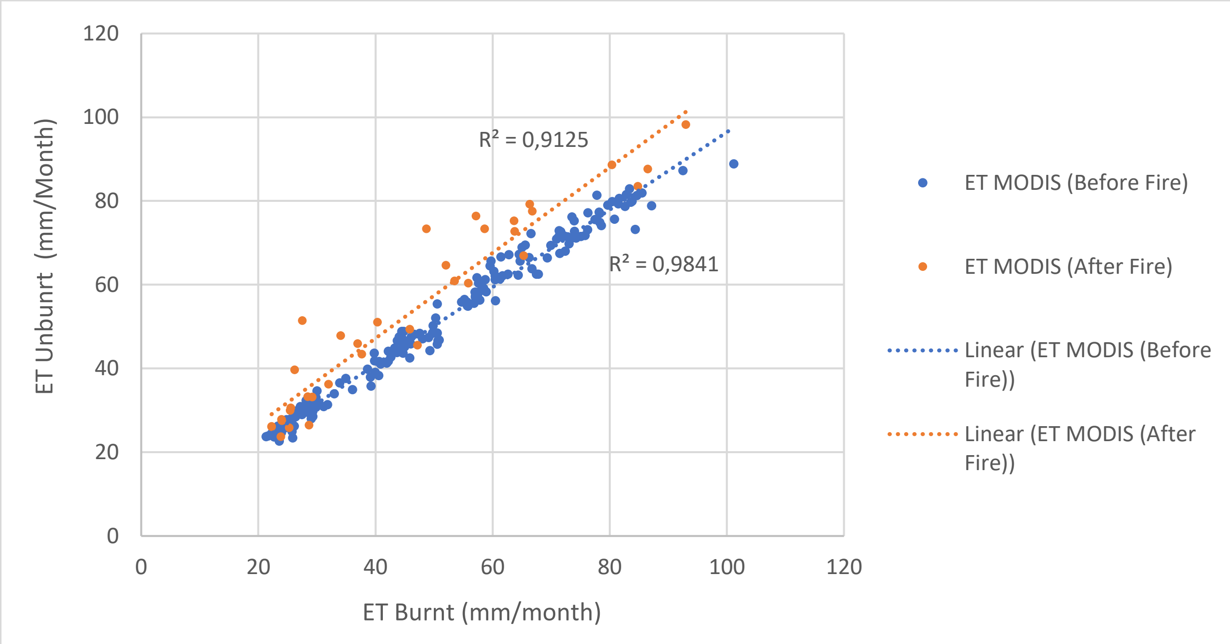


Figure 40 - ET trends from before and after the fire for the burnt and unburnt areas in the Leiria Pine Forest.

Regarding PET, the values range from approximately 30 to 250 mm/mm along the time series and the correlation between the data observed in the bunt and unburnt areas is about 0.998. Since the PET is a parameter that consider the ideal conditions and a specific (constant) type of vegetation for both cases, it is expected that the values do not change much over time even after the fire.

4.5) Crop Coefficient

The crop coefficient values represent the combination of effects plants may suffer due to changes in several parameters such as crop characteristics, plant height, rate of development, leaf area index, planting date, degree of canopy cover, canopy resistance, soil and climate conditions and management practices (Porkony, 2019).

These values are normally calculated experimentally, but since the use of vegetation indices like NDVI obtained from satellite can help estimate parameters related to vegetation phenology, they can also be used as a tool to monitor the Kc variations in time and space (Duchemin *et al.,* 2006).

The estimation of Kc from the NDVI composites taken from MODIS was calculated according to approach developed by Duchemin et al. (2006), summarized in Equation (3) and applied in the literature for several crop species (Nunes *et al.,* 2016; van der Slik, 2014; Ferrara *et al.,* 2010; Campos *et al.,* 2010; Vuolo *et al.,* 2008).

$$Kc = a.\left(NDVI - NDVI_{\min\_Kc}\right) \tag{3}$$

According to the proposed methodology, the relation between NDVI and Kc must be obtained using the months where the crop transpiration occurs under low stress conditions, to do so, a filtering of the extremes (very dry and very wet months) was done in the NDVI dataset. The filtering was done considering that the Kc values would be only used if the rainfall and ET deficit were both lower than the thresholds established based on the maximization of the correlation between NDVI and Kc (Rainfall < 22.93 and ET Deficit < 229.02). After the filtering procedure, the correlation between the NDVI and Kc increased from 0.47 to 0.60 for the time series (Figure 41).

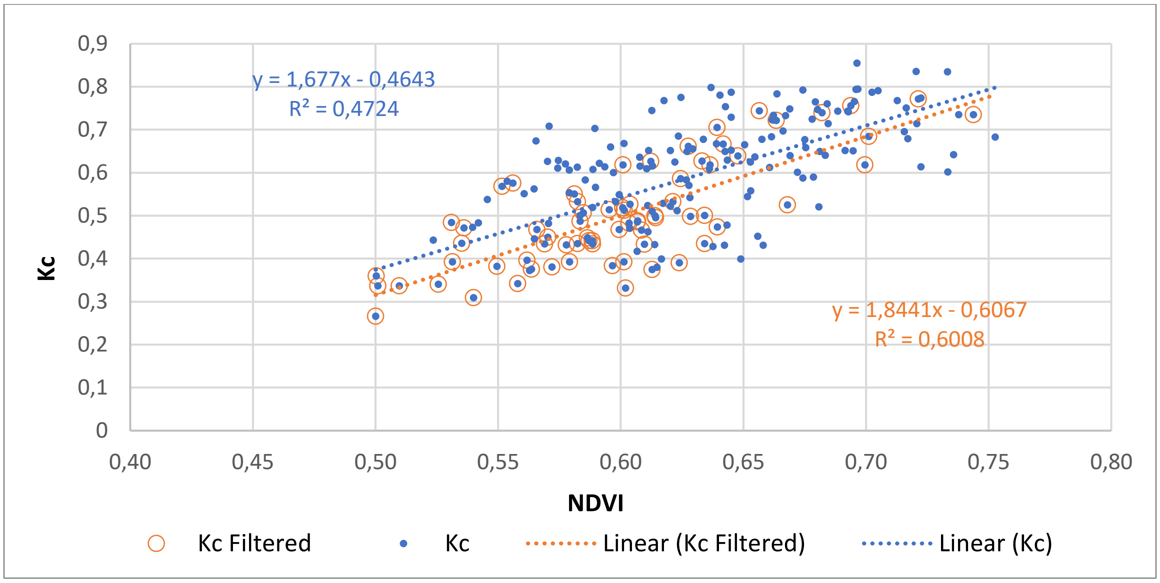


Figure 41 - Correlation between NDVI and Kc before and after filtering.

The FAO Irrigation and Drainage Paper nº56 (Allen *et al.,* 1998) stablishes that the Kc values should not be lower than 0.3 which is the value that represents a bare soil with occasional precipitation. Therefore, for pine forests with NDVI below 0.5 the Kc will remain constant at 0.3.

By plotting the NDVI against the Kc values and using linear regression, an equation relating these parameters will be adjusted and applied to the NDVI series from before the fire (2001-2017) taken from MODIS to generate a temporal series of Kc for the Leiria Pine Forest (Figure 42).

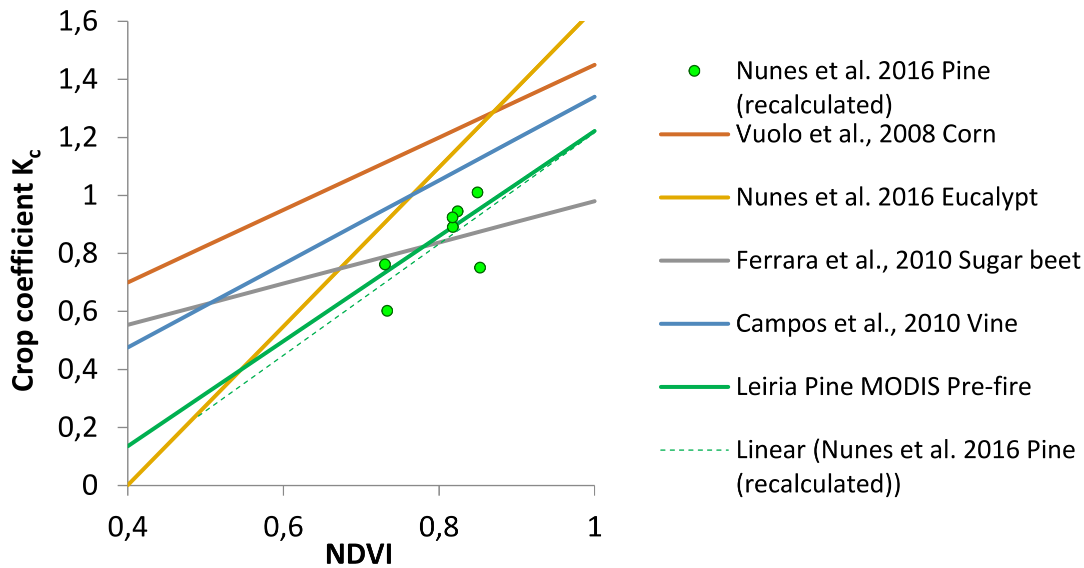


Figure 42 - Relation between crop coefficient and NDVI indices taken from the literature for several crop species.

The correlation obtained is shown in Equation 4 and it was very similar to the ones obtained in other studies for the crop coefficient of pine trees (Nunes *et al.,* 2016; van der Slik, 2014). The parametrization of the Kc is a limitation of the research, because although the obtained relation is consistent for the data, several factors including soil and canopy properties, climatic conditions and crop characteristics might affect the outcome of the calculations.

$$Kc = 1.8109.(NDVI - 0.3251) \quad (4)$$

The Figure 43 shows the temporal evolution of the crop coefficient calculated using the NDVI from the MODIS satellite. The values before the fire range from 0.32 to 0.77 along the times series and the lowest ones are usually observed between July and August while the higher ones between December and January which follows the NDVI, and precipitation patterns discussed in Chapter 4.2. According to Allen et al. (1998), there is a strong relation between the Kc and the soil humidity, that acts like a limiting factor for evapotranspiration, which explains why Kc decreases in the drier months (July-August) and increase in wet ones (December-January).

The same methodology was applied to stablish the relation between NDVI and the Kc after the fire occurred in October 2017. The results are shown in Figure 44.

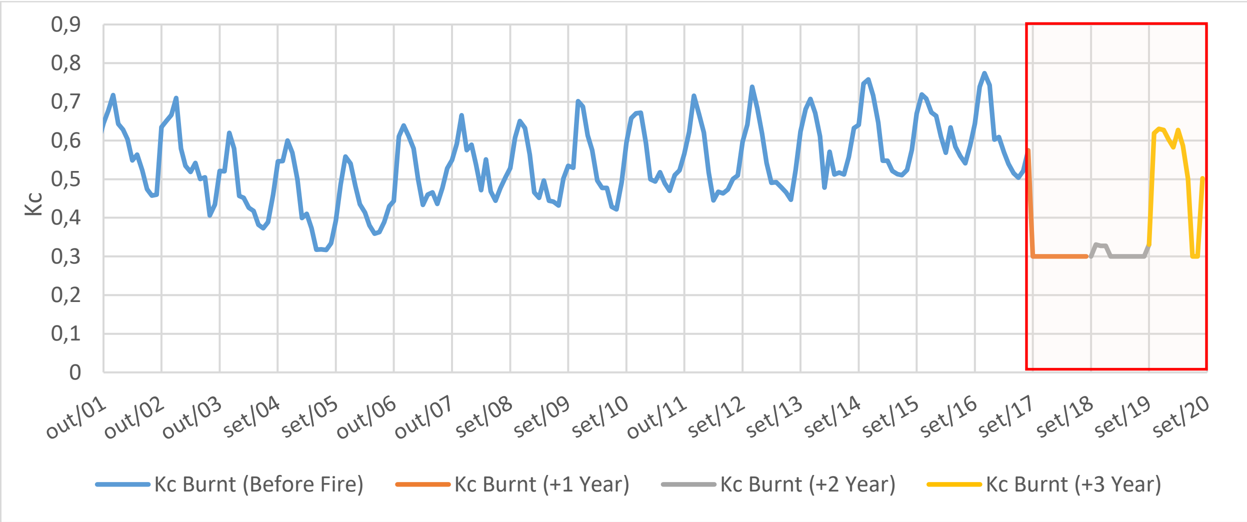


Figure 43 - Crop Coefficient (Kc) for the period from 2001-2020 calculated using the NDVI index from MODIS.

Although the correlations before the fire (0.60) and two years after the fire (0.64) are significant, in the first year after the fire, the NDVI vs. Kc relation do not exist (Figure 44). This is probably related to the death of the vegetation. In this case, when the vegetation is inactive, the evapotranspiration is controlled mainly by the evaporation in the soil, which means that the only factor controlling the evaporation of the water will be the soil humidity. A correlation with r^2=0.43 was obtained between the Kc 1 year after the fire and the precipitation in the area, which is very low to stablish a valid equation, so using the values proposed in Allen et al. (1998), crop coefficients of 0.6 and 0.3 were assumed in the wet and dry months, respectively.

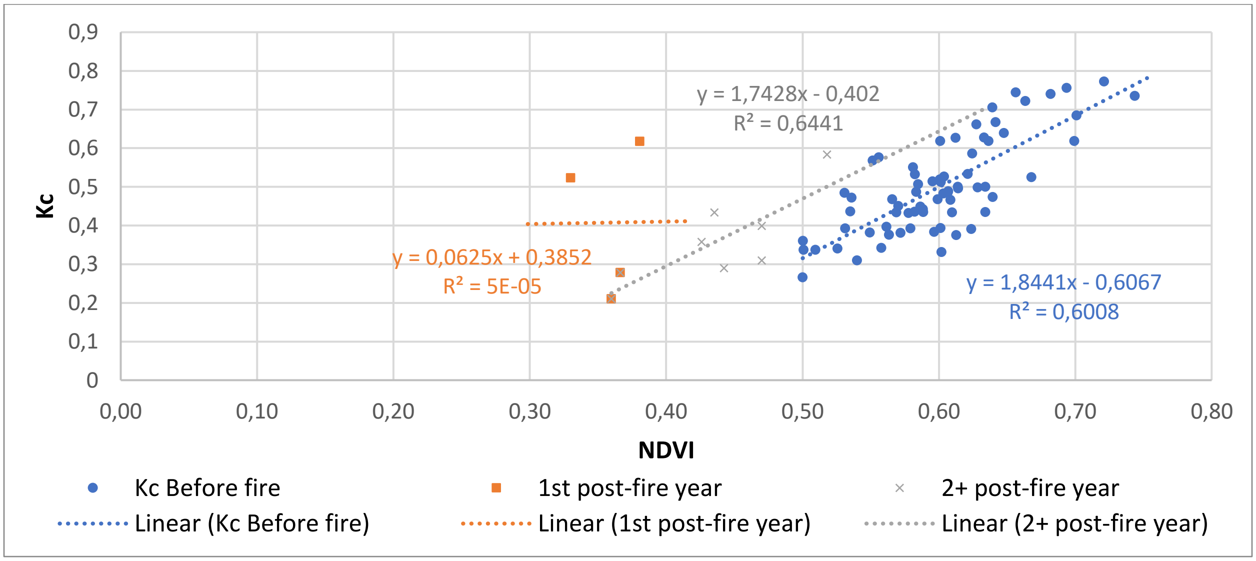
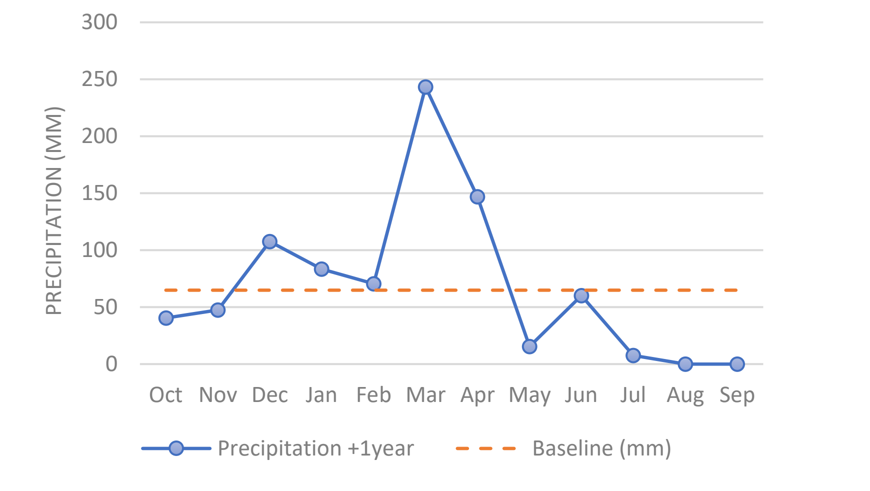


Figure 44 - Correlation between NDVI and Kc for the years after the fire.

In order to stablish Kc values for the first year after the fire using the methodology proposed by Allen et al. (1998), the average annual precipitation for the period of 1995-2016 was used as a baseline and the monthly values of oct-2017 to sep-2018 were compared to it. Values above the baseline (64.88 mm) were considered wet months and the values below, the dry ones (Figure 45).



Baseline	64.88	
Oct	40.47	Dry
Nov	47.59	Dry
Dec	107.70	Wet
Jan	83.37	Wet
Feb	70.51	Wet
Mar	243.29	Wet
Apr	147.03	Wet
May	15.57	Dry
Jun	60.20	Wet
Jul	7.63	Dry
Aug	0	Dry
Sep	0	Dry

Figure 45 - Analysis of precipitation patterns in the first year after fire.

After two years of the fire, a new correlation (r^2=0.64) can be observed between NDVI and Kc in the Leiria Pine Forest area (Figure 44). This is probably related to the progressive growth of vegetation in the burnt area, which makes the NDVI start increasing slowly as new species start to develop in the area.

Although the Portuguese Institute of Forest Conversation estimates a reforestation of 2400 ha by 2022, nowadays, the vegetation in many parts of the area is essentially composed by bushes and scrubs from invaders species. The relation 2 years after the fire is given by Equation 5 and it is similar to the one described by Campos et al. (2010) for vines, which corroborates the hypothesis of the predominance of a smaller and more scrubby type of vegetation.

$$Kc = 1.7428.(NDVI - 0.2607) \tag{5}$$

The relations before and 2+ years after the fire are presented in Figure 46. It can be observed that the trends from before and 2+ years after the fire are parallels to each other, but the curve from before the fire has a lower crop coefficient for the same NDVI values and, consequently a lower water use than the one from after the fire.

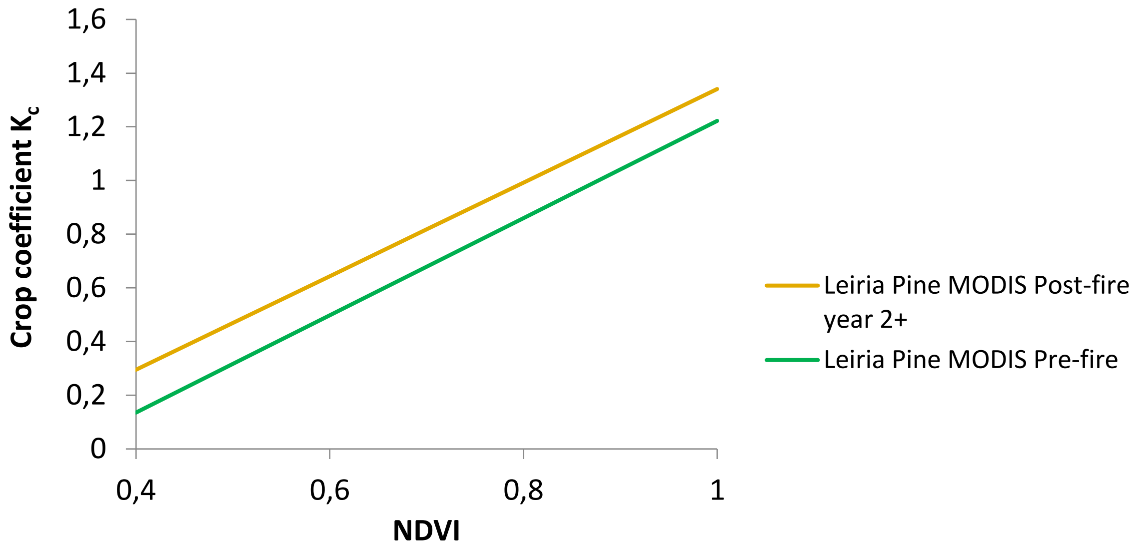


Figure 46 - Relation between crop coefficient and NDVI index before and 2+ years after the fire in the Leiria Pine Forest.

Considering the increasing investments, the advances in the reforestation program and the goals to be achieved until 2030, it is expected that the NDVI keeps getting closer to the values pre-fire due to the increasing development of native vegetation, which would also approximate the NDVI vs Kc relation to the one from before the fire (Equation 3).

4.6) Reference Evapotranspiration (ET0) – Hargreaves Method

Precise estimations of evapotranspiration are essential in the evaluation of water resources availability and crucial in recharge estimation procedures. ET may be affected by several parameters including climatic factors (solar radiation, temperature, wind velocity), crop parameters and soil properties (Allen *et al.,* 1998). A broad variety of methods have been used to calculate ET, considering not only their precision, but also the availability of data for application.

After the publication of the FAO 56 Report, the Penman-Monteith method was defined as the standard methodology to estimate the reference evapotranspiration (ET_0), although it requires climatic data that are not often available for many regions. So based on the available data, the ET_0 was estimated using the Hargreaves Method (HS) (Hargreaves & Samani, 1985), which is a semi empirical method that uses extraterrestrial radiation and temperature data to estimate daily ET_0 according to Equation 6.

$$PET_{HS} = C_H . 0.408 R_0 . (T + 17.8)\sqrt{T_{max} - T_{min}} \tag{6}$$

Where, C_H is the Hargreaves coefficient, the values of 0.408 corresponds to the inverse of the latent heat flux of vaporization at 20◦C, R_0 is the extraterrestrial radiation in mm.day-1 evaporation equivalent and T is the daily mean temperature (◦C) and T_{max} and T_{min} are the daily maximum and minimum temperature values, respectively. The climatic data for the period of 1995 to 2020 was taken from the E-OBS database for the study area and applied to the HS equation. The results are shown in Figure 47.

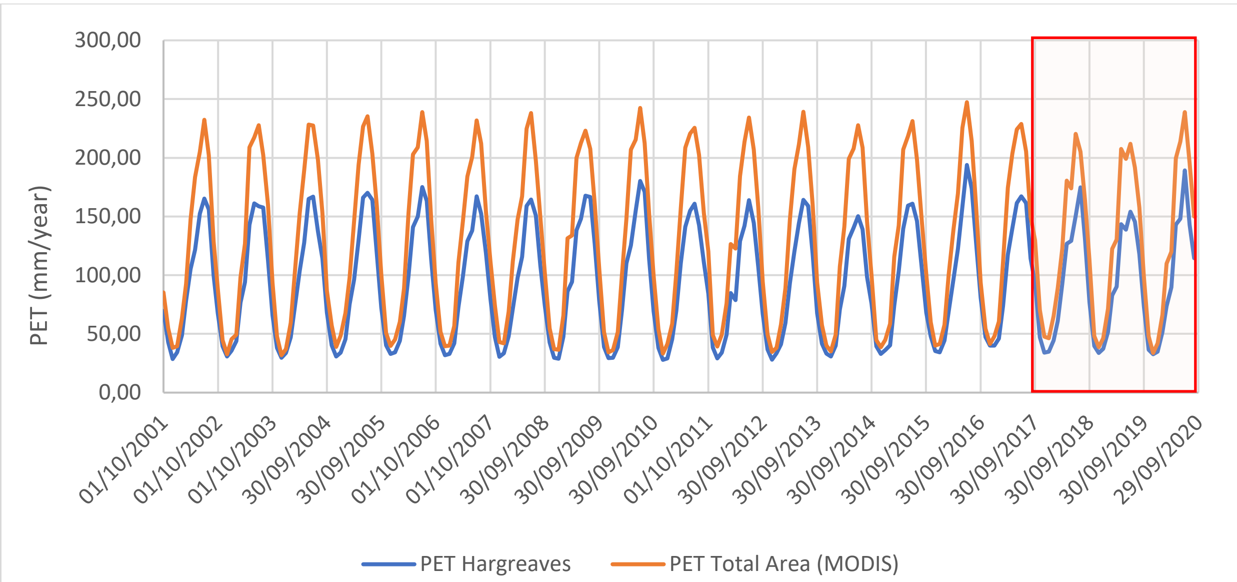


Figure 47 – Times series of Potential Evapotranspiration estimated using the Hargreaves Method (blue) and the Satellite data obtained from MODIS (orange) for the period of 2001-2020 in the Leiria Pine Forest.

The results in Figure 47 show that although correlation between the time series is high (r^2=0.980) the data from MODIS shows values up to 40% higher than the ones estimated by the methodology proposed by Hargreaves (PET-HS). We can observe that the values are very similar for both methods in the winter months and as the temperature gets higher, the values for the satellite get higher than the ones obtained by the Hargreaves Method.

In order to double check the accuracy of the PET values obtained by the Hargreaves method and their representativeness of the study area, PET values were estimated using the Penman-Monteith (PM-FAO) method using the data from the Monte Real Meteorological station for the period of January/2002 to December/2009. The location of the meteorological station and the period analyzed was chosen based on the available data. The results are shown in Figure 48.

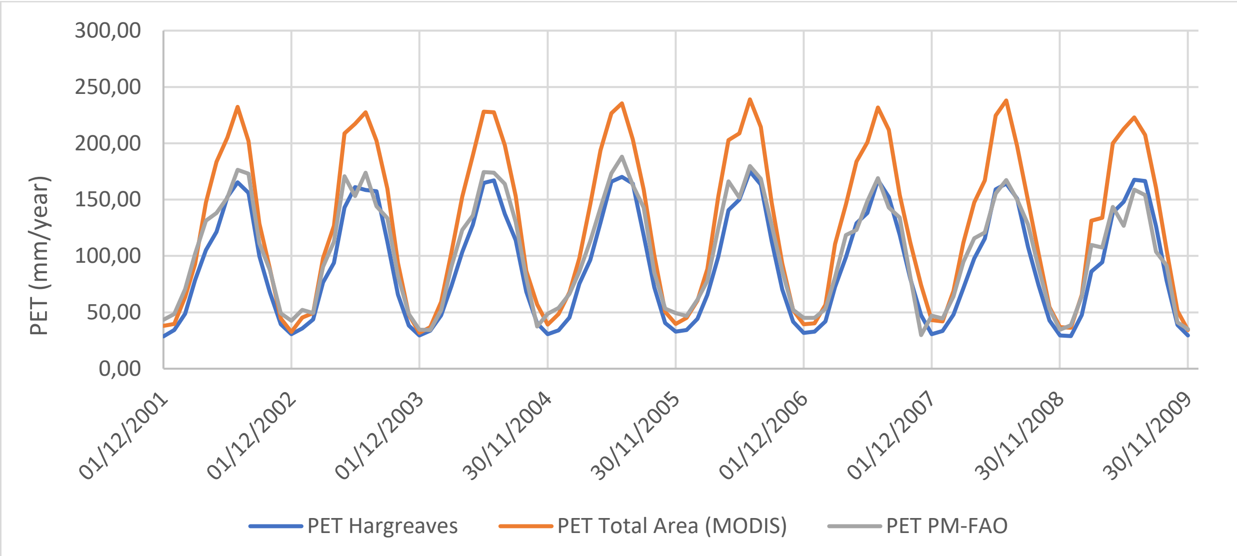


Figure 48 – Comparison of PET values estimated using different methods for the study area.

The results obtained using the HS and PM-FAO methods present very similar values for the analyzed period with a high correlation (r^2 =0.955) between them, and although the PET values from MODIS are higher than the PET-HS and PET-PM-FAO they also present good correlations. Therefore, the satellite is most likely systematically overestimating the PET values in the summer months decreasing the reliability of the direct use of satellite data in the simulations.

The MOD16 algorithm is based on the Penman-Monteith equation adapted to remote sensing, and according to Ruhoff et al. (2011), presents two main constraints: the estimation of the stomatal conductance and the estimation of the evaporation directly from the soil, that may vary from 0 to 80%, especially in areas with low leaf area index. The estimation of ET using remote sensing techniques is not made directly, instead it uses other remote sensing products such as land surface temperature, vegetation indices and leaf area index. Thus, its results depend on the quality of the input data that sometimes have very low spatial resolution (110 km) when compared to the output resolution (1 km).

PET and ET data estimated using MODIS (MOD16 products) satellite are consistent when the landcover classification is correct. Otherwise, parameters like vapor pressure deficit and minimum air temperature for stomatal conductance constrains are wrongly selected resulting in less accurate ET estimations (Ruhoff *et al.,* 2011).

Benali et al. (2012) reported low accuracy of the MODIS satellite data in the coastal stations. The authors believe that it is probably related to the spatial variation of relative humidity due to thermal inertia, which would influence the energy available for sensible heating on the surface and, therefore. According to them, the position of the Azores anticyclone during the summer, allows mesoscale circulations to dominate the surface flow, favoring sea breezes and enabling advection masses of moist air from the sea to the land, which increases the fog periods in summer in the center of the West Portuguese coast, where the Leiria Pine Forest is located. This could explain the higher accuracy in the winter months and the increase in the satellite values in the summer.

4.7) Crop Adjusted PET (PET_{CA})

The values of Crop Coefficient (Kc) and PET-HS were used to obtain the values of the crop adjusted potential evapotranspiration (PET_{CA}) that will be applied in the recharge calculations. The PET_{CA} consists of an adjustment of the PET considering the vegetation in the study area and is calculated by multiplying the PET and the Kc. The relation between these parameters is given by Equation 7 below.

$$PET_{CA} = PET_{HS} \times K_c \tag{7}$$

Where Kc is the crop coefficient calculated from the NDVI obtained in MODIS database, PET-HS is the potential evapotranspiration calculated using the Hargreaves method (Hargreaves & Samani, 1985), and PET_{CA} is the crop adjusted evapotranspiration in the area.

PET_{CA} was calculated for both burnt and unburnt areas based on the different Kc time series obtained from the NDVI data and the results are shown in Figure 49. By comparing the PET_{CA} from the Burnt and Unburnt areas we immediately notice that the unburnt area has lower values than the burnt area, it is a reflex of the behavior of the crop coefficient due to the more homogeneous (vegetated) land cover in the burnt area than in the unburnt.

The graph shows an abrupt decrease in the PET_{CA} in the burnt area after the fire, which is completely according to the expectations due to the significant reduction in the vegetation after the fire. About 6 months after the fire, the PET_{CA} starts increasing gradually reaching similar values from the average PET_{CA} before the fire, suggesting a recover of the vegetation as discussed in the previous chapters.

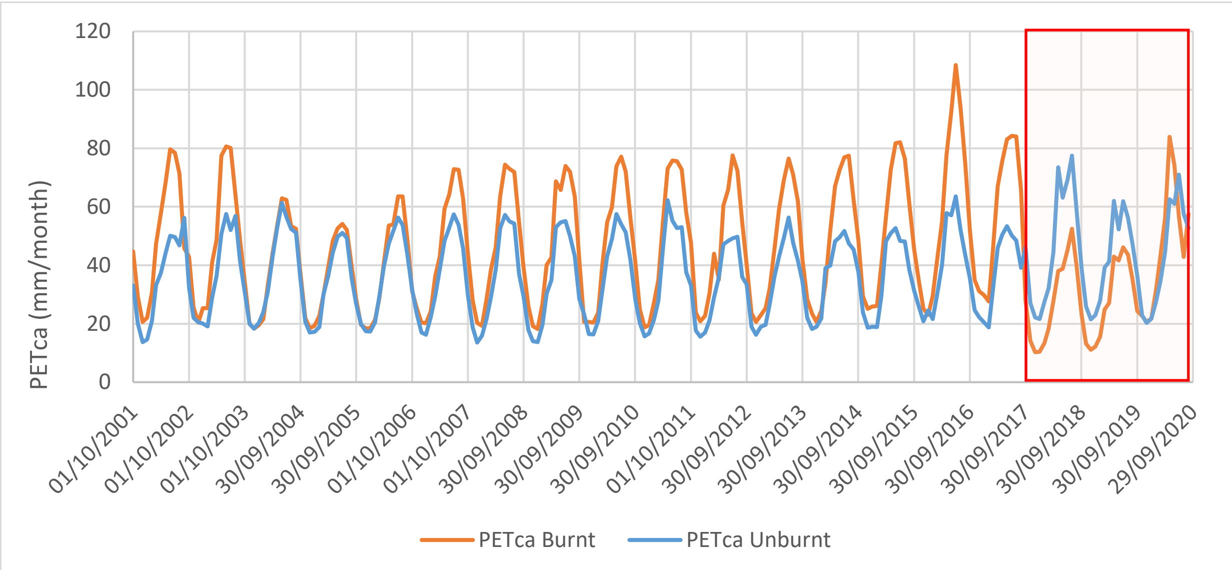


Figure 49 – Crop Adjusted Evapotranspiration (PET_{CA}) estimated for the burnt and unburnt areas of the Leiria Pine Forest.

Another interesting observation is that in the middle of 2003, the values of PET_{CA} in the burnt area reduce reaching the same range observed in the unburnt zone, and gradually increasing until recover average values in 2007. This behavior is probably related to a fire occurred in the Leiria Pine Forest in August 2003 that burnt about 25% of the forest. Since its magnitude was much smaller than the fire from 2017 (that burnt 86% of the forest) the effects in the PET_{CA} are not as strong but can be seen in the time series.

ET is a crucial component of the water balance and the complexity of the processes and interactions able to influence it increases the difficulty in determining it. By comparing different methodologies, we can have a better idea on the influence of climatic, geographic and land cover changes in this parameter, and define the better strategy to be applied according to the specific characteristics of each case. When possible, field observations are also an excellent tool to have a better picture on how things are happening and adjust the analysis to better fit the real conditions, especially after extreme conditions such as droughts, wildfires, and floods.

The use of PET and ET data from MODIS is a limitation in this study because it is too high compared to the other methodologies applied. Nevertheless, despite the overestimation of this parameters by the satellite in the study area, the relation between PET (MODIS) and ET (MODIS) is still valid, even though the values themselves are not.

5) Groundwater recharge

According to Freeze & Cherry (1979), "groundwater recharge can be defined as the entry into the saturated zone of water made available at the water-table surface, together with the associated flow away from the water table within the saturated zone". In simple terms, it can be described as the descending water flow that when infiltrating, reaches the aquifer system resulting in an additional volume to the groundwater reservoir. This process is based on the complex combination between energy and moisture occurring in the zone between atmosphere and subsurface (Smerdon, 2017).

Recharge processes may take place due to simple percolation of rainfall or irrigation water into the subsoil, or as an interaction between rivers, lakes and other water bodies that may eventually lose water to the aquifer increasing its storage.

Groundwater recharge may have its origin from several sources including natural ones such as precipitation and rivers and lakes' contribution or anthropogenic processes like irrigation and artificial recharge and is influenced by climate variables, vegetation, and subsurface characteristics in a specific region such as: occurrence, duration and intensity of precipitation, temperature, humidity, wind velocity, character, and thickness of the soil/rock above the water table, surface topography, vegetation, and land use (Smerdon, 2017; Arnold *et al.,* 2000).

Although groundwater recharge is one of the main parameters to be evaluated when we think about groundwater management and availability, its estimation in very challenging, because it is hard to measure it directly and shows high spatial and temporal variability, adding uncertainties to the estimations (Healy, 2010).

Estimate groundwater recharge requires a broad knowledge and understanding not only on the water flow, but also the climate and geological conditions in the area in order to represent in the best way possible the main processes occurring. There are several methods to estimate recharge, like the Thornthwaite-Matter method (1955), Penman-Grindley method (Penman, 1950; Grindley, 1967), chloride mass balance (Allison & Hughes, 1978), water level fluctuation method (Healy & Cook, 2002), application of numerical models, among other tools. The choice about the best method to each case will depend on the availability of data, precision required in the analysis, the size of the system to be analyzed and the hydrogeologic conditions.

In most of the cases, the absence of data is a major limitation when performing this type of analysis, and to overcome this, it is necessary to adequate the methodology considering the best possible representation of reality that can be achieved with the available data (Healy, 2010).

Remote sensing tools have been extensively applied in recharge estimations (Khan *et al.,* 2020; Carvalho-Santos *et al.,* 2019; Gemitzi *et al.,* 2017; Bruner *et al.,* 2004) due to their capability of provide reasonably quality data with longer temporal and spatial distribution. The main limitations imposed by the use of these techniques when compared to meteorological models are related to mainly to cloud cover, scale factor and low data acquisition frequency (Ruhoff *et al.*, 2011).

The present work aimed to investigate the possible effects of forest fires in groundwater recharge and, in order to achieve it, data with good temporal (before and after the fire) and spatial (burnt and unburnt areas) distribution was needed. Since the study area does not have an established monitoring network and observational datasets were not available, the use of remote sensing data was considered the best option to estimate groundwater recharge even considering the limitations imposed by this decision.

The development of a methodology to evaluate groundwater recharge in a given area involves continuous improvement as new data is available. The comparison between different methodologies is a strategy applied worldwide to minimize the uncertainties and increase the confidence in the results (Collenteur *et al.,* 2020; Labrecque *et al.,* 2020; Yin *et al.,* 2011; Sibanda *et al.,* 2009; Wanhfried & Hirata, 2005).

5.1) Methodology

The water budget method is one of the most widely applied due to its highly theoretical background (Moon *et al.,* 2004), and consists of account all the entrances and exits until the rootzone depth in the soil. The difficulties presented in the direct calculations of the water budget boosted the development of several indirect methods to estimate the water budget using meteorological variables, that are often easier to acquire. The water balance components can be measured (temperature and precipitation) or estimated using semi-empiric formulas (PET and AET).

Although the indirect methods sometimes are the only viable option to estimate groundwater recharge, they are based on simplifications of the natural processes and often, are not able to return precise results. However, even with the intrinsic limitations of these kind of method, they have been applied extensively to provide insights about the best management strategies considering the sustainability of the groundwater resources.

The water balance is calculated for a given area in a specific interval of time, so it enlightens us about the changes in water availability, deficit and surplus and provides a better understanding about the variability of these factors over the studied period.

The Thornthwaite-Matter (1955) is a simplified method that consists in obtaining the groundwater recharge volume using a basic equation described in Eq. 7 below.

$$R = P - ESC - ETR - \Delta S \tag{7}$$

Where P = Precipitation (mm), ESC = Surface flow out of the study area (mm), AET = Actual Evapotranspiration (mm), ΔS = Variation in the water storage in the unsaturated zone (mm) e R = Groundwater Recharge (mm).

The advantages of this methodology consist mainly in the higher availability of this type of data, reduced time in the application and its low cost. Being an indirect method that considers many estimated variables, its main limitation is related to the difficulty in measuring the uncertainties (Wanhfried & Hirata, 2005) which may compromise its accuracy.

5.2) Recharge model

The groundwater recharge simulations were conducted for the burnt and unburnt areas of the Leiria Pine Forest using the Easybal software developed by the Hydrology Group of the

Universitat Politècnica de Catalunya (UPC) for the period of 2001-2020 and the interpretation was divided in two periods: before and after the forest fire occurred in October 2017.

The Easybal software uses climate data (temperature and precipitation), PET and soil property data (field capacity, initial humidity, permanent wilting point, rootzone, lamination value) to determine groundwater recharge in a daily, monthly and annual temporal scale.

The daily climate data (temperature and precipitation) time series was taken from the E-OBS database for the period of 2001-2020 and processed using Python. NDVI indices taken from the MODIS satellite with an interval of 16 days for the period of 2001-2020 were used to estimate vegetation crop coefficient (Kc) based on empiric relations cited in the references. The crop coefficient is used to calculate the adjusted evapotranspiration (PET_{CA}) depending on the type of vegetation in a given area. Kc values reflect land cover changes imposed by the wildfire. Irrigation was neglected in the area since it consists mainly of forest.

Besides the climate time series data, Easybal also need soil parameters that were estimated based on the references available for the study area and calibrated during the simulations (Figure 50). These parameters are explained below.

- Field Capacity (volumetric content) – It represents the moisture in the soil after the water contained in the macropores is drained by gravity action.
- Wilting Point (volumetric content) – It represents the minimal amount of soil moisture that plants require not to wilt.
- Initial Soil Moisture (volumetric content) – It represents the quantity of water contained in the soil in the beginning of the simulation.
- Root Zone (R_z) (m) – It is the zone of the soil profile that is penetrated by plant roots.
- Lamination Value (L_v) (mm/month) – It represents the maximum soil water content value before runoff starts.

The field capacity and wilting point values used are an average of the experimental results available on the Infosolos database from Marques (2010).

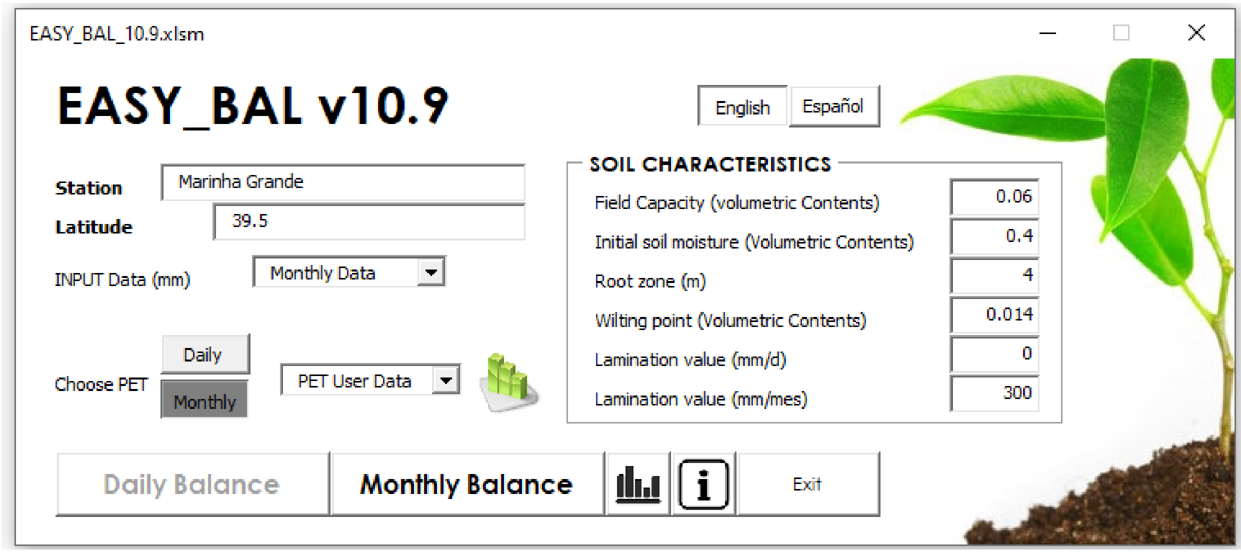


Figure 50 - Easybal software set up example.

It is widely known that plants and their roots are able to affect soil physical properties like infiltration rate, moisture content and aggregate stability (Gyssels *et al.,* 2005). These effects influence directly the water balance, since the presence of plants not only interfere in the evapotranspiration parameter but is also able to decrease runoff by increasing infiltration capacity of the soil.

The root zone parameter was defined based on the relation between chloride concentrations in the unsaturated zone and root depth of the vegetation explained in Appelo & Postma (2005). (Figure 51).

The higher chloride concentrations in the soil profile are present in the rooted upper zone. Below the heathland, the chloride concentrations are low and variate very little, and the higher values can be seen in the first meter of the soil profile while below the pine trees, the chloride concentrations are much higher and vary widely, with the higher concentrations presented in the first 4 meters of the soil profile (Moss & Edmunds, 1989; Appelo & Postma, 2005).

So being, the R_z in the Leiria Pine Forest was assumed to be 4m before the fire considering that the vegetation consisted mainly of pine trees and 0.5m after the fire, when new and more shrubby vegetation started to grow in the area.

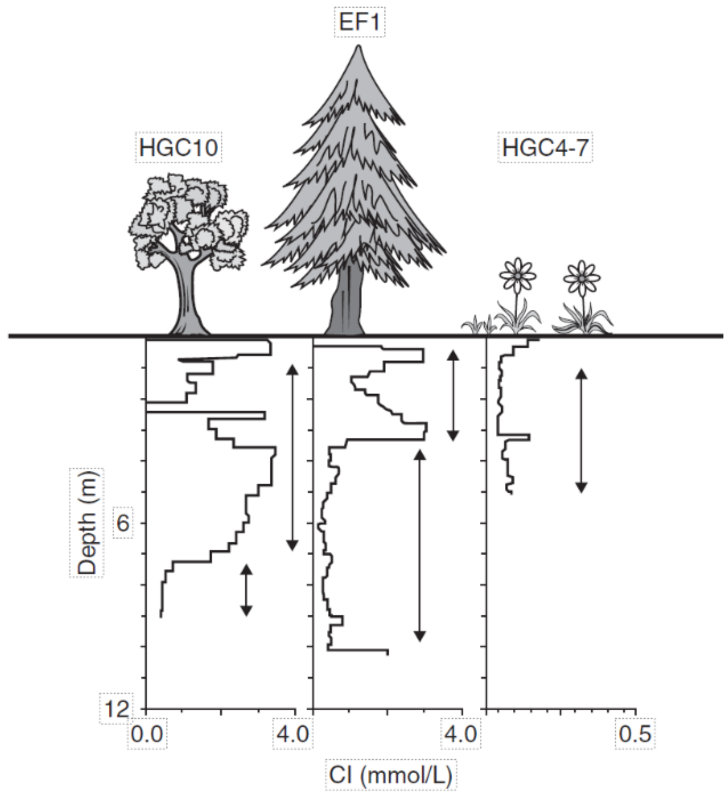


Figure 51 - Chloride profiles in the unsaturated zone of the Sherwood sandstone below birch woodland, Cyprus Pine and heathland (Moss & Edmunds, 1989).

The lamination was estimated based on the characteristics of the study area. The high infiltration capacity and hydraulic conductivity of the very well-sorted sand dunes, combined with the low slope topography suggest that runoff in the area would only occur in exceptional cases and can be neglected in normal conditions. Based on this, several simulations were performed using different lamination values and it was set on 300 mm/month, considering that above this value, the recharge estimations do not get affected (Runoff = 0).

Several studies report that the use of monthly time steps can substantially underestimate groundwater recharge (Mileham *et al.,* 2009; Herrera-Pantoja & Hiscock, 2008), so in order to account the concentrated recharge caused by high events, the recharge simulations included a direct groundwater recharge of 20% of the precipitation (Stigter, 2014).

The Easybal model is a simplification of the reality, and the limitations in its application in this study case include: (1) it does not provide a spatially distributed estimation of groundwater recharge, instead, it presents an average (daily, monthly or annual) value; (2) Easybal does not consider some soil properties that may influence recharge in cases of wildfires such as soil water repellency; (3) The soil parameters insert in the model are based on the available references, (4) errors related to the use of remote sensing data to estimate PET_{CA}, and (5) Groundwater recharge is the output of the model and values of AET used in the calculations are not discriminated.

5.2.1) Simulations Results

The first step to understand the changes in groundwater recharge after the fire is to understand it before the event. In order to have a better picture of its behavior and increase the accuracy in the analysis, a normalized recharge was calculated to be applied in the comparisons as what would be expected if the fire had never happened.

This normalized recharge was estimated using monthly averages of the crop coefficient time series from before the fire (oct2001-sep2017) to calculate a normalized PET_{CA}, that were extrapolated to the period after the fire in the burnt and unburnt areas. After that, these values were compared to the ones observed for groundwater recharge in the area. The results for the unburnt area (UA) are given in Figure 52.

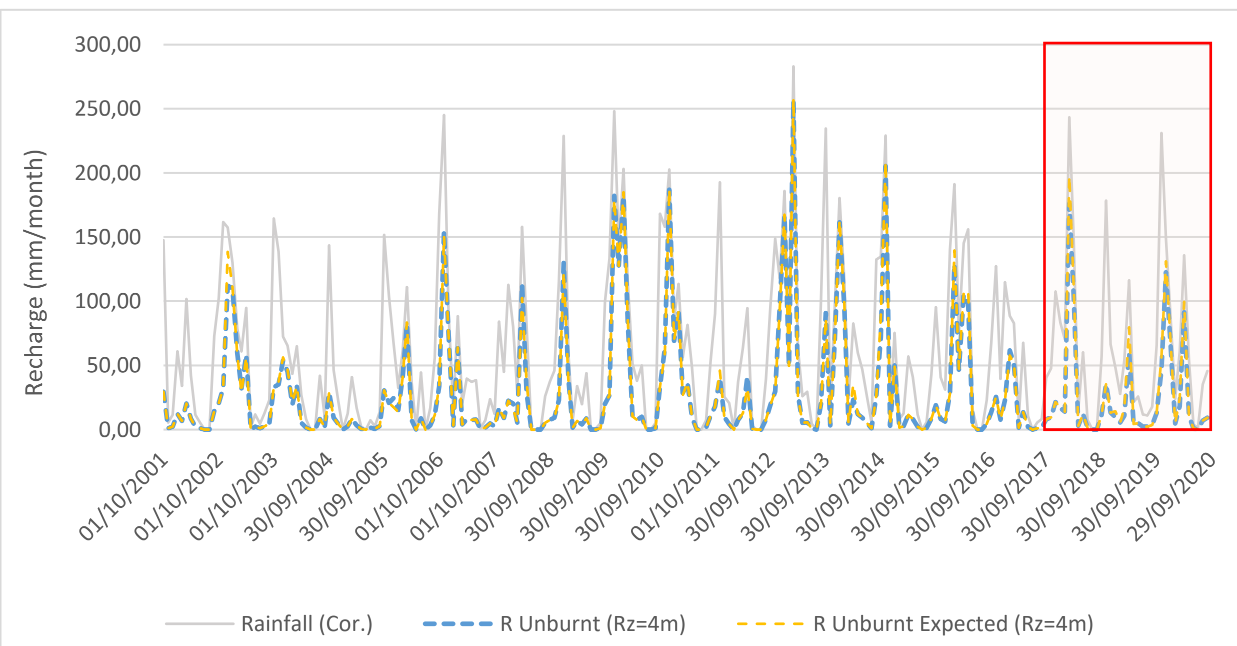


Figure 52 - Groundwater recharge in the unburnt area and normalized groundwater recharge using the average monthly PETca values with 20% precipitation as direct recharge before and after the fire.

Although the annual recharge obtained in the results for the U_A are slightly lower than the expected values obtained using the normalized average calculations, the average recharge do not vary much between the periods before and after the fire and both cases present correlations with r^2= 0.97. Since the expected values were obtained based on the monthly average of the times series and replicated for all the years, interannual variability is not taken into account, which may lead to the small differences between the expected and observed values. Besides, these differences can be seen in the time series both before and after the fire, which is another indication that it is probably not related to the event itself.

The same simulations were made in the burnt area (BA) of the Leiria Pine Forest and the results are shown in Figure 53.

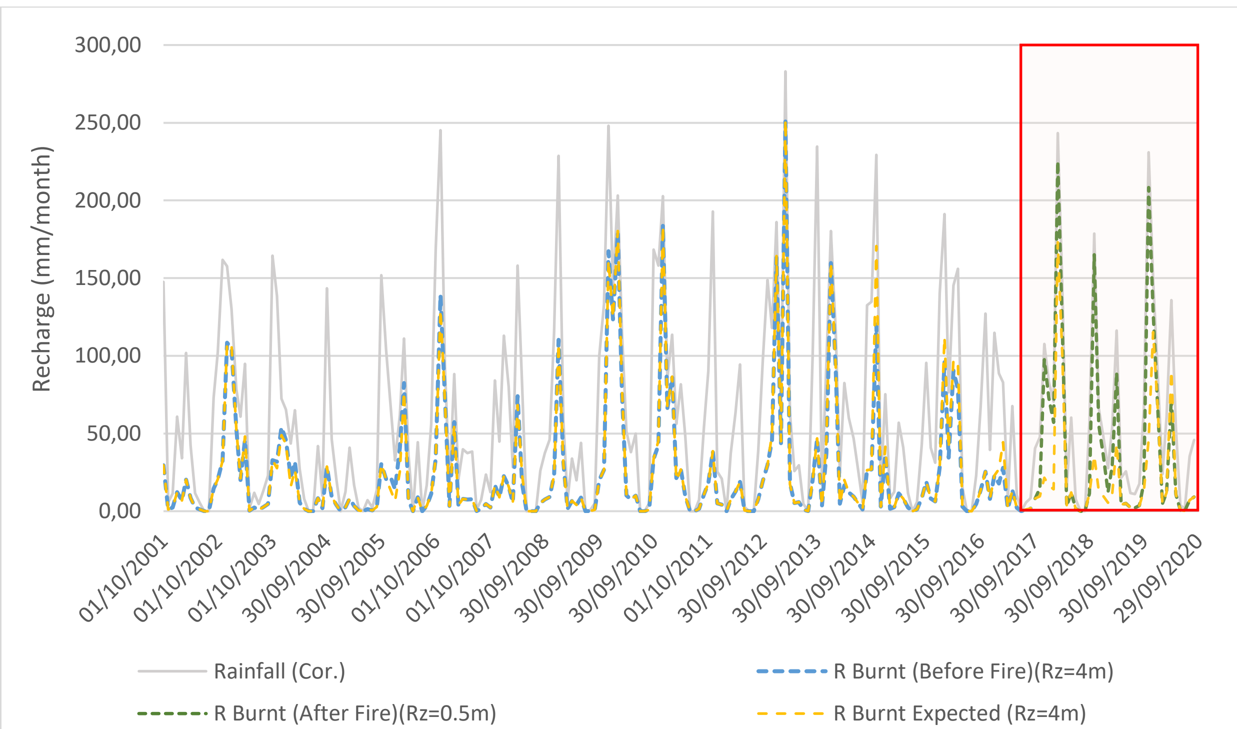


Figure 53 – Groundwater recharge in the burnt area and normalized groundwater recharge using the average monthly PETca values with 20% precipitation as direct recharge before and after the fire.

The average annual groundwater recharge in the burnt area (B_A) in the period before the fire was approximately 37% of the precipitation in the area, and the results are very similar using the times series data and the expected values (r^2=0.99). After the fire, the groundwater recharge in the B_A increases about 50% in the first year, 30% in the second and 17% in the third year when compared to the expected values.

The correlation between the simulated groundwater recharge and the expected recharge values drops from an r^2=0.99 to r^2=0.78 in the years after the fire due to the removal of the vegetation and the decrease in the PET_{CA} that would not occur in normal conditions (Figure 54).

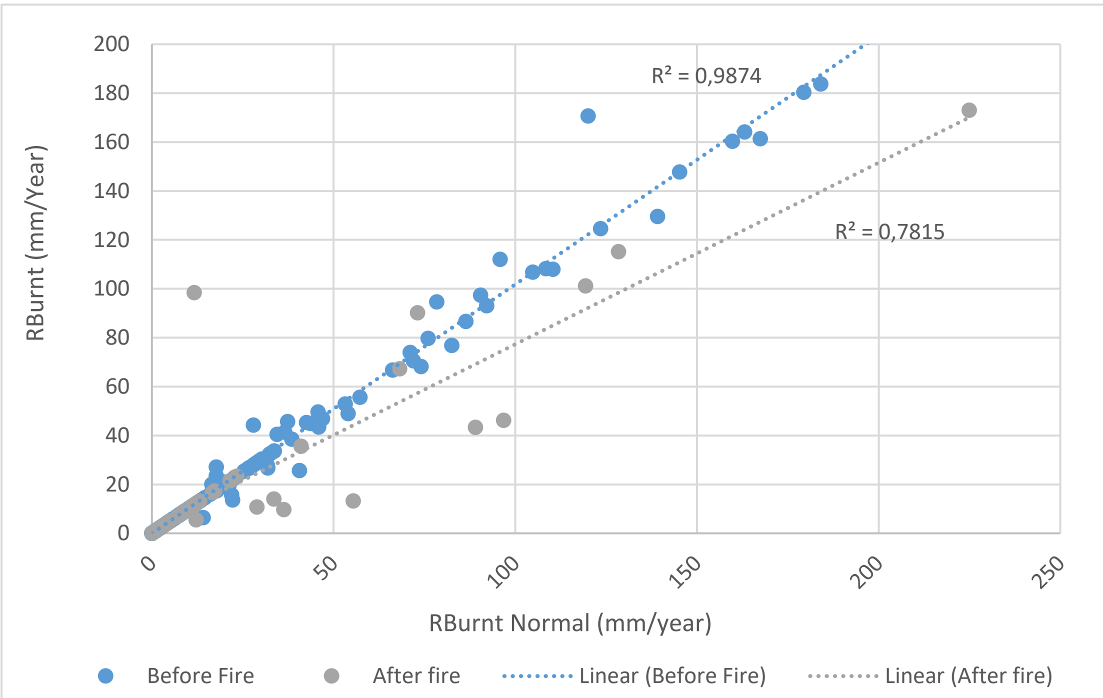


Figure 54 - Correlation between the groundwater recharge in the burnt area and the normalized (expected) groundwater recharge if the fire had never happened.

The total recharge in the aquifer was calculated by doing the weighted mean of the B_A and the U_A obtained from the Easybal simulations plus a direct recharge of 20% of precipitation ((RBurnt + (2 x RUnburnt))/3 + Direct Recharge) and the results are presented in Figure 55.

According to the simulations presented in the study, the groundwater recharge is slightly smaller in the U_A after the fire and considerably higher in the B_A, which means that the aquifer recharge should be globally higher than the expected after the event.

Results show that the groundwater recharge in the aquifer before the fire was about 40% of precipitation and increases approximately 15% in the first year, 7% in the second and 3% in the third year after the fire when compared to the expected values. The correlation between the simulated and expected data before the fire is r^2=0.994 and after the fire it decreases a little to r^2=0.95.

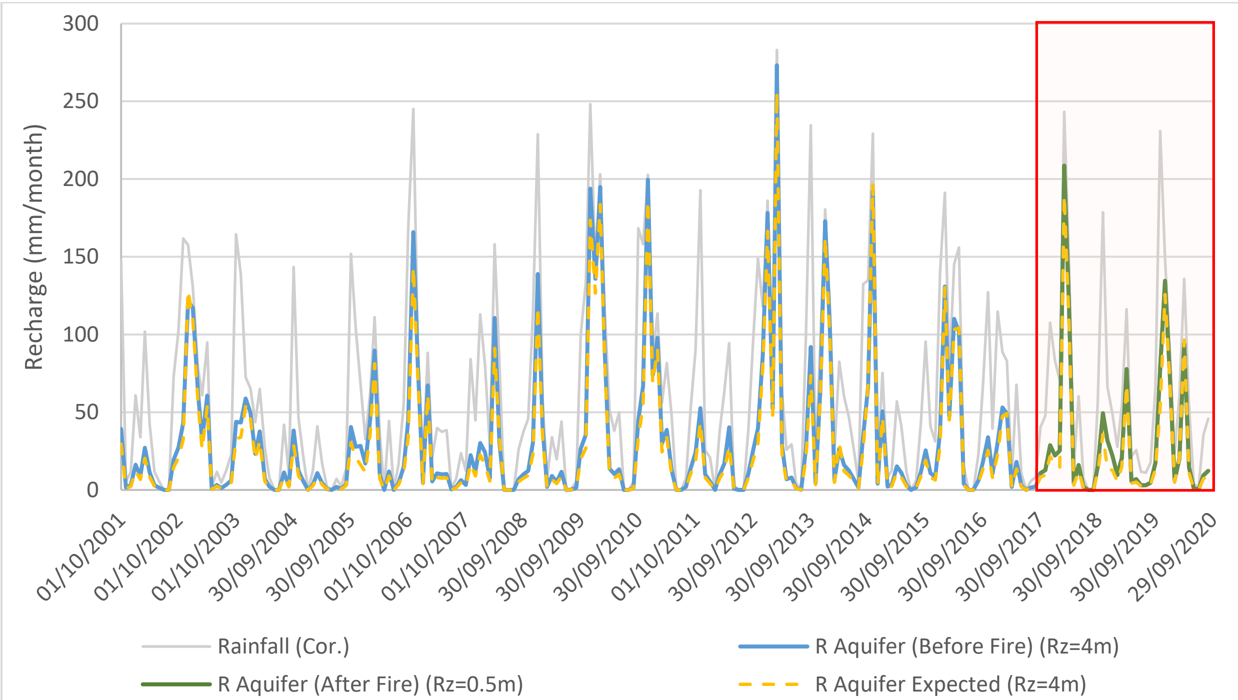


Figure 55 - Groundwater recharge in the total extent of the Vieira de Leiria-Marinha Grande Aquifer.

This attenuation in the increasing when comparing the recharge in the aquifer and in the B_A is probably because the B_A corresponds only to 1/3 of the area while the Unburnt non-affected area corresponds to 2/3 of it, provoking this dilution in the increase of the recharge.

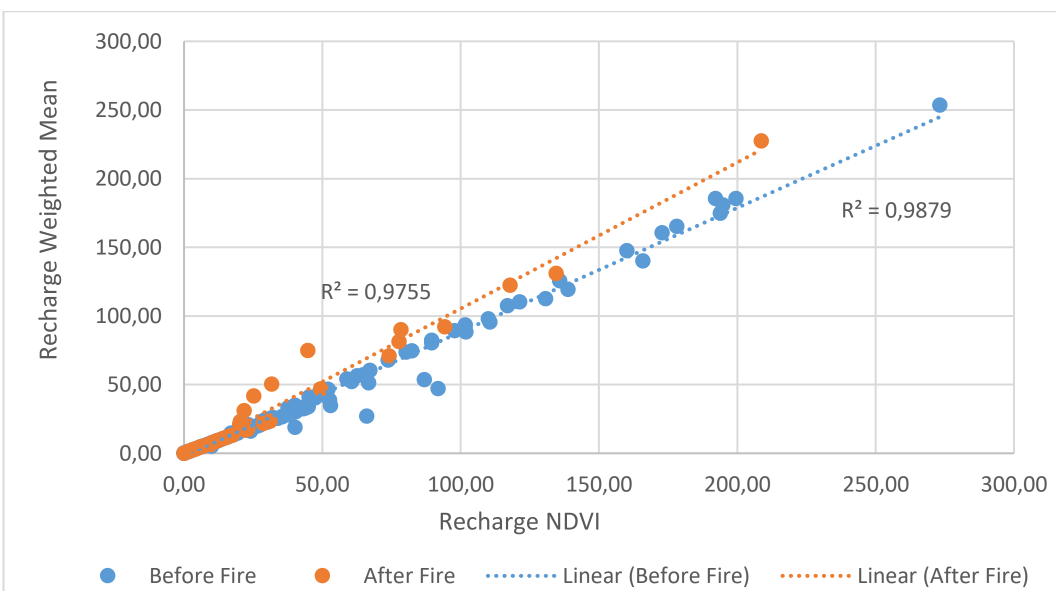


Figure 56 - Correlation between the groundwater recharge obtained by the weighted average of the burnt and unburnt areas and groundwater recharge of the whole aquifer obtained directly using NDVI indices.

In order to verify these results, the total recharge of the Vieira de Leiria-Marinha Grande Aquifer was also calculated using the directly the NDVI data obtained from MODIS satellite. The results simulated have a r^2=0.988 correlation with the ones obtained by the weighted average of B_A and U_A for the period before the fire and r^2= 0.976 after (Figure 56).

6) Impact of Wildfires

Several communities around the world are supplied by water coming from forest catchments that are potentially vulnerable to wildfire and climate change (Bladon *et al.,* 2014; Hallema, 2016). So, understanding the adverse effects of wildfires on the hydrological processes that govern water quantity and quality in these regions is especially important to propose effective management and adaptation measures (Loiselle *et al.,* 2020, Balocchi *et al.,* 2020).

In order to evaluate the impacts of wildfire events it is essential to understand the physical processes occurring in the watershed. Sometimes, technical difficulties such as the lack of monitoring networks and difficult the access in areas after wildfires, make hard to assess the impacts of these events in the field. Because of this, remote sensing techniques have been applied by many researchers (Chaplot, 2007; Feikema *et al.,* 2011; Folton *et al.,* 2015; Ebel *et al.,* 2016; Malagó *et al.,* 2017) to investigate watershed responses to climate change, land cover changes, disturbances, and extreme weather events (Rodrigues *et al.,* 2019).

The impacts of wildfires can be felt not only in the environmental sphere, but they can also represent economic, social and cultural losses to the population through properties and cultural patrimony destruction, damages in infrastructure, biodiversity and human life losses (Efthimiou *et al.,* 2020).

The role of wildfires as an agent of vegetation and geomorphological change has been widely discussed (Shakesby, 2011) and although the events have been considered not only natural, but also essential in the environment, the discussion about their impacts has been an important topic due to their increase in the number, frequency, and intensity in many regions of the planet.

The most obvious and discussed environmental impacts of wildfires are related to the vegetation losses, disruption of soil's physical properties and soil erosion in the affected areas and they have been discussed in many papers (Efthimiou *et al.,* 2020; Coop *et al.,* 2016; van Leeuwen *et al.,* 2010; Lentile *et al.,*2007; Shakesby, 2006; Shakesby *et al.,* 1993). The changes in vegetation and soil properties such as infiltration rates, porosity, conductivity and storage capacity may exert considerable influence on the hydrologica functioning of the watershed including alteration of evapotranspiration rates, increasing runoff and sedimentation rates and, partitioning of precipitation by forest canopies, ultimately influencing the amount of water that percolates the soil recharging the water table.

Table 4 - Summarized changes in hydrologic processes produces by wildfires (Adapted from Neary *et al.*, 2005).

Hydrologic Process		Specific Effect
Interception	↓	The reduction of vegetation acting as a barrier between precipitation and the soil surface reduces water loss by interception.
Transpiration	↓	Temporary elimination of transpiration, increasing streamflow and soil moisture content.
Infiltration	↑	Increased by the reduction in evapotranspiration and negligible runoff in the study area.
Baseflow	↑	Increased by the reduction in evapotranspiration.

The results showed in the previous section show an increase in groundwater recharge after the wildfire occurred in the Leiria Pine Forest in October 2017 when approximately 86% of the pine trees were burnt. Some of the factors controlling the post-fire changes in recharge rates in the area are summarized in Table 4 and discussed below.

6.1) Vegetation Loss and Evapotranspiration

Vegetation influences the water cycle in several ways controlling interception processes, water retention, runoff, infiltration rates and transpiration from plant canopy. The evapotranspiration represents the largest loss of water in the hydrological cycle (Neary *et al.,* 2005) and since it consists of the sum of the direct evaporation from soil, plants leaves and water bodies, and the transpiration of the vegetation in a given area, it is widely affected by the occurrence of wildfires due to the changes in plant physiological properties (amount of leaves and root distribution) that control the amount of water being intercepted, evaporated and/or transpired (Anurag *et al.,* 2021).

Interception is the process by which vegetation and accumulated litter act as a barrier protecting the soil from the direct contact with precipitation. A significant amount of precipitation intercepted by vegetation returns to the atmosphere by evaporation, representing a reduction in the amount of water entering the soil profile (Neary *et al.,* 2005).

Forest areas usually evaporate more water than areas covered with shorter vegetation due to greater rainfall interception (Valente *et al.,* 1997). Several studies report a rainfall interception in temperate coniferous forests is between 10-60% of precipitation (e.g. Valente *et al.,* 1997; Teklehaimanot *et al.,* 1991) and the main factors responsible for that are the bigger canopy storage and aerodynamic conductance.

The occurrence of wildfires drastically alters plants' physiological properties by eliminating the leaves in the burnt area, reducing the leaf area index (LAI) and consequently, the specific area of leaves acting not only as a barrier to between the rainfall and the soil, but also as a favorable surface for evaporation to occur. The reduction in interception causes a decrease in soil moisture, increasing water yields and in runoff processes.

Since the transpiration processes occur in the leaves of the plants, the elimination of these leaves by the fire, besides affecting interception, also causes a sharp decrease in transpiration, which increases soil moisture and the streamflow (Neary, 2002).

The modification in plant physiological properties have the potential to induce changes in hydrological processes including groundwater recharge through the decrease in interception, evaporation and transpiration by plants' canopy and consequent decrease in evapotranspiration and increase in the net precipitation available for streamflow (Anurag *et al.,* 2021; Poon & Kinoshita, 2018).

The decrease in the crop adjusted potential evapotranspiration in the burnt area of the Leiria Pine Forest after the fire was 92% in the first year, 18% in the second year and 0% in the third year when compared to the expected values, which suggests a recover of ET after three years due to the growth of new vegetation in the burnt area. We also observe a similar behavior in the period between the end of 2003 and 2006, this is probably related to the fire that happened in August 2003 (Figure 57).

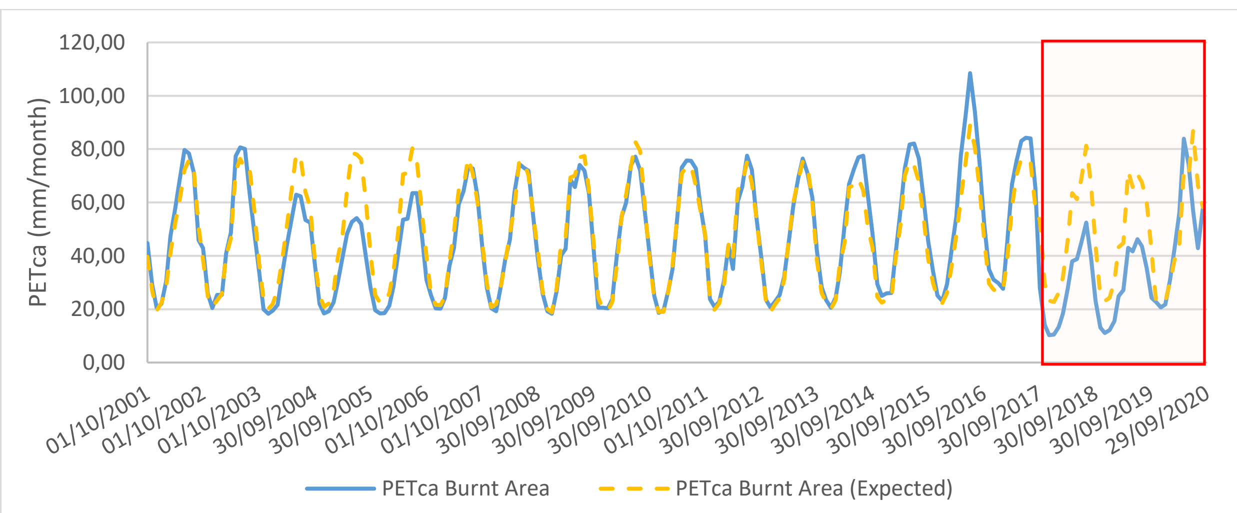


Figure 57 - PETca expected (if the fire has never happened) and PETca observed in the burnt area of the Leiria Pine Forest.

Results from Häusler et al. (2018) also show a reduction in ET values in fire-affected areas in Portugal, mostly in the first year after the fire. Analysis in the second year after the fire showed a significant recover in ET values towards the pre-fire conditions that the authors defined as in the natural variations range, although it does not mean that the forest has returned to its former conditions.

The recovery in ET values in the Leiria Pine Forest after the first year may also be related to the fast growth of a bushy and scrubby vegetation in the area just after the fire that according to Garcia-Estringana et al. (2010) has high rain interception in Mediterranean climate, contributing to increase in ET.

The application of NDVI to calculate PET_{CA} may also represent a limitation in terms of recovery evaluation, since besides the changes in vegetation cover contemplated by satellite calculations, the potential increase in SWR and other factors may also limit water storage decreasing ET after the fire (Nunes *et al.,* 2016).

Although the reduction in ET certainly contributes to the increase in groundwater recharge (50% in the first year, 30% in the second and 17% in the third year when compared to the expected values) most likely, it is not the only factor responsible for it (Figure 58).

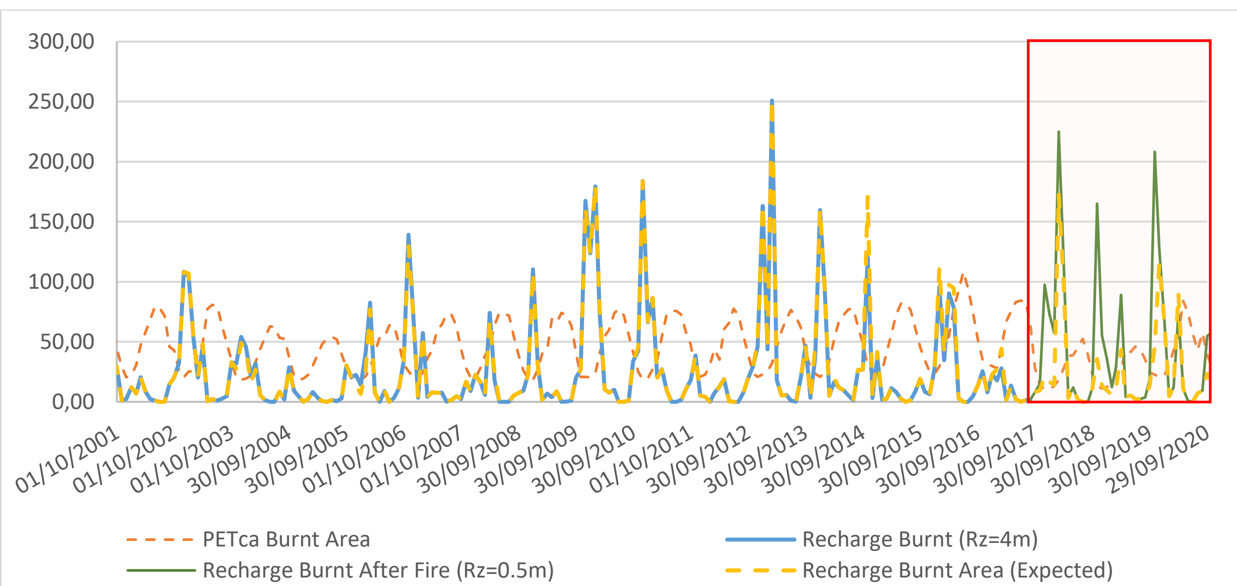


Figure 58 – Relation between estimated PETca and the estimations of expected and observed groundwater recharge in the burnt area.

Besides the increase in soil water due to the decrease in ET, in some cases, the loss of vegetation may have other kind of effect. According to Hyde et al. (2007), the removal of the canopy's shades and the insulation provided by the litter, combined with the increase in the heat may lead to hyper-desiccated soils, increasing capillary suction and greater lag times to maximum infiltration rates.

6.2) Runoff and Water Repellency

Soils in specific conditions of climate and vegetation may develop soil water repellency (SWR) after the occurrence of a fire. It can decrease infiltration rates and involves physical and chemical processes with important implications for plant growth, soil erosion and runoff (Nunes *et al.,* 2016; Santos *et al.,* 2013). Since SWR development is directly related to the presence of organic matter, it can be found in both fire and non-fire environments.

The water-repellent (hydrophobic) layer is formed when the heat from the fire vaporizes organic substances that are transported downwards and subsequently condensed after temperature decreases (Ferreira *et al.,* 2008; Neary *et al.,* 2005). After it, water cannot infiltrate through, increasing overland flow and surface erosion in the affected regions. According to DeBano (1981) the formation of the water-repellent layer occurs when soil temperature rise above 176ºC and it is destroyed when temperature is above 288ºC.

Despite soil SWR has been reported in many cases worldwide, its mechanisms are not completely clear yet. In fire environments, the effects can be highly variable depending on the soil heating regime, type of organic matter consumed and the amount of available oxygen during burning; in some cases, fires could even decrease the intensity of SWR, while increasing its persistency (Nunes *et al.,* 2016).

SWR is highly variable in time and space and varies nonlinearly with soil moisture content, with the formation of a transition zone between wettable and repellent conditions where the soil can act as repellent or wettable (e.g. Rueda *et al.,* 2015). Although there is not a threshold defined for completely disappearance of the water-repellent layer, according to MacDonald and Huffman (2004), it should be about 10% in unburnt areas up until 28% in severely burnt areas.

Several studies found SWR to be most likely associated with the finer cross fraction of the soil (Santos *et al.,* 2013; Leelamanie & Karube, 2011; Doerr *et al.,* 2006; De Jonge *et al.,* 1999), although it could happen the opposite way in different conditions. Studies also points to a relationship between SWR and soil organic matter content and/or quality, but further research is necessary to clear the subject (Santos *et al.,* 2013).

According to Coelho et al. (2004), fire severity is a determinant factor in spatial variability of water repellent soils in burnt areas. The higher severity is related to more extreme and homogeneous SWR, while in low severity areas, the effect is more heterogeneously distributed, favoring the development of preferential flow paths which may locally increase infiltration in some areas (e.g. Diamantopoulos *et al.,* 2013; Blanco-Canqui & Lal, 2009).

The effects of severity and spatial distribution of SWR are highly dependent on the analyzed scale, so overland flow and erosion rates tend to decrease significantly in wider systems when compared to smaller ones due to the connectivity of water and sediment processes. In a catchment scale, the spatial variability of SWR, and the existence of preferential infiltration areas can decrease the impact of SWR on water balance, especially if the fire intensity is not homogenous. (Ferreira *et al.,* 2008).

Among the factors able to affect the water balance, several authors point runoff as a very important one in cases of wildfire (Loiselle *et al.,* 2020; Chen *et al.,* 2015; Ebel *et al.,* 2012). The organic matter in the soil contributes to soil structure and porosity and is deeply affected by fires. The loss of vegetation and changes in the soil structure after a wildfire may reduce soil roughness, triggering severe erosion, reducing infiltration rates and increasing the availability of loose sediments (Ferreira *et al.,* 2008). At burnt areas, the absence of physical barriers (vegetation) enhances the transmission of water and sediments from the slopes to the channels.

When speaking specifically about wildfires, normally the SWR tend to be homogeneous, except for some preferential flow paths created by macropores, which favors overland flow and sediment transport. Thus, in burnt areas, the combination of reduced interception and increased SWR, with consequent decrease in infiltration capacity are the main factors responsible to enhance runoff generation and erosion rates (Ferreira *et al.,* 2008; Neary *et al.,* 2005).

The understanding of the consequences of wildfires is quite complex because the extension of the effects will be directly linked to specifics characteristics of the area (topography, soil organic matter content, vegetation, etc.) as well as the fire characteristics (severity, intensity, extension). In the Leiria Pine Forest case, we believe that the smooth topography combined with a geological background consisting mostly of very well-sorted sand dunes, with high hydraulic conductivity favors infiltration keeping the overland flow to a minimum, even after the fire.

According to the fire severity analysis in Fernandes & Guiomar (2018) the fire in the Leiria Pine Forest had predominantly extreme (37.1% of the burnt area), high (27,4%) and very high (17,6%) severities. The areas with Moderate to low fire severity were about 18% of the total burnt area.

This distribution was expected to enhance the SWR effect in the area. However, the favoring of macropores formation by the combustion of pine trees' roots probably contributed to hamper the effects of the formation of the SWR layer. Besides, the persistence of this layer is related to the soil moisture content, considering that the fire in Leiria occurred in the beginning of the wet season, the increase in precipitation could have contributed to the fast disappearance of the water repellent layer.

6.3) Rooting System

Another very important factor capable of influence the infiltration rates and consequently groundwater recharge, is the rooting system. After a wildfire, the complete elimination of the vegetation may have severe and long-lasting consequences.

The rooting system is responsible for stabilizing the soil, influencing its effective hydrological depth and increasing flow roughness. The prolonged heating of the soil during a fire produces temperatures that essentially sterilize the upper part of the soil killing microbial population, small invertebrates, insects and plant roots. It leads to an increase in soil and organic matter loss, modification of porosity, alteration in aggregate stability, depletion of nutrients, increase in bulk density and sediment yield, and decrease in infiltration (Efthimiou, 2020; Neary *et al.,* 2005; Shakesby, 2011; Hyde *et al.,* 2007; Shakesby & Doerr, 2006; Inbar *et al.,* 1998).

The creation of macropores by soil cracks, high stone content and/or plants roots has been reported to explain why high runoff coefficients after fire are observed in micro-scale studies and are not reflected in catchment scales in the Mediterranean region (Shakesby, 2011; Doerr *et al.,* 2003). While the macropores can be filled with ashes and fine soil particles decreasing even more infiltration rates (Martin & Moody, 2001), in specific conditions these macropores may act as preferential pathways to water infiltration, increasing it instead reducing.

The presence of extensive macropores systems in burnt pine areas due to the combustion of rotten root systems may hamper the SWR effect, helping in the control of excessive overland flow (Ferreira *et al.,* 2008).

According to Neary et al. (2005) vegetation changes have a smaller effect on subsoil properties that influence soil water storage, so they are not likely the main drivers of the catchment hydrological cycle. However, if these vegetation changes affects not only infiltration, but also evapotranspiration, as in the Leiria Pine Forest, it may influence soil water storage.

Different types of plants have different root depths. During the groundwater recharge simulations for the Leiria Pine Forest the root depth parameter (Rz) was modified from 4 meters before the fire to 0.5 meters after it based on the available references and changes in vegetation observed during fieldwork (Appelo & Postma, 2005). Deeper roots like the ones from pine trees can penetrate deeper into the soil and abstract water directly from the shallow aquifer (water level at 2m depth in the study area). After the fire, the weakening and removal of the pine roots and subsequently substitution by other vegetation species (shrubs and bushes) with shallower root depths (Rz=0.5m), that are only able to use water stored in the topsoil layer, may have contributed to the increase in groundwater recharge.

7) Conclusions

Groundwater recharge estimation is a complex process because it requires comprehension of the physical conditions of the study area as well as all the processes and interactions conditioning recharge. There are several methodologies that may be applied to overcome the challenges involved in these estimations, since most of them consist of simplifications of the reality, choosing the best one should always consider the characteristics of the study area (e.g. climate, soil type, geology, topography) and the quality of the available data.

The use of remote sensing allowed us to overcome the limitations of temporal and spatial distribution of data in the study area, and although its use still presents several uncertainties related to the calculation, interpolation, atmospheric conditions, accuracy of the satellites depending on the physical and atmospheric conditions, they represent a very powerful methodology. Nevertheless, the interpretation of the results should always be critically analyzed and validated using field data when possible. In order to check the results, increase accuracy and give more credibility to the results, the PET data from MODIS was compared to the results calculated using PM-FAO and the Hargreaves method revealing limitations in the use of the ET satellite data.

Understanding how to gather and process satellite information, as well as its limitations considering the algorithms and calculations behind it was a challenging process that helped in the critical analysis and interpretation of the results in each step of the study.

The ET estimations in the Leiria Pine Forest show a significant decrease in the first year after the fire and a fast recovery leading almost to the pre-fire values in the third year after the fire. These results show that the decrease in ET is probably not the only factor increasing recharge in the study area since the simulations estimate an increase of 17% in recharge in the third year after the fire. Other factors such as higher precipitation and/or reduction of ET due to lower atmospheric demand could compensate land cover changes, increasing groundwater recharge (Hawtree *et al.,* 2015). Since the Easybal simulations are mainly controlled by changes in ET and soil properties, an efficient way to enhance its estimations is the improvement in climate and soil property characteristics' data.

It is difficult to isolate the changes in rooting depth, SWR, runoff and climate factors and the interactions among themselves. Nevertheless, the geological background and topographical conditions seem to exert a decisive role in prevent increase runoff in the Leiria Pine Forest, which combined with decreasing ET and presence of preferential flow paths (macropores) could result in the increase in groundwater recharge in the Leiria Pine Forest region.

Although SWR has been reported by several studies as an important factor after the occurrence of wildfires, it does not seem to have a big influence in the study area regardless the severity of the fire. It could be related to the small persistence time of the water repellent layer due to climate conditions after the fire or even the presence of the macropores that favored infiltration in the area attenuating SWR effects.

Thus, predicting the hydrological impacts of wildfires in a watershed is a very complex process that requires a combined understanding of the consequences of land cover and soil properties changes, as well as climate variability (Hawtree *et al.,* 2015). The recharge simulations consist of a simplification of the reality and have several uncertainties and limitations associated. The main limitations detected in the present study include: (1) Model simplifications do not account for SWR; (2) Poor estimation of soil properties based on the references that do not consider possible changes after the fire; (3) Accuracy in the satellite data used to estimate crop adjusted evapotranspiration, and (4) Difficulty in isolating wildfire and climatic variability effects.

Some recommendations in order to continue improve the knowledge brought up about the wildfires impacts on the Vieira de Leiria-Marinha Grande Aquifer by the present research are: (1) Deeper analysis of the meteorological conditions in the study area to better understand the role of climatic variability; (2) Installation of meteorological and groundwater monitoring networks to improve data availability and model calibrations; (3) Collection of monthly samples to gather Cl concentration data for future validation of recharge modelling estimations; and, (4) Investigation of the potential impacts on groundwater quality in burnt and non-burnt forest area.

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