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PhD in Risk and Environmental, Territorial and Building Development

Coordinator: Prof. Francesco Fiorito

XXXVIII CYCLE
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Department of Civil, Environmental,
Building Engineering and Chemistry

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Development of a simplified hydrological model
to investigate wetland regimes for conservation
and restoration purposes

Development of a simplified hydrological model to investigate wetland regimes for
conservation and restoration purposes

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06



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ed essendo stato ammesso a sostenere l'esame finale con la prevista discussione della tesi dal titolo:

Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes

(Sviluppo di un modello idrologico semplificato per lo studio dei regimi idrici delle zone umide finalizzato alla conservazione e al ripristino)

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	Abstract Wetlands represent one of the planet's main environmental resources, as they provide a wide range of ecosystem services, including flood protection, carbon sequestration, and support for biodiversity. The complexity that characterizes wetlands, on the one hand, ensures their ecological efficiency, but on the other hand, results in a significant vulnerability to various pressures, mainly related to socio-economic factors (e.g., population growth, urbanization, and the consequent demand for water and land resources) and climatic factors (e.g., rising temperatures and altered precipitation regimes). Coastal wetlands are even more vulnerable, as they are also subject to phenomena such as sea level rise or salinization of the aquifer that often feeds them. The degradation and progressive disappearance of wetlands is now a globally recognized issue, which has led to the adoption of regulations and measures to counter this phenomenon and, where possible, allow their restoration and protection. In this direction, it is essential to rely on observations and/or models capable of characterizing the state of wetlands and their possible evolution under a variety of conditions. Modelling, in particular, is increasingly used to support decision-makers in identifying mitigation and adaptation strategies, through the simulation of key dynamics (ecological, hydrological, biogeochemical) and the development of alternative scenarios. This work focuses on hydrological modelling for wetlands, analyzing the wide range of solutions already adopted in the literature and aiming to respond to one of the most relevant open research questions in the field: how to reliably estimate the water requirements of wetlands so that they can be considered alongside anthropogenic demands in water planning and management processes. To this end, a hydrological model, WetMAT, is proposed. It was developed to reproduce the hydrological regime of wetlands in a simple and parsimonious way and is capable of providing, through its outputs, useful data to support decision-making for the conservation of these valuable ecosystems. The model aims to overcome the constraints imposed by the limited data availability typical of these contexts and is conceived as a dynamic tool capable of adapting to the high climatic and, consequently, hydrological variability that often characterizes such environments. WetMAT is applied to two significantly different case studies: the Doñana wetland in Spain and Torre Guaceto in Italy. Despite their differences, these areas share some features such as belonging to the Mediterranean climate zone, their coastal location, and the strong conflict between anthropogenic pressures (mainly agricultural) and the conservation of the natural environment. This work shows how the same model, adapted according to the issues of the case study, its spatial extent, and hydro-geomorphological characteristics, can be used to estimate variables such as the extent of flooded areas, their depth, or salinity trends in order to provide useful insights for the protection and conservation of wetlands, promoting a holistic planning and management approach to water resources.			Coordinator: Prof. Francesco Fiorito XXXVIII CYCLE Curriculum: CEAR-01/B DICATECh Department of Civil, Environmental, Land, Building Engineering and Chemistry		
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Politecnico
di Bari

Department of Civil, Environmental, Land, Building Engineering and Chemistry

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Doctoral Thesis

Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes

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EXTENDED ABSTRACT (eng)

Wetlands represent one of the planet's main environmental resources, as they provide a wide range of ecosystem services, including flood protection, carbon sequestration, and support for biodiversity. The complexity that characterizes wetlands, on the one hand, ensures their ecological efficiency, but on the other hand, results in a significant vulnerability to various pressures, mainly related to socio-economic factors (e.g., population growth, urbanization, and the consequent demand for water and land resources) and climatic factors (e.g., rising temperatures and altered precipitation regimes). Coastal wetlands are even more vulnerable, as they are also subject to phenomena such as sea level rise or salinization of the aquifer that often feeds them. The degradation and progressive disappearance of wetlands is now a globally recognized issue, which has led to the adoption of regulations and measures to counter this phenomenon and, where possible, allow their restoration and protection. In this direction, it is essential to rely on observations and/or models capable of characterizing the state of wetlands and their possible evolution under a variety of conditions. Modelling, in particular, is increasingly used to support decision-makers in identifying mitigation and adaptation strategies, through the simulation of key dynamics (ecological, hydrological, biogeochemical) and the development of alternative scenarios. This work focuses on hydrological modelling for wetlands, analyzing the wide range of solutions already adopted in the literature and aiming to respond to one of the most relevant open research questions in the field: how to reliably estimate the water requirements of wetlands so that they can be considered alongside anthropogenic demands in wa-

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keywords: *wetlands; hydrological modelling; environmental water requirements; environmental restoration; water resources management*

EXTENDED ABSTRACT (ita)

Le zone umide costituiscono una delle principali risorse ambientali per il Pianeta, dal momento che forniscono un gran numero di servizi ecosistemici, tra cui la protezione dalle alluvioni, il sequestro di carbonio e il sostegno alla biodiversità. La complessità che caratterizza le zone umide per un verso ne garantisce l'efficienza ecologica, per altro verso ne causa una significativa vulnerabilità rispetto a diverse pressioni, legate a fattori principalmente socio-economici (ad es., crescita demografica, urbanizzazione e conseguente necessità di risorse idriche e del suolo) e climatiche (ad es., aumento delle temperature e alterazione del regime delle precipitazioni). Le zone umide costiere risultano ancora più vulnerabili, poiché soggette anche a fenomeni come l'innalzamento del livello del mare o la salinizzazione della falda che spesso le alimenta. Il degrado e la progressiva scomparsa delle zone umide è ormai un problema riconosciuto a livello globale che ha spinto all'adozione di normative e provvedimenti per contrastare tale fenomeno e laddove possibile, permetterne ripristino e tutela. In tale direzione, è fondamentale disporre di osservazioni e/o modelli in grado di caratterizzare lo stato delle zone umide e la possibile evoluzione delle stesse in una molteplicità di condizioni. La modellistica, in particolare, è sempre più utilizzata per supportare i decisori nell'individuazione di strategie di mitigazione e adattamento, attraverso la simulazione delle dinamiche principali (ecologiche, idrologiche, biogeochimiche) e lo sviluppo di scenari alternativi.

Questo lavoro si concentra sulla modellistica idrologica per le zone umide, analizzando l'ampio spettro di soluzioni già adottate nella letteratura e cercando di rispondere a una delle domande di ricerca aperte di maggiore interesse per il settore: come stimare

in maniera affidabile i fabbisogni idrici delle zone umide affinché possano essere considerati unitamente ai fabbisogni antropici nei processi di pianificazione e gestione della risorsa idrica. A questo scopo viene proposto un modello idrologico, WetMAT, sviluppato per riprodurre il regime idrologico delle zone umide in modo semplice e parsimonioso, ed in grado di fornire, attraverso i suoi risultati, dati utili al supporto decisionale per la conservazione di questi preziosi ecosistemi. Il modello mira a superare gli ostacoli dettati dalla limitata disponibilità di dati in questi contesti, e si configura come uno strumento dinamico capace di adattarsi all'elevata variabilità climatica e di conseguenza idrologica che spesso caratterizza tali ambienti. WetMAT viene applicato a due casi studio significativamente differenti ovvero la zona umida di Doñana, in Spagna e quella di Torre Guaceto, in Italia. Nonostante le loro peculiarità, queste aree condividono alcune caratteristiche come l'appartenenza alla fascia climatica mediterranea, la posizione costiera e il forte conflitto tra pressioni antropiche (principalmente agricole) e conservazione dell'ambiente naturale. Il lavoro mostra come lo stesso modello, adattato in base alle problematiche del caso studio, alla sua estensione e caratteristiche idrogeomorfologiche possa essere in grado di stimare variabili come l'ampiezza delle aree allagate, la loro profondità o l'andamento della salinità per fornire soluzioni utili alla sua tutela e conservazione delle zone umide, promuovendo una pianificazione e gestione olistica della risorsa idrica.

parole chiave: : zone umide; modellistica idrologica; fabbisogni idrici ambientali; ripristino ambientale; gestione della risorsa idrica

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INTRODUCTION

The Earth's surface features a topography characterized by elevations and depressions creating a wide variety of landscapes and ecosystems. Topography, water availability, and climatic conditions are the main factors controlling and determining nearly all ecosystems across the globe. Among these terrestrial ecosystems, such as mountains, deserts, and forests, there is another vital ecosystem that warrants attention: the wetland ecosystem. As the name suggests, wetlands are lands characterized by the presence of water, either permanently or intermittently. These areas typically occur along the edges of watercourses and water bodies, in topographically low-lying regions. Wetlands serve as interfaces between land and water, playing crucial ecological roles (Balasubramanian, 2011), saturated or inundated areas for long enough periods to influence soil, plants and animals that live there (Cherry, 2011). A general definition for the term 'wetland' was given in the Ramsar Convention in 1971, when wetlands became relevant to the international community: before this, it was very difficult to define them, both because of their heterogeneous characteristics and because of the different meanings they had for the various groups of people involved in the study and management of these environments (Scholz, 2011). According to the Ramsar Convention (1971) they are defined as follows: *"Wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water, the depth of which at low tide does not exceed six meters"*.

In recent years, as attention to these ecosystems has increased, differing opinions have emerged regarding the distinction between wetlands and other types of water bodies such as lakes or ponds. Recent studies, including Richardson et al. (2022), have sought to distinguish ponds from wetlands, arguing that ponds differ in terms of surface area, depth, proportion of emergent vegetation, and, most notably, biogeochemical characteristics that set them apart from both lakes and wetlands. Such a distinction, grounded in ecosystem evidence, could indeed improve environmental monitoring and legal protection. In practice, however, ponds like wetlands can be ei-

ther artificial or temporary, and current national and international regulations, as well as restoration protocols, generally do not differentiate between them. Moreover, in the scientific literature, ponds are often discussed within wetland contexts (Coccia et al., 2024) or used interchangeably with wetlands (Smalling et al., 2021). For this reason, the present work adheres to the most widely accepted definition of wetlands, that of the Ramsar Convention, since no universally recognised or officially endorsed criteria currently exist for further distinction.

Since ancient times, wetlands have played a predominant role in the development of civilisations, such as in Mesopotamia or Egypt (Butzer, 1976), providing sustenance in the form of agriculture and livestock farming, and religious beliefs thanks to flooding, but also developing technical thinking for their management. Traces of this strong relation can also be found in Native American tribes (LaPier, 2017), particularly in relation to spiritual aspects, and in the Aboriginal cultures of Australia. The advancement of technology in Europe during the Middle Ages, on the other hand, led to wetlands being considered “wastelands” that needed to be reclaimed for agriculture or urban development, serving only an aesthetic function (Menotti and O’Sullivan, 2012). In Asia, too, wetlands have contributed to the rise of cultures being the basis for rice farming and the foundation of various religions (Ardana et al., 2024).

Wetlands are still hotspots of productivity and biodiversity today: they are home to life ranging from microbes to large mammals, including amphibians, fish and birds (Smardon, 2014). These ecosystems support such biological activity thanks to the many ecosystems functions they perform, which are summarised here in a brief overview, according to the classification of ecosystem services provided by the Millennium Ecosystem Assessment, 2005, which divides them into provisioning, regulating, cultural and supporting services. Starting from regulating services, wetlands have function of water purification: they act as natural filters (Irwin et al., 2018) physically, slowing down water flows and allowing sediments and particles to deposit, chemically (adsorption, chemical precipitation) and biologically (biodegradation) (Trepel, 2010). This leads to a reduction in nutrients such as phosphorus and nitrogen, through plants and microorganisms that also absorb pollutants such as nitrates up to

90% or heavy metals up to 60% (Shravani et al., 2025). Wetlands are also globally recognised for carbon sequestration and the resulting climate regulation. Stagnant water creates an anoxic condition that slows down organic decomposition, meaning that the carbon sequestered by plants is not immediately released into the atmosphere in the form of carbon dioxide but accumulates in the soil (Mitsch et al., 2013). This has the disadvantage, under certain conditions (Neubauer and Verhoeven, 2019), of methane emissions originating from the anaerobic decomposition of carbon, a greenhouse gas that is much more polluting than carbon dioxide in the long term, the balance of which is still being studied (Bridgham et al., 2013). In any case, wetlands, natural carbon reservoirs that occupy only 3% of the global surface area, store about one-third of the global carbon in the soil (Gorham, 1991; Maxwell et al., 2024), and in coastal environments it is now common to talk about “blue carbon” (Choudhary et al., 2024). In addition to carbon sequestration, the climate regulation of wetlands also occurs through evapotranspiration and temperature reduction (Zhang et al., 2022; Brazel et al., 2024). By absorbing excess water during extreme events through the soil and slowly releasing it, wetlands perform a flood control function reducing flood peaks (Segundo Leon et al., 2018; Peker et al., 2025) and thanks to their flow-slowing function, wetlands act as natural reservoirs for groundwater recharge favouring the regulation of the hydrological cycle (Wu et al., 2020; Nepal et al., 2024; Simon et al., 2023). Erosion control is also a particularly important function of wetlands in coastal areas: thanks to the vegetation typical of these ecosystems, they create a natural barrier to waves and currents, slowing down their flow (Taylor-Burns et al., 2023). On the other hand, root systems also hold soil and sediments together through friction, preventing soil loss, including that caused by wind (Gupta et al., 2025).

Provisioning services of wetlands relate to material products that can be extracted from these areas: freshwater and water for irrigation, food derived from fish or shellfish (Ghosh et al., 2024), construction materials (Salem and Mercer, 2012) and both renewable and non-renewable fuels (Telford, 2009), medicines (Stekovic et al., 2023). From the point of view of the cultural services provided by wetlands, these include recreation and tourism (Xu and He, 2022), the spiritual and identity value of the

area in which the wetland is located, in line with what has been described above regarding wetlands in the development of civilisations, and the aesthetic and inspirational value of these ecosystems (Tribot, 2018).

From the point of view of ecosystem services, in addition to the nutrient cycling and soil formation functions already mentioned among regulating and supporting services, one of the most important roles of wetlands is the provision of habitat for biodiversity. Although habitat provision is sometimes described in broad terms as underpinning provisioning benefits (for example by supporting commercially valuable species), major ecosystem service frameworks generally classify it as a supporting or regulating service, because it maintains the ecological conditions on which other services depend (Millennium Ecosystem Assessment, 2005).

According to U.S. Fish & Wildlife Service (2024) wetlands host almost 40% of all plant and animal species on the Earth. The biodiversity provided by wetlands, on the other hand, contributes to the stability of the other functions of the wetland itself in a virtuous circle, also improving the stability and resistance to disturbances of food chains through complex network structures. This biodiversity includes plant communities (Sonnier et al., 2022), microbial diversity that contributes to the nitrogen, phosphorus and carbon cycles and owes its presence to a wide variety of biotic and abiotic factors (Song et al., 2024), and finally, faunal diversity.

Within this fauna, particular mention should be made of amphibians, which are the animals that are best suited to live in an amphibious ecosystem, they would not be able to live in other ecosystems and are therefore among the most at risk (Demartin et al., 2024). Insects also account for 60% of the diversity found in wetlands (Liu et al., 2025). Globally, and particularly in the tropical areas, however, birds are the most important and emblematic group of fauna in wetlands, as they use these environments for nesting, foraging and, above all, as key stopovers on their migratory routes between continents (Deboelpaep et al., 2022).

Wetlands, therefore, due to the value of the services they provide, are increasingly considered Nature-based Solutions (NBS) for more rapidly achieving some of the Sustainable Development Goal indicators (SDGs), particularly those related to water,

biodiversity, climate and risk reduction (Seiffollahi-Aghmiuni et al., 2019). The notion of Nature-based Solutions (NBS) is relatively recent and still lacks a globally standardised definition, as different international frameworks emphasise distinct aspects of ecosystem functioning and societal benefits. In general, NBS are understood as measures that protect, sustainably manage or restore natural or modified ecosystems to address societal challenges in an effective and adaptive way while simultaneously providing benefits for biodiversity and human well-being (European Commission, 2015). The concept emerged from the increasing recognition that traditional grey infrastructures alone are insufficient to cope with climate change impacts, water scarcity, ecosystem degradation and disaster risks, and that ecosystem processes can often deliver more resilient, cost-effective and multifunctional solutions. Within this broader framework, wetlands are frequently cited as emblematic NBS because their intrinsic ecological functions, such as hydrological regulation, water purification, carbon storage and habitat provision, directly contribute to multiple policy objectives. Their ability to buffer floods and droughts, enhance biodiversity, improve water quality and support climate adaptation makes them particularly relevant in the NBS discourse and explains why they are increasingly integrated into sustainable water management and climate-resilience strategies.

Various attempts have been made to quantify the economic value of the services described above; the absence of a market price for most of them is precisely what motivates these efforts, which rely on indirect or non-market valuation approaches to attribute an economic value to these services. These estimates are made at both global and local levels and vary greatly depending on the historical period in which they are made, or geographical location, for example only in the United States these services are valued billions of dollars annually (TEEB Synthesis, 2010).

Among non-market valuation methods, the “replacement cost” approach is adopted to estimate the value of an ecosystem service from the cost of replacing it with an artificial or technological alternative. By contrast, the “hedonic pricing” method does not simply assess the value of real estate within an area; rather, it seeks to isolate the contribution of specific environmental attributes, such as proximity to wetlands or as-

sociated landscape and ecological benefits, to property prices by means of multiple regression analysis (Sander and Haight, 2012; Ignatyeva et al., 2022).

Although it is not possible to make a precise estimate, it has been supposed that wetlands have a value per hectare five times that of tropical forests, which are recognised as one of the most valuable ecosystems on Earth (Costanza et al., 1997; Costanza et al., 2014). Globally, the economic value attributed to wetland ecosystem services is estimated to represent roughly 45% of the total economic value of all ecosystem services, while other estimates put this percentage even higher. Among the ecosystem services with the highest economic value are flood control (De Groot et al., 2012), water purification and carbon sequestration (Stockmann et al., 2013). According to Ramsar Convention Secretariat (2018), approximately one in eight people on Earth depend on wetlands and their ecosystem services (particularly drinking water and food supplies) for their livelihood.

Although estimates may vary due to changes in definitions or measurement methods, the scientific community agrees that wetlands currently cover around 6% of the Earth's surface (Maltby and Barker, 2009; Finlayson et al., 2018) all over the World with the only exception of Antarctica. There is also unanimous agreement that wetlands are being lost at a very rapid rate, even three times faster than forests, according to WWF estimates, with a loss of 35% between 1970 and 2015 (Ramsar Convention Report, 2018) and a loss of over 87% since 1700 (Davidson, 2018) which translates into an extinction risk for 25% of all wetland species. Wetland losses in 20th century were almost three times faster than in previous centuries, with about 1% of remaining wetland destroyed each year. Although this trend is a cause for concern everywhere, such losses are more pronounced in developing countries due to soil changes for agriculture, unlike in developed regions where conservation measures are more active today. According to a study by Reis et al. (2017), which combines ground and satellite data, Asia, the region with the largest area of wetlands, has very low levels of wetland protection. The causes of this depletion will be listed and discussed below. The overview presented so far aims to show how, despite the unanimously recognised importance of these ecosystems, if the data on the decline and

disappearance of wetlands are equally unequivocal, this is due to their underestimation and underrepresentation in the policies of local and international agencies (Brander, 2006; Wang et al., 2025). It is useful to distinguish conceptually between the drivers that lead to wetland loss (i.e., conversion into other land uses) and those that cause wetland degradation (i.e., reductions in ecological quality, structural complexity and functional integrity). Although these processes are often interconnected, they act through different mechanisms, generate distinct sets of problems and call for differentiated management and policy measures.

The threats to wetlands are both recent and historical and include both direct human actions and indirect disturbances. First and foremost, the primary and earliest cause of wetland loss is drainage and conversion of land for “productive” uses. During the 20th century, many wetlands were drained to eradicate diseases such as malaria particularly in Mediterranean areas (Rizzetto, 2025). Following the Ramsar Convention framework, the drivers of wetland change are commonly distinguished into direct (or proximate) drivers such as drainage and land conversion, and indirect (or underlying) drivers, including demographic, economic and policy-related processes. Recognising this distinction is important, as the two groups of drivers operate at different scales and require different management approaches (Ramsar Convention Secretariat, 2018). Agricultural development is therefore cited as the main direct driver of such conversions, but it is in turn shaped by indirect drivers such as economic growth and population density (van Asselen, 2013). The soil that makes up wetlands is often suitable for agriculture, and farmers themselves received during centuries economic incentives to invest in these areas, creating a divergence between private and public interests (Cortus et al., 2011).

One cause of wetland loss that is perhaps rarely mentioned is deforestation: such a change in the land use, not necessarily for agricultural purposes, in neighbouring basins or directly in wetland areas causes erosion and eutrophication of downstream wetlands (Woodward et al., 2014).

Another significant cause of wetland disappearance is the construction of infrastructures that alter the hydrological regime of the territory. Such works include levees

(Knox et al., 2022) and water diversions for irrigation or regulation of watercourses by means of dams. A notable example is the Mesopotamian Marshes in Iraq, a wetland that was almost completely drained at the end of the last century following the construction of a dam. Dams' construction in particular cause a significant reduction in sediments in river deltas (Edmonds et al., 2023), which in turn leads to the impoverishment of coastal wetlands in particular (Rodriguez-Santalla and Navarro, 2021), made vulnerable by coastal erosion exacerbated by rising sea levels (Khalifa et al., 2024). Pumping systems also contribute to this problem: the depletion of groundwater due to overexploitation for agricultural purposes alters the natural flow of the watercourses that feed the wetlands (de Felipe et al., 2023), causing them to dry up. Pumping groundwater for coastal wetlands also leads to salinisation (Amores et al., 2013), another cause of wetland ecosystem degradation (Antonellini and Mollema, 2010).

The alteration of wetland hydrology is an issue for their survival, for the flora and fauna present there and sometimes even for the dependent human communities (Al-Mudaffar Fawzi et al., 2017).

Wetlands are increasingly affected not only by infrastructure that alters the hydrology of the area, as described above, but also by infrastructure such as roads and urban development in general, which disrupt the life of ecosystems because they are located on adjacent land, interrupting the continuity between wetlands or disturbing the well-being of local flora and fauna (Findlay and Bourdages, 2001).

Another well-documented threat to wetlands is pollution in various forms. Wetlands receive runoff from human activities upstream, which accumulate contaminants along the watercourse. In particular, this refers to nitrogen and phosphorus from agricultural fertilisers, which cause eutrophication of flora, weakening and decomposing roots (Deegan et al., 2012), livestock waste, pesticides (Relyea, 2005; Battaglin et al., 2009), and heavy metals from industrial activities. It has been said that wetlands have a water purification function, but in some cases, they are overloaded with nutrients, as in the case study of Chesapeake Bay in the United States (Kettenring et al., 2011). According to a UNESCO report (2017) about 80% of the world's wastewater, particu-

larly in developing countries and near highly urbanised centres, is released into water bodies such as wetlands, but these flows are not adequately treated. Pollution is now also entering wetlands through plastics (Ouyang et al., 2022) and microplastics (Li et al., 2024), which accumulate and degrade wetland habitats, ultimately contributing to the decline or local loss of many animal species, particularly waterbirds. This type of threat does not only affect wetlands in the strict sense but also affects the entire food chain linked to them.

As already described for changes in land use for agricultural purposes, another threat to wetland systems worldwide is the overexploitation of resources (Butt et al., 2021) for human consumption, such as overfishing and overhunting of wetland fauna, which take the form of destructive methods such as the use of dynamite or electric fishing, as well as the overexploitation for drinking water, which is also necessary for ecosystems (Tickner et al., 2020). A study has shown that lead from excessive hunting is a major pollutant in wetlands in some areas (Arcega-Cabrera et al., 2014).

Overharvesting of birds and mammals also impacts biodiversity, with the production of excessive waste that creates imbalances in resource use. It has long been known that peat mining for energy purposes has destroyed entire wetlands (Winkler and DeWitt, 1985). In most cases where such overexploitation occurs, the local population is unaware of the economic value of wetland systems, especially in developing areas, so sustainable conservation is not preferred to degradation in order to favour other types of activities.

Another ecological threat comes from other ecological ‘competitors’, namely invasive species that threaten the biological balance. Invasive species include both plants (Hovick et al., 2023), which proliferate in nutrient-rich wetlands due to pollution, and animals such as non-native animals and fish that are introduced into such environments for farming purposes, particularly in coastal wetlands. Invasive plants often form monocultures that alter habitat structures, reducing biodiversity (Zedler and Kercher, 2010) and modifying the structure of the wetlands themselves (D’lorio, 2008). Invasive animal species contribute to erosion and therefore to the loss of marshes (Carter et al., 1999) but also to the disappearance of the biodiversity of

other animal species present, as in the case of Burmese pythons in Everglades National Park, Florida (Dorcas et al., 2012). These phenomena are secondary effects of the other stresses mentioned above, such as pollution, changes in wetland hydrology, or even the presence of agricultural runoff, rich in fertilisers for the spread of invasive algae. Once established, these species are difficult to eradicate without targeted action.

In addition to the threats described above, there is the widely recognised threat of climate change (Salimi et al., 2021), which, while not a direct cause of wetland degradation, contributes to exacerbating the problems already mentioned. Changes in precipitation and temperature patterns directly alter the hydrology of wetlands, ecosystems that are highly dependent on climate for evapotranspiration processes (Mitsch et al., 2010), leading, for example, to frequent droughts (Sandi et al., 2020) or alterations in the hydroperiod, a fundamental characteristic of wetlands. Certainly, in some regions, those with arid or semi-arid climates, the impact of climate change is more pronounced (Wang et al., 2023). Even in areas that are already humid, where extreme weather events such as hurricanes or major floods have increased, there is a threat to the ecosystem (see the case of damage to coral reefs), partly because the increase in heavy rainfall contributes directly to the effects of pollution described above. Due to high temperatures and reduced rainfall, the increase in the number of fires (Williams-Jara et al., 2022), which, despite developing naturally in peat bogs and some plants being able to regrow quickly after a fire, damage most of them and therefore reduce biodiversity, including fauna, as animals are unable to survive without the sustenance and shelter provided by the surrounding vegetation. Added to this are the effects of reduced baseflow due to changes in soil permeability, increased water turbidity due to ash, and loss of peat, with years of storage being wiped out.

The effects of climate change are particularly pronounced in coastal wetlands, as they are also affected by sea level rise and salinisation. Sea level rise is a global phenomenon caused by the melting of glaciers, which affects wetlands through permanent flooding, leading to changes in dynamics and hydroperiod, as well as erosion. A study by Blankespoor et al. (2014) highlights how a 1 m rise in sea level has disas-

trous ecological and economic effects on approximately 64% of the world's coastal wetlands, drawing attention to the need to take urgent measures to stem the phenomenon. Other studies have shown that in the Mediterranean (Schuerch et al., 2025) and specifically in Nile Delta regions (Coleman et al., 2008), wetlands will tend to disappear completely by 2100 due to flooding.

The advance of the sea and inundations also causes salinisation, a phenomenon already analysed when discussing the overexploitation of groundwater. Salinisation leads to the extinction of freshwater species, as well as chemical alterations in the soil (Herbert et al., 2015).

The collapse of ecosystem health, which has been described several times in relation to the threats listed above, is itself a threat to the survival of wetlands. Projections for wetland biodiversity indicate extinction rates that would make this the sixth mass extinction in the history of the Earth. Some amphibian species or invasive species are also set to increase with changes in the hydroperiod or climate change in general, but for most reptiles and mammals, a sharp decline is expected (Bellard et al., 2012). This leads to reduced pollination and seed dispersal, disruption of food webs and a decrease in resilience to disturbances in the ecosystem itself (Song et al., 2024). For example, pollution or overfishing not only leads to a decrease in fish in wetlands, but also to a lack of food for birds, other fish and mammals that depend on those fish. This changes the routes taken by birds, with further effects on the loss of biodiversity (Carenton et al., 2024).

In summary, the threats to wetlands are interconnected and create a vicious circle: all the factors described above degrade wetlands and, at the same time, a degraded wetland is more sensitive to the effects of these factors and less able to counteract them. While it is now universally recognised through the 2018 Ramsar Convention on Wetlands that wetlands are the most threatened ecosystems in the world with the highest rates of species loss and extinction risk, it should also be noted that coastal wetlands are the most affected and threatened in this category (Kirwan and Megonigal, 2013): they face a dual threat, both from land, mainly due to anthropogenic effects, and from the sea, due to phenomena mainly attributed to climate change.

Whether directly or indirectly, all these pressures on wetlands originate from human activities. In fact, in the so-called Anthropocene, the current era characterised by the strong impact of humans on territorial, structural and climatic processes, water processes are not excluded from these transformations. Falknemark et al., 2019, analyse the concept of water resilience in the Anthropocene, emphasising how water is both a victim of change and an agent of transformation and a source of resilience, even if, once a certain stress threshold is exceeded, only a non-linear collapse of systems can be observed, with an irreversible change of state. This is what is happening today in many of the cases described, regardless of the nature or location of the wetlands (Day et al., 2024; Liu et al., 2021; Ning et al., 2024).

For this reason, given the irreversibility of these processes and the urgency of finding solutions, the main laws, actions, regulations, policies and other initiatives being taken globally and locally to defend ecosystems are hereafter outlined.

The historical evolution of wetland conservation and protection policies follows the evolution of wetland conservation awareness. The current state of wetlands is the result of centuries of unsustainable practices and little interest in these ecosystems. Today, the whole world is committed to international treaties, continental and national laws, and local initiatives, including the recent so-called Nature Restoration Law, i.e. the Regulation (EU) 2024/1991 of the European Parliament and of the Council of 24 June 2024.

Until the 1920s, wetlands were associated with unhealthy environments infested with malaria, limiting agriculture or, in any case, spaces that, if drained, could acquire value because they were otherwise unusable. The governments of various states themselves encouraged the drainage of wetlands: the Swamp Land Acts of 1800 in the United States (Bogue, 1951; Pintarich, 1980), which promoted agriculture, are well known. In Europe, during the same period and until about the middle of the 20th century, almost all states carried out massive wetland drainage projects, such as that of the Pontine Marshes, among the others in Italy during the Fascist era, to eliminate the problem of malaria and provide farmland for the population (Sevink et al., 2023). The

perception of governments and the people was therefore reflected in the laws, which themselves promoted the elimination of wetlands.

Scientific interest in wetlands sprouted in the mid-20th century, with the emergence of the first conservation groups following devastating floods, the first cases of species extinction and the loss of migratory routes following the drainage of certain areas. One of the first and most important initiatives that marked a turning point for wetland conservation was the 1971 Ramsar Convention on Wetlands, named from the Iranian city where the Convention was signed. This act is the first globally recognised environmental agreement, implemented primarily to protect bird migration routes and referring to the “wise use of wetlands”, i.e. their conscious management. For the first time, wetlands are recognised as ecosystems, and today Ramsar has more than 170 member countries, providing unanimous guidelines for the conservation of the World's most important wetlands. The sites included in the Convention are designated as “Wetlands of International Importance”. Among the most important Ramsar sites, we mention here the Okavango Delta in Botswana, the Camargue in France and the Everglades in the USA, sites scattered throughout the world and now for years at the centre of a rich scientific production. In the Mediterranean basin, we should mention the wetlands of Doñana in Spain and the Nile Delta in Egypt, whose membership of Ramsar is an indication of their global importance for biodiversity and conservation.

Following the Ramsar Convention, awareness of wetland conservation has grown in many countries and national policies have been implemented. Starting from 1970s, Asian countries such as Japan (Hamman, 2018) and India (Prasad et al., 2002) developed national programmes, Canada adopted in 1991 a Federal Policy on Wetland Conservation, while several European countries, including Italy (Montemaggiori, 1996), established a network of national reserves, albeit not recognised within the framework of the Ramsar Convention, with targeted conservation and protection measures. These policies and initiatives also involve more careful land-use planning, for example by establishing buffer zones around wetlands or limiting urban development in their vicinity.

The Ramsar Convention itself has also continued to evolve over time: although it does not have a legally binding regime but relies on the political will of the various member states, the guidelines issued by Ramsar for the protection of its sites have increased over time, as starting initiatives such as World Wetlands Day and collaboration in stating other international agreements (UNESCO World Heritage Convention, 1977; Berne Convention, 1981; Barcelona Convention, 1985 or Paris Convention, 1978). The Montreux Record has also been established, which is a list of Ramsar sites under threat on which national courts can take action, such as the EU Court of Justice enforcing Natura 2000 rules, or in Italy through voluntary River Contracts. In 2018, the Ramsar Convention released a very important report, the Global Wetland Outlook, which includes recommendations and observations on wetland trends and is considered an official and authoritative source in all its member states. A Special Edition was published in 2021 to mark the 50th anniversary of the Ramsar Convention, and an updated version will be presented in 2025.

The protection of wetlands is also included in the Sustainable Development Goals (SDG) adopted in 2015 by the United Nations in Agenda 2030. Of particular interest to these ecosystems are SDG 6 on clean water and sustainable use of water resources, SDG 13 on combating climate change, and SDG 15, which specifically mentions wetlands and biodiversity.

Many countries, although not with specific laws, have integrated wetland protection into environmental protection laws. In 1972, the Clean Water Act (CWA) in the United States imposed federal protection on wetlands through the regulation of discharges and pollutants. It also established that any alteration of a wetland must be authorised by the Army Corps of Engineers, effectively making it illegal to drain wetlands without adequate planning (Hough and Robertson, 2008). This mitigation law halted the degradation of wetlands in the USA and even encouraged the restoration of some damaged areas, as in the case of the Everglades (Erwin, 1990).

In Europe, the Birds Directive (Council of the European Communities, 1979) and Habitats Directive (Council of the European Communities, 1992) were the first laws to mention wetlands, both to protect migratory routes and to create an international net-

work of protected areas (Natura 2000 Network). The Water Framework Directive (European Parliament and Council, 2000), on the other hand, focused on water in general, indirectly included wetlands through directives on sustainable water management to ensure their good ecological status. In 2020, as part of the European Green Deal, the EU Biodiversity Strategy for 2030 (European Commission, 2020) was adopted with the aim of halting and reversing biodiversity loss in Europe. It mentions wetlands but, above all, it inspired the EU Nature Restoration Law (European Parliament and Council, 2024), that explicitly mentions wetlands as ecosystems to be protected, as well as restored where they have already been degraded. The Regulation (EU) 2023/1115 (European Parliament and Council, 2023) on deforestation-free supply chains, in the context of recognising the relevance of non-forest natural ecosystems evaluates the possible extension of its scope to include wetlands, peatlands and other high-biodiversity ecosystems.

In Australia, the Environment Protection and Biodiversity Conservation Act (Commonwealth of Australia, 1999) also focused on wetlands (particularly those identified by Ramsar Convention) as well as on mapping migratory species to ensure suitable resting places along their routes. Recently, China, which is among the world's leading countries in terms of wetland loss, also passed the Wetland Conservation Law (National People's Congress of the PRC, 2022), its first law explicitly including these ecosystems. Significant policies have also been developed recently in several African countries, such as South Africa and Uganda, where the sustainable use of wetlands is addressed through indigenous community management policies (Dini and Everard, 2016).

Other global cooperation initiatives include the Convention on Biological Diversity (CBD) (United Nations, 1992), and its recent Kunming-Montreal Global Biodiversity Framework (GBF) (Convention on Biological Diversity, 2022), which sets global targets for the conservation and restoration of ecosystems, including wetlands; the UN Framework Convention on Climate Change (United Nations, 1992), in which wetlands are repeatedly mentioned as carbon storage sites, as well as environments to be protected; and the Sendai Framework for Disaster Risk Reduction (United Nations Office

for Disaster Risk Reduction, UNDRR, 2015), which considers wetlands to be barriers against natural disasters. In the Mediterranean context, a Ramsar satellite network, MedWet, aims to protect wetlands by safeguarding their cultural aspects, as well as leveraging issues dear to the context, such as water scarcity and tourism in relation to wetlands. Organisations such as WWF and BirdLife have also increasingly focused on restoration projects and raising public awareness over time. Policy approaches have shifted from specific measures to a holistic approach, such as the Water Energy Food Ecosystem (WEFE) Nexus or the concept of Integrated Water Resources Management (IWRM). From a regulatory point of view, there is clearly a change in mentality and sensitivity towards wetlands, both through specific laws and through additions to laws for the protection of the environment, the territory or water, even if conservation and, above all, restoration are difficult to implement in practice, especially in contexts that are already heavily degraded.

This is often due to limited monitoring capacity, driven not only by financial constraints but also by technological limitations. Among the most common actions undertaken in the field of wetland restoration are the re-flooding of areas that were previously drained by removing drainage systems, the reconnection of tributaries to rivers or rivers to wetlands by removing embankments (Knox et al., 2022), effectively restoring natural hydrology. In some cases, it is necessary to restore the natural geomorphological processes of an area, and to this end, sediment is replenished (Piercy et al., 2023) to counteract sea level rise or subsidence phenomena. Living shorelines are a NBS implemented for geomorphology and habitat restoration (Bilkovic et al., 2025).

Another widespread restoration technique is the planting of native plant species, such as rushes or mangrove plants in tropical environments, eradicating invasive species by mowing or mechanical removal of plants. This process of soil stabilisation and biodiversity restoration requires a lot of time and management of invasive species (Reinhardt Adams et al., 2025). Attempts are being made to remove sources of pollution in the soil and water of wetlands, phytoremediation and the implementation of

sustainable agricultural practices in basins: although these processes appear to be useful, they also require time and planning.

Certainly, all of the above-mentioned actions have already led to some positive cases (Ioana-Toroimac, 2024), which give us confidence in this policy, legislative, social, scientific and action trend, however, it should be noted that these processes, whether they involve planning to protect the territory or redeveloping areas, are complex challenges involving multidisciplinary approaches to predict the effectiveness of proposed actions. In this context, the development and implementation of mathematical, statistical and simulation models is increasingly pursued for quantitative investigations. For example, when implementing a restoration or management action in a wetland area, it is necessary to predict and quantify its effects not only in the present but also, and above all, in the long term, including under variable conditions, for example due to climate change.

In this regard, the concept of environmental or ecological flow for river systems has been developed and studied for years in a consolidated manner, being now a cornerstone of hydrological and ecological water management (Acreman and Dunbar, 2004). This concept helps to define quantity, timing and quality of water flows required to sustain freshwater and estuarine ecosystems. However, applications of environmental flows to wetlands remains limited and poorly formalized, despite their emerging ecological relevance: this depends on difficulties in defining a single environmental flow value or regime for these contexts highly influenced by multiple sources. As noted by the MedWet technical report on environmental water requirements (2020) existing methodologies for environmental flow assessment, largely developed for fluvial systems, are rarely adapted to the conditions of wetlands, especially temporary or groundwater-fed ones, which are highly sensitive to variations in water availability. Consequently, the establishment of environmental water requirements for wetlands is still in its infancy, representing a critical research and management gap in the context of integrated water resources planning.

There are already many types of models designed or used for these purposes, ranging from ecology to hydrology and economics, and the choice of model to use is also

delicate (Twomey et al., 2024), depending on the objectives to be achieved and the data and resources available (time, budget). Hydrological modelling, among other things, is still a relatively unexplored topic in the field of wetlands due to the difficulties in finding data and schematising processes, but it is strictly necessary for quantifying flows and organising the actions to be taken at both the regulatory and implementation levels.

Within this context, this work therefore aims to investigate the potential and limitations of implementing a simple hydrological model for unravel hydrological regimes of a wetland in order to analyze conservation and restoration scenarios for its sustainable management, answering the following research questions: i) is it feasible to characterise and subsequently replicate the key hydrological attributes and regimes of wetland hydrology, particularly in complex contexts characterised by strong intra- and inter-annual variability such as the Mediterranean wetlands, through a parsimonious hydrological model? ii) can a hydrological model provide useful output for the sustainable management (in terms of decision support and planning) of a wetland subject to the anthropogenic and climatic pressures mentioned in this Introduction? iii) If so, would it also be possible to easily scale this model, making it suitable for addressing different challenges related to various geographical and hydrological contexts?

The work is structured as follows. Chapter 1 provides an overview of the state of the art in wetland modelling, first classifying the various types of wetlands and then the models that have been implemented to study and manage them, identifying the context in which this work is placed and the reasons for its implementation.

Chapter 2 describes the implementation of a hydrological model (WetMAT model) in a specific hydrological context, in order to test its basic processes and the validity of its outputs: specifically, the model is used for monitoring flooded areas of the Doñana wetland in Spain, one of the largest and most important wetlands in Europe. A synthetic index of hydrological response is proposed and analysed, specifically developed for temporary wetlands, with the aim of characterising their hydrological regimes and estimating associated water requirements. Chapter 3 shows the attempt to adapt the same modelling approach in another case study in southern Italy, very dif-

ferent from Doñana in terms of extension and features, where water salinity of wetland ponds added additional complexity to the modeled hydrological processes.

In the Discussion and Conclusions sections, results from the two model implementations will be compared and commented, highlighting the limitations and possible future developments of the used approach.

1. STATE OF THE ART IN WETLAND MODELLING

This chapter provides an overview of the current state of wetland modelling. Section 1.1 introduces the different types of wetlands and outlines their classification according to specific characteristics. Section 1.2 presents the main modelling approaches applied to wetland systems. Section 1.3 examines the key objectives of wetland modelling, while Section 1.4 discusses the current challenges and limitations associated with these approaches. Finally, an overview of the WetMAT model, with its water balance framework, is proposed (Section 1.5) and its peculiarities within the context of the wetland modelling is analysed.

1.1 *Wetland Classifications and types*

The Technical Report 12 of the Ramsar Convention, (Ramsar Convention Secretariat, 2024) states that although wetlands cover only about 6% (Mitsch and Gosselink, 2015) of the Earth's surface, they are home to about 40% of all known plants and species in the world. This is because they belong to different types and have several characteristics, appearing under the most diverse hydrological, geomorphological, climatic and biological conditions. For this reason, there are various classification based on the characteristics taken into consideration, and the main ones will be analysed here.

A widely used classification, particularly in the United States, is that of Cowardin et al. (1979), which is ecological but primarily defined by five major wetland systems based on a differentiation by landscape and further subdivisions based on hydrology: Marine and Estuarine (coastal wetlands such as estuaries, mangroves, coastal marshes), Lacustrine (lake-associated wetlands), Riverine (river floodplains) and Palustrine (inland freshwater wetlands such as marshes or swamps, bogs and fens).

In this type of classification, once wetlands have been divided into subsystems, further characteristics are specified, mainly regarding their hydrological nature, which are often used to describe a wetland in greater detail. These hydrological characteristics refer to the permanence of water in the system throughout the year (permanent or

temporary/seasonal wetlands), to the source of the water (surface water-fed or groundwater-fed wetlands, the latter being fed by aquifers or springs) or to the dynamics of the flow within them (stagnant, such as marshes, slow-flowing, such as floodplains, or fast-flowing, such as river wetlands). To describe a wetland within this classification, reference is often made to the dominant vegetation (e.g. forested swamp or herbaceous marsh). As mentioned above, the Cowardin classification is ultimately ecological in focus: marshes, environments that support biodiversity, are defined as characterised by emergent soft-stem vegetation and frequent inundation, swamps are wetlands with woody vegetation and standing water, bogs are environments of acidic and rain-fed peat accumulation, while fens, also wetlands formed by peat accumulation and notable for carbon sequestration, receive groundwater inputs. In 1990, the Ramsar Convention itself devised an official classification system to facilitate uniformity of concepts and terms when referring to the multitude of existing wetlands. Ramsar classification divides wetlands into three main categories: marine and coastal wetlands, inland wetlands, which may also be saline but are not influenced by tidal dynamics as coastal wetlands, and artificial wetlands. These three macro categories contain a total of 42 types of wetlands, each uniquely identified by a code that specifies the characteristics listed above, such as ecological indication, water permanence, extent and, in the case of artificial wetlands, their function. The explicit definition of constructed wetlands, emphasising their importance throughout the world, is a distinctive feature of the Ramsar classification.

A classification that is completely different from those described above, which are more descriptive, is the Brinson's hydrogeomorphic classification (HGM) (Brinson, 1993), used by the US Army Corps of Engineers with the aim not of cataloguing wetlands but of associating them according to their functional status (in terms of ecosystem services). HGM classifies wetlands into seven main classes based on three criteria: the geomorphic environment, i.e. the shape of the terrain and its topographical position in the landscape; the primary water source feeding the system (direct precipitation, surface runoff or groundwater); and the hydrodynamics of the wetland, i.e. the movement, direction and energy level of the water within it. The seven classes are

therefore Depressional (isolated basins such as prairie potholes), Riverine, Mineral Flats, Organic Flats, Tidal Fringe, Lacustrine Fringe and Slopes (on slopes with groundwater discharge as fens) and characterise a wetland functionally. For example, a wetland classified as “palustrine” in the Cowardin system due to its dominant vegetation, is defined in HGM as “riverine” as it mainly considers the hydrological interaction of the wetland with the river channel.

Apart from these major official classification systems, wetlands are often differentiated based on their hydroperiod, or flooding regime (ephemeral, temporary, seasonal vs. permanent wetland), as in the case of Robinson et al. (2024), who identified Swedish wetlands based on their regime and surface dynamics (‘spring surging’ as opposed to ‘summer flooded’). In general, it can be said that hydrology is considered the only universal basis for hierarchical classification: different and broader classifications, such as that of Junk (2024), still have subcategories that refer to hydrological characterisation.

In the Mediterranean basin, wetlands exhibit distinct climate-related characteristics, often forming only during the wet season or in particularly humid hydrological years. To capture these specificities, classification frameworks such as the MedWet Habitat Description System (Farinha et al., 2005), similar to the Cowardin system, prove especially useful, as they incorporate modifiers related to hydrological regime and water salinity. Moreover, since remote sensing provides powerful means to monitor these dynamic ecosystems, the MedWet system has been explicitly designed to facilitate the classification of Mediterranean wetlands through such methodologies.

Sometimes certain types of wetlands fall under other classifications that are not specifically designed for wetlands. The European Nature Information System (EUNIS)(Davies et al., 2004) has been adopted in Europe: designed to support environmental directives such as the Habitats Directive and the Birds Directive, this system classifies all types of habitats, including wetlands, which include “mires, bogs and fens”, environments that are flooded for at least half of the year.

As we have seen, there are many types of wetlands, and there are also many classification systems based on the perspective observed. Ecologically, we can refer to the

dominant vegetation, hydrologically we can refer to the water source or hydroperiod, and it is also useful to classify a context based on its ecosystem function. Each of these systems has its own critical issues and limitations: while the Ramsar classification, although inclusive, seems to consider only ecological factors, Cowardin's classification uses quantitative thresholds to distinguish between systems, which seems limiting in view of other ecological characteristics. Ultimately, although numerous approaches exist, they are not redundant; rather, each addresses specific needs, as will become evident in the following discussion on models implemented for wetland studies.

1.2 Wetland modelling approaches

Starting from the mid-20th century, with the growing recognition of the ecological significance of wetlands and the consequent need for their systematic study and conservation, modelling approaches tailored to these ecosystems began to emerge as fundamental tools for managing these ecosystems whose hydrological and ecological dynamics are inherently complex and highly variable in time and space. The need for modelling stems both from scientific and practical concerns: on the one hand, models allow researchers to formulate and test hypotheses on wetland functioning; on the other, they provide decision-makers with predictive tools to guide conservation, restoration, and management efforts. In this respect, modelling can be seen as a cycle of iterative learning rather than a final product, encompassing the development of perceptual models, their formalisation into conceptual or procedural structures, comparison with empirical data, and the revision of hypotheses (Beven, 2012). Over time, these approaches have evolved along a continuum of complexity, ranging from simple empirical formulations to highly sophisticated and integrated modelling frameworks, following the broader trajectory of hydrological modelling and moving from fixed approaches to more flexible ones. Historically, simple water balance models were introduced to quantify storage, inflows, outflows. The expansion of research questions, ranging from flood mitigation and nutrient removal to habitat provision and climate regulation, led to a proliferation of different approaches: no single model type

has proved universally suitable because the appropriate choice always depends on the dominant process of interest, availability of data and purpose of application (Fenicia and McDonnell, 2022). Modelling is a way to bridge gaps in empirical knowledge, integrating limited observations with theoretical representations of processes and in this sense, they serve not only as predictive instruments but also as heuristic devices, allowing to explore hypotheses on wetland behaviour under both current and future conditions. In order to provide an overview of wetland modelling approaches, these can be broadly classified according to the way they represent processes or their temporal and spatial heterogeneity, with particular reference to the typologies commonly adopted in hydrological modelling, the main focus of this work. Each category presents specific strengths and limitations; consequently, the selection of an appropriate approach is contingent upon data availability, the processes of interest, and, most importantly, the overarching objectives of the modelling exercise (Pandi et al., 2021). Empirical models are based on statistical relationships derived from observed data, with little or no direct representation of the underlying physical processes. For this reason, they are often referred to as *black-box approaches*, as the focus lies on system outputs rather than on the internal mechanisms driving them (Stella and Warner, 2020). In such models, the links between input variables (e.g., climatic drivers) and output variables (e.g., wetland water levels or surface outflows) are established exclusively through statistical regression of extensive datasets. The availability of long-term observational records is therefore crucial to identify patterns and derive reliable empirical relationships. A notable advantage of these approaches is that they typically require only a limited set of parameters, which makes them particularly suitable for preliminary assessments or as sub-components within more comprehensive modelling frameworks. For instance, within hydrological models, empirical formulations are frequently employed to estimate processes such as infiltration or evapotranspiration (Vishwakarma et al., 2024). The earliest empirical models in this field focused on estimating the influence of specific catchment characteristics, such as wetland extent, on output variables like streamflow (Johnston et al., 1990). Empirical approaches are also frequently employed in wetland studies to simulate pollutant removal processes,

where contaminant attenuation is commonly represented using first-order kinetic models. These models, particularly applied in the context of constructed wetlands, are effective in capturing the exponential decay of pollutant concentrations over time. Nevertheless, due to their reliance on site-specific conditions and limited data requirements, such models are constrained in their general applicability, and more physically based modelling approaches are generally preferable when available (Wei et al., 2025). An emerging and increasingly sophisticated strand of empirical modelling is represented by approaches based on Artificial Neural Networks (ANN). Applications of ANN in wetland research have expanded considerably in recent years, addressing diverse objectives. Early studies concentrated on modelling water purification processes (Song et al., 2013; Amanollahi et al., 2017). In parallel, subsequent contributions extended their use to the estimation of wetland extent and depth, often relying on hydrological input data such as observed flow records (Rezaeianzadeh et al., 2015; Saha et al., 2021). More recently, ANN-based models have been adopted for wetland identification and classification (Talukdar et al., 2021; Hu et al., 2021) and have also proven useful in predicting the treatment efficiency of constructed wetlands (Singh et al., 2011). Notably, ANN are now being explored as decision-support tools, for instance in the prioritization of restoration areas across large wetland landscapes (Maleki et al., 2020). In any case, even for the purposes for which they are most commonly used, empirical models in wetland modelling are mostly used in combination with process-based models (Noori et al., 2020).

Conceptual models constitute an intermediate level of representation, in which the hydrological system is described as a set of interconnected elements that reproduce real key processes as rainfall-runoff, evapotranspiration, infiltration in a simplified manner, for instance through the abstraction of reservoirs and flows. The parameters employed in these models retain a physical meaning, yet they are generally estimated through calibration rather than direct measurement. Owing to this semi-empirical character, conceptual models are often referred to as *grey-box models* (Devia et al., 2015). The development of conceptual modelling in hydrology can be traced back to the mid-20th century, with one of the earliest and most influential examples being the

Stanford Watershed Model IV (SWM), introduced by Crawford and Linsley in 1966. Among the main advantages of conceptual approaches are their computational efficiency, which facilitates the simultaneous exploration of multiple scenarios (Kavetski and Fenicia, 2011), their relatively modest data requirements for calibration and validation, and their ability to capture essential hydrological processes across different spatial and temporal scales. Nevertheless, the application of conceptual models is less suitable in contexts where low structural and parametric uncertainty is required, or where issues related to equifinality and non-uniqueness of model results must be avoided. In this category, lumped conceptual models schematise the basin, or wetland, as a single unit (Mackay et al., 2014), while there are also semi-distributed models in which the study area is divided into sub-units, albeit not according to a very fine grid. In both cases, wetlands are treated as one or a series of reservoirs. One of the first models in this category, according to Bengtson (1999), was developed by Ogawa and Male that used the lumped rainfall-runoff model HEC-1 as early as 1986 to model wetlands as reservoirs and control runoff flows through channels modelled in HEC-2, in a study of flood mitigation peaks.

Within the family of lumped conceptual models, the Hydrologiska Byråns Vattenbalansavdelning (HBV) model, originally developed in Sweden to simulate snow storage and melt processes, is particularly emblematic. Its structure has proven versatile for long-term water balance studies and can therefore be applied to wetlands, which also act as water storage systems (Bergström, 1992). The majority of conceptual models, however, are semi-distributed. Among these, the Soil and Water Assessment Tool (SWAT) is one of the most widely used: it divides a basin into sub-basins and hydrological response units (HRU) defined by the homogeneity of physical characteristics. Wetlands can be represented as individual HRU, although specific extension modules have been developed to better capture their complex interactions (Rahman et al., 2016). Another example is the Hydrological Simulation Program, FORTAN (HSPF), which is capable of simulating basin-scale dynamics, including wetlands, and has been applied for estimating pollutant transport in rice-producing wetlands (Jeon et al., 2007). Similarly, the Hydrologic Engineering Center's Hydrologic Model-

ling System (HEC-HMS), developed by the U.S. Army Corps of Engineers, is a semi-distributed conceptual model widely employed for flood forecasting and runoff management. Nevertheless, it provides only a simplified representation of wetlands, treating them essentially as storage reservoirs (Tang et al., 2020). Conceptual models that have been successfully applied to wetlands include DRAINMOD, which is widely used in agricultural contexts and for the management of wetlands with shallow aquifers, proving particularly valuable for detailed field-scale analyses (Petru et al., 2014). Another relevant example interesting for wetlands is TOPMODEL (Zhang et al., 2016), which is especially suited to floodplain environments, as it effectively represents the subsurface flow component sustaining these systems (Xi et al., 2022). Conceptual approaches have also been employed for highly specific wetland applications, such as the case of cutover peatlands, where volume-change dynamics play a key role in hydrological functioning (Kennedy and Price, 2005).

In summary, conceptual models occupy an intermediate position between empirical and physically-based models: they incorporate the most relevant hydrological processes while simplifying their mathematical representation. Moreover, they are able to capture essential spatial heterogeneity without imposing excessive demands in terms of data availability or computational resources.

A further step in modelling complexity is represented by physically based models, which explicitly simulate the underlying physical processes and, in almost all cases, adopt a distributed parameterisation. These models solve the fundamental conservation equations of mass and energy governing water flows, and in some cases also account for sediment and solute transport. Owing to their mechanistic formulation, they are often referred to as *white-box models*. Their implementation, however, entails substantial computational demands and requires extensive site-specific data, as well as significant expert knowledge. Moreover, they are subject to well-known challenges related to spatial and temporal scaling. When correctly implemented, their high level of detail enables them to provide valuable insights into multiple characteristics of a catchment system. Many physically based models concentrate on hydrodynamic processes: well-established examples for surface water include MIKE SHE, MIKE 11,

HEC-RAS 2D, and ANUGA, frequently applied for purposes such as floodplain inundation mapping (Vaughn, 2019). Other models, such as HYDROGEOSPHERE and MODFLOW (the latter with specific wetland-related packages), are designed for groundwater and saturated zones, supporting advanced and integrated simulations of wetland dynamics, including nutrient cycling and biogeochemical processes (Faruque Abesh et al., 2024).

In wetland applications, physically based models often require the solution of highly complex equations, such as the Richards equation for simulating soil moisture or the Saint-Venant equations for representing surface water flow in cell-based schematisations. As a result, they are generally applied in specific contexts, such as the design of constructed wetlands (Wei et al., 2025) or in large and ecologically significant case studies, including the Everglades in Florida, the Okavango Delta in Botswana (Bauer et al., 2006), and the Niger Inner Delta in Mali. These examples share the characteristic of being extensive wetland systems with intricate flooding dynamics that require detailed spatial representation. Such models demand highly specific input data, including high-resolution digital elevation models (DEM), vegetation parameters, and multiple boundary conditions, such as long-term runoff and climatological series. Recent advances have enabled the replacement of some of these inputs with remote sensing products, for instance, DEM or evapotranspiration estimates (Di Vittorio and Georgakakos, 2021), although these datasets are not always accessible or easily processed. Consequently, the challenges associated with data availability and quality remain a major limitation for the calibration and validation of distributed models.

Due to their physically based formulation, these models can be applied to predict wetland responses under conditions different from those used for calibration and validation, since their functioning is not exclusively dependent on calibration. Their distributed nature also enables the representation of spatial heterogeneity, which is particularly valuable in large and complex wetland systems. Nevertheless, the use of a physically based model does not guarantee accuracy: parametric uncertainty tends to increase with the number of parameters, which in turn can lead to substantial overall

uncertainty. For this reason, the application of such models is most appropriate when a detailed, fine scale understanding of wetland processes is specifically required.

Figure 1 presents a timeline of rainfall–runoff modelling developments, the broader category of hydrological models, from Peel and McMahon (2020). While this figure refers to a broader context than wetland-specific modelling, it is noteworthy that the historical trajectory of rainfall–runoff models closely parallels the evolution just described. In both cases, the discussed model complexity has progressively increased over time, evolving from early empirical formulations to physically based approaches and, more recently, to ANN models, which themselves can be regarded as an evolution of the original empirical methods often employed in support of other modelling frameworks.

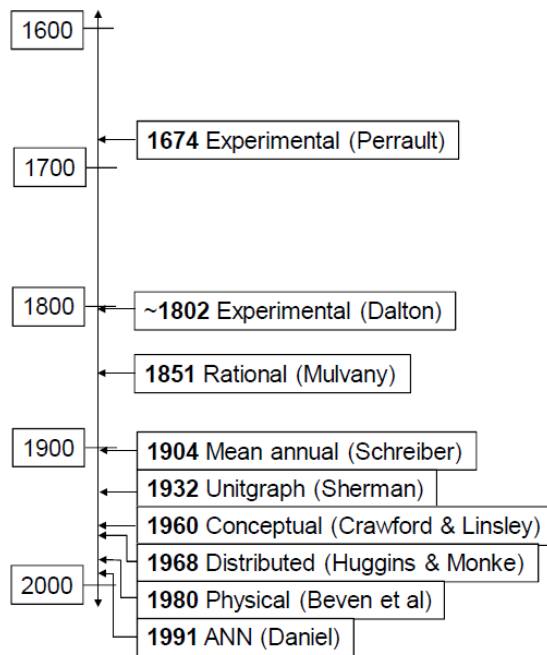


Fig. 1 – Temporal evolution of rainfall–runoff hydrological models (Peel and McMahon, 2020) under copyright). Although referring to a broader class of models than those specifically applied to wetlands, the figure illustrates the general trajectory of increasing model complexity that also characterises wetland modelling

A further category of wetland models, representing a step beyond those described thus far and increasingly widespread in recent years, is that of ecohydrological or, more generally, integrated models. While these models are often physically based, they are distinguished by their broader scope, as they not only simulate hydrological processes but also incorporate vegetation dynamics, nutrient cycles, and other ecological components. Particular attention has been devoted to peatland models, which explicitly consider carbon accumulation in addition to wetland extent (Kleinen et al., 2012). Such approaches allow for the coupling of water quality with nutrient dynamics, or the integration of habitat tracking modules within hydrodynamic models. A notable example is provided by Chui et al. (2011), who developed a hydrological model based on Richards' equations for variably saturated groundwater flow in the Everglades, subsequently coupling it with Lotka–Volterra type equations to simulate vegetation dynamics. Other integrated models place greater emphasis on the representation of evapotranspiration, which in simpler hydrological models is typically assumed to be constant.

Integrated models for wetlands frequently place emphasis on water quality, particularly in the context of designing constructed wetlands, which (as previously discussed) represent a central focus in wetland modelling. A prominent example is provided by the CW2D module of the HYDRUS software, which has been widely applied to simulate water flow in constructed wetlands while simultaneously modelling microbial processes responsible for the removal of carbon, nitrogen, and phosphorus (Pal et al., 2024). More recently, integrated models have also been employed for climate-related assessments, incorporating modules for wetland extent and greenhouse gas emissions; the ORCHIDEE-WET model, developed specifically to predict methane fluxes in wetlands, is a notable case in point (Ringeval et al., 2013). The strength of such models lies in their capacity to evaluate multiple aspects of wetland functioning simultaneously. Nevertheless, practical applications achieving this level of integration remain relatively rare due to the inherent complexity of the models and their extensive data requirements. Furthermore, the strong interdependencies among modules often complicate calibration, validation, and uncertainty analysis. In the domain of ecologi-

cal restoration, however, the adoption of such integrated approaches is becoming increasingly indispensable for achieving holistic management of wetland ecosystems (Zhou et al., 2016).

Figure 2 highlights the progression of model typologies from the simplest to the most complex, indicating the key elements introduced at each stage.

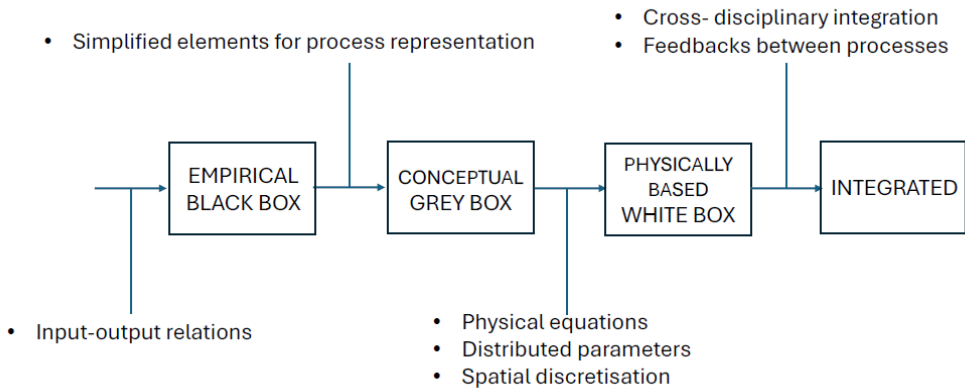


Fig. 2 – Evolution of wetland modelling approaches.

1.2.1 Spatial and temporal scales in wetland modelling

In addition to the main modelling approaches discussed, careful consideration must be given to the spatial and temporal scales at which models are implemented. These scales are not merely a matter of resolution but strongly influence the type and accuracy of processes that can be represented. On the one hand, the choice of scale is driven by the objectives of the modelling exercise, on the other hand it also depends on the modelling framework adopted. Finally, the definition of scale is inherently constrained by data availability, since the capacity to simulate processes at high resolution relies on the existence of adequate observational datasets. The interaction among these factors highlights the importance of selecting an appropriate scale in order to ensure both the reliability and the interpretability of model results. From a spatial perspective, wetland models span scales ranging from point-based applications,

or those focusing on a single wetland (on the order of metres to hectares), to regional and even global frameworks encompassing thousands of square kilometres. Similarly, the temporal dimension varies considerably: some models are designed to simulate individual hydrological events lasting hours or days, whereas others are applied to seasonal dynamics, multi-year periods, or even decadal trajectories. The following section provides examples of these variations and illustrate their implications for modelling practice.

Local-scale models typically focus on a single wetland when it is relatively small, or on specific portions of a system when the wetland extends over a larger area. These models are characterised by a high level of spatial detail, often incorporating microscale topographic variation or vegetation structure, with resolutions that can reach the centimetre scale. Such fine detail, however, comes at the cost of substantial computational demands and extensive data requirements. Local-scale approaches are particularly well suited for analysing vertical balance components, where detailed representation is essential. As noted above, a key application that necessitates this level of resolution is the study of pollutant treatment processes, which are often the primary objective in the design and management of constructed wetlands (Brisson et al., 2024).

Watershed-scale models typically encompass larger wetlands or multiple wetland systems, covering areas on the order of tens to hundreds of square kilometres. At this scale, modelling approaches often shift from physically based formulations to lumped or semi-distributed structures (Evenson et al., 2018; Liu et al., 2008). The focus is placed on capturing interactions among system components, such as ponds, rivers, groundwater, and in some cases coastal exchanges. While fine-scale detail is generally unattainable at this resolution, basin-scale models are well suited to implementing water balance assessments that support a wide range of hydrological and management applications.

At an even broader spatial scale, wetland modelling includes regional and continental applications. These models are highly aggregated, covering extensive and heterogeneous areas. They typically operate through grid cells, often 10 km or larger, but at-

tempts are still being made to represent wetlands, for example phenomena involving coastal ones (Zhang et al., 2018). At the global scale (the widest spatial domain) wetlands are generally characterised by generic data, such as approximate values of soil moisture. Although wetlands are sometimes considered within global applications (Mc Fadden et al., 2007), hydrological processes are usually represented at finer spatial scales to capture their dynamics more realistically.

In recent years, increasing efforts have been devoted to applying the same modelling frameworks across multiple spatial scales, from local to global, particularly in the context of climate change evaluation and other global drivers that are ultimately expressed at fine scales, e.g. carbon storage (Zhang et al., 2021; Bridgham et al., 2012) or the management of ecosystem services delivered by wetlands (Janse et al., 2019). Nonetheless, this remains a major challenge, as most wetland models are not inherently transferable across scales. Key parameters, such as those related to soil moisture or vegetation dynamics, are difficult to upscale when an entire continent is represented by only a few grid cells. One approach to addressing this issue is nested modelling (Wang et al., 2018), whereby outputs from a global model (such as climate projections or sea-level rise scenarios) serve as boundary conditions for a higher-resolution model focusing on a specific wetland.

From a temporal perspective, the situation closely mirrors that observed for spatial scales. Event-based models are frequently applied in flood studies, where wetlands play a crucial role in mitigating storm peak damage (Haque et al., 2022). These models, which emphasise immediate hydrological responses, typically employ hourly or daily data and tend to overlook longer-term climatic drivers, such as evapotranspiration trends. Other models focus on extended periods (ranging from months to years) and therefore incorporate past dynamics, seasonal variability, and interannual fluctuations. Conceptual models are most often applied at this scale, operating with daily, monthly, or annual inputs. This temporal resolution is particularly relevant for wetlands, as it enables assessment of hydroperiod characteristics, which are critical for ecological considerations such as avifaunal habitat requirements and water balance studies. At the longest timescales, models addressing the impacts of climate change

on wetlands, as well as those designed to support sustainable and holistic management, typically operate on multi-decadal horizons (Bennion et al., 2024; Kern et al., 2012).

As from the other modelling characteristics discussed above, the choice of the temporal scale must be guided not only by data availability but also by the intended purpose of the model. Biogeochemical processes such as carbon sequestration unfold over centennial timescales, and peatland models therefore require very long simulation horizons. By contrast, water quality dynamics are often driven by processes such as fluctuations in dissolved oxygen concentrations, which demand sub-hourly resolution. Consequently, constructed wetlands designed for water treatment typically require very short temporal scales, reflecting a situation analogous to that already noted for spatial scale considerations. It is therefore essential to carefully select the appropriate scales (or, where necessary, to employ nested models) in order to represent each process at the scale most suitable for its dynamics. This approach helps to minimise uncertainty and to avoid generating overly detailed or costly estimates that are not relevant to the broader context under investigation.

The spider diagram in Figure 3 provides a qualitative synthesis of the main characteristics of the wetland models discussed, based on the reviewed literature. For spatial and temporal scales, the assigned scores increase with the extent of the scales considered. As illustrated, the most comprehensive models also exhibit the highest levels of complexity.

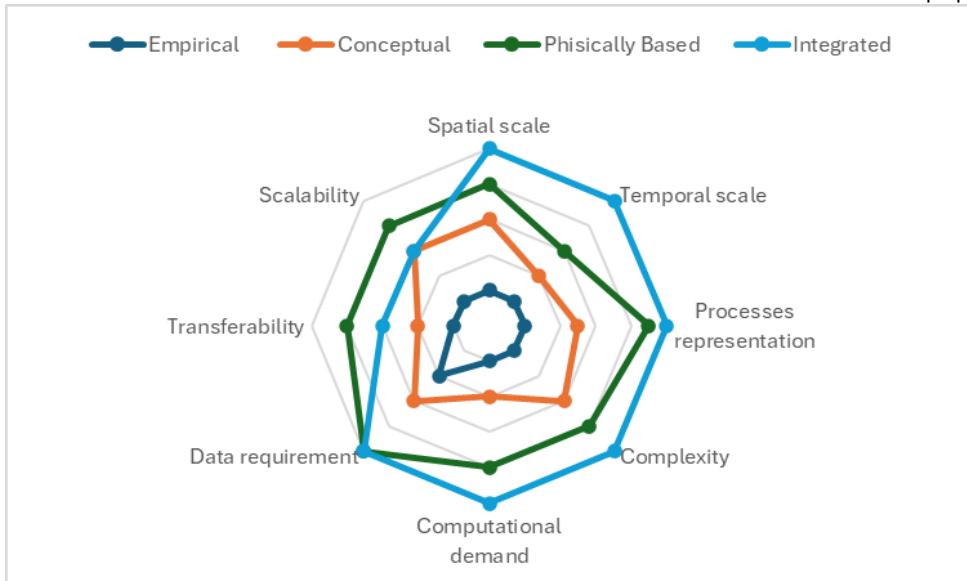


Fig. 3 – Comparison between wetland model types based on key characteristics.

1.3 Main objectives in wetland modelling

As already seen in the Introduction, wetlands are ecosystems of increasingly crucial importance in different domains. Consequently, the purposes for which wetland modelling is implemented are diverse and reflect this multifunctionality. In the literature, modelling applications typically focus on hydrological regulation (e.g. flood attenuation and groundwater interactions), water quality improvement (including nutrient and pollutant removal), carbon and greenhouse gas dynamics, and support for management and restoration strategies (Mitsch & Gosselink, 2015; Mitsch et al., 2013). The following section summarises the main modelling application objectives most frequently addressed in the scientific literature, providing a framework for understanding the different directions in which wetland modelling has evolved.

Wetlands are often described as 'natural sponges' due to their ability to retain and gradually release water. Their role in flood mitigation has been recognised for decades (Watson et al., 2016), and research has traditionally focused on understanding to what extent wetlands attenuate peak flows and how their loss or restoration can alter downstream flood risk. In this context, modelling approaches are frequently used

to provide quantitative assessments, for example by estimating the storage capacity of wetlands of different sizes. Conceptual models typically represent wetlands as simple reservoirs, while physically based models allow for the simulation of flood wave propagation within the system (Marsooli et al., 2016). This objective has been pursued since the 1980s, and more recent advances in remote sensing have further enhanced model performance by enabling volumetric mapping of soil and topographic depressions using DEM data. Today, flood mitigation through wetlands is increasingly framed within the broader context of NBS, with wetlands considered among the most promising options. Recent studies have integrated them into scenario-based models to evaluate their effectiveness, demonstrating that wetlands are particularly valuable for managing moderate and frequent flood events, largely due to small-scale topographic depressions, whereas engineered levees or artificial reservoirs remain more effective for extreme floods (Lo et al., 2021). Given the difficulty of conducting full-scale experiments, wetland modelling continues to represent one of the most powerful and widely applied tools to investigate their role in flood risk reduction.

Another major field of wetland modelling concerns contaminant removal and water quality control. This research area is particularly advanced in the case of constructed wetlands, which may also serve additional purposes such as NBS for flood mitigation, but are primarily designed for pollutant treatment. In this context, models are used to predict the concentrations of contaminants (such as phosphorus, nitrogen, biochemical oxygen demand, Biochemical Oxygen Demand [BOD], or heavy metals) in flows passing through wetlands, and, in the case of constructed wetlands, to inform design and dimensioning. Approaches range from simple empirical formulations to more complex frameworks, most of which assume steady-state flow conditions. Although these models are mainly developed with a chemical–biological perspective, they must also account for hydraulic processes, as exemplified by the widely applied “Tanks-in-Series” (TIS) model (Von Sperling et al., 2013). In some cases, modelling approaches couple flow with biogeochemical reactions, for instance to represent processes such as denitrification or plant uptake. In this field, physically based models that solve Richards’ equation for flow in porous media are widely employed, often in combina-

tion with reactive transport modules to simulate substances such as Chemical Oxygen Demand (COD), oxygen, or ammonium (Dong et al., 2019). A key limitation of these approaches is that the parameters required are highly specific and often need to be calibrated under laboratory conditions. Furthermore, solid transport modelling is frequently incorporated, making it essential to account for the fluid dynamics of the system (Defo et al., 2017; Chu and Rediske, 2012).

Models addressing water quality and contaminant removal are designed to respond to a broad spectrum of research and management questions, ranging from the design of constructed wetlands dimensioned to treat specific flow rates to the estimation of wetland areas to be restored for reducing pollutant loads in rivers. Key challenges remain in adequately representing the influence of wetland vegetation on hydrological and biogeochemical processes (Luo et al., 2019), as well as in dealing with uncertainties related to boundary conditions, such as the effects of seasonal temperature variations on plant activity and their consequent impact on flows and contaminant dynamics. This specific application is referred to as phytoremediation. In the context of constructed wetlands, it is particularly important to identify the most suitable plant species to achieve the desired water quality outcomes, as these systems are engineered ecosystems. In some cases, models have been applied and validated so extensively that simplified steady-state analytical solutions have been derived and subsequently incorporated into official guidelines, such as the German ATV-DVWK standards. Dynamic models, on the other hand, are capable of illustrating the effects of different management strategies, for example by simulating variations in water levels over time. These approaches can also reach high levels of complexity, particularly when developed in integrated frameworks (Wei et al., 2019). Nevertheless, this field represents one of the most mature areas within wetland modelling, supported by well-established frameworks and guidelines.

A more recent, yet equally pressing, application of wetland modelling concerns the assessment of climate impacts and the related biogeochemical processes. In this field, research questions focus on how wetlands respond to climate variability (particularly in hydrological terms) and how, in turn, wetlands contribute to climate miti-

gation through processes such as greenhouse gas emissions and carbon sequestration. Addressing these questions requires models that incorporate hydrological dynamics, as described in previous sections, but that are also coupled with climate and biogeochemical components. To explore wetland responses to climate change, hydrological models calibrated and validated under past or present conditions are often forced with future climate projections. In these applications, the uncertainties inherent in the hydrological model are compounded by those of the climate model; nevertheless, such studies have been instrumental in demonstrating the vulnerability of wetlands, particularly temporary ones, to changing climatic conditions. This knowledge is of considerable importance for informing adaptation strategies and management plans. Models of this kind are now widely used and analyse the climate change in different contexts, for example in mountainous areas (Lee et al., 2015) or on prairie wetlands (Johnson and Poiani, 2016).

Models for this challenging purpose are also widely used on a global scale, despite their high uncertainty: the SWAMPS-GLWD model, for example, uses satellite data and hydrological balance algorithms to estimate flooded areas on a monthly and global basis (Gedney et al., 2024).

Wetland emission models focus predominantly on methane, as wetlands represent the largest natural source of this greenhouse gas worldwide. Given the global relevance of methane storage and emissions, modelling approaches in this field are undergoing continuous development. Typically, soil conditions are represented under anaerobic regimes (where methane is generated) based on variables such as water depth and temperature, with specific sub-models then applied to simulate methane production and transport. For example, the Community Land Model (CLM) includes a wetland-specific module that estimates methane fluxes under saturated soil conditions, and numerous applications of this framework can be found in the literature (Zuo et al., 2022). Nevertheless, global methane estimates remain highly uncertain, in large part due to differences in the wetland modelling approaches used to generate them (Melton et al., 2013). Considerable efforts are therefore being directed towards improving model calibration and validation, including the use of remote sensing prod-

ucts. One recent example is represented by SatWetCH₄ model (Bernard et al., 2024), which integrates satellite-derived flood data to identify methane emission hotspots and their seasonal dynamics.

Another class of models linked to climate effects focuses on carbon sequestration. Since the early 2000s, research has sought to quantify carbon storage in wetlands (Pant et al., 2003), with particular emphasis on peatlands and mangroves. These models typically simulate plant growth, peat accumulation, and carbon fluxes under varying groundwater conditions. The balances developed within such frameworks have demonstrated that climate change or wetland drainage can shift these ecosystems from functioning as carbon sinks to becoming net carbon sources, thereby contributing directly to climate change (Roulet et al., 2025; Yang et al., 2025). In general, climate-related wetland models, and especially those addressing carbon storage, are applied at broad spatial and temporal scales, although local studies targeting individual wetlands are also reported (Nugent et al., 2018). Climate and biogeochemical models are inherently designed for integration with hydrological and climate components, as these processes cannot be considered in isolation. This interdependence, however, also increases model uncertainty and complicates validation, particularly given the scarcity of long-term observational datasets, since relatively few wetlands are subject to sustained gas flux monitoring. Despite these challenges, such models provide a concrete and promising means of exploring real-world scenarios at the global scale. This is of critical importance, as the effects of climate change and their management can no longer be overlooked, and wetlands remain among the few ecosystems that can contribute to mitigation rather than exacerbate the problem.

Another recent motivation for modelling wetlands, closely linked to climate change, concerns ecological conservation and habitat modelling. Models provide a relatively rapid and effective means for predicting ecological responses within these complex environments (Kumar and Zhao, 2011). In this context, the primary research questions relate to conservation planning, for instance, how the breeding habitat of a given species might evolve under changes in wetland hydrology or climate, or which areas should be prioritised for protection in order to safeguard regional biodiversity. Some

applications focus specifically on vegetation communities. Traditional approaches have relied on statistical correlations between species occurrence and hydrological parameters such as wetland depth or hydroperiod, whereas more recent developments increasingly employ dynamic ecohydrological models (Liu et al., 2022). A notable example is the Everglades Landscape Model (ELM) (Fitz et al., 2004), designed to simulate hydrology and nutrient dynamics in order to predict vegetation change. The ELM has been widely used in scenario analysis due to its distributed and predictive capabilities. However, models of this kind require extensive site-specific data; while highly valuable for local management, they are generally not transferable to other systems.

Habitat modelling of wetlands is often undertaken with a specific emphasis on water-bird populations. Characterising flooding regimes, including areas subject to intermittent inundation, provides essential insights into the ecological requirements of species for feeding and breeding (Haig et al., 2019). While scenario analyses have traditionally focused on extreme climatic events, such as prolonged droughts, there is growing recognition that anthropogenic drivers (Lehikoinen et al., 2017) often exert a stronger and more predictable influence on habitat availability. As a result, scenario-based modelling increasingly incorporates human-induced alterations, such as land-use change or water abstraction, which can be explicitly schematised. Nevertheless, the uncertainty associated with both climatic and anthropogenic scenarios remain a central challenge and must be carefully addressed in model interpretation.

In many wetlands, particularly coastal systems, avifaunal modelling is often coupled with fish habitat modelling. A recurring research question concerns how these habitats may be altered following the construction of infrastructure such as dams, water-course diversions, or, conversely, the restoration of river systems. Hydrodynamic habitat models tailored to specific species, such as CASiMiR (Zingraff-Hamed et al., 2018) and MesoHABSIM (Negro et al., 2021), are increasingly employed to address these issues with a high degree of detail. With the refinement of modelling techniques and the growing demand for integrated outcomes, models designed to assess ecosystem services are also becoming more widespread (Janse et al., 2019). These ap-

proaches are typically simplified but are often combined with other modules, for instance by incorporating economic dimensions that are frequently overlooked in the more ecologically focused applications. A well-known example is InVEST, a modelling framework that identifies the main drivers of ecosystem services provided by wetlands (Zhu et al., 2024) and supports the evaluation of optimal restoration strategies (Hu et al., 2020). Such models therefore operate at the interface between two fundamental purposes of wetland modelling: ecological conservation and decision support. Ecological conservation models for wetlands, in general, address thresholds and non-linear responses, recognising that these ecosystems can shift between alternative stable states (e.g. clear vs turbid water) depending on drivers such as nutrient inputs. By incorporating feedback and tipping points, such models support restoration planning, highlighting the need to reduce pressures below critical thresholds. While uncertainty remains high due to the complexity of ecological interactions, scenario analysis makes these models valuable tools for exploring potential futures and create a general framework for guiding conservation decisions.

Having outlined the more specific purposes for which wetland models are implemented, it is now important to consider a broader application that cuts across multiple modelling approaches: decision support. In the aftermath of natural disasters or increasingly frequent emergencies linked to water scarcity, such as droughts, water shortages, and competing demands, Decision Support Systems (DSS) are increasingly incorporating wetlands (Zare et al., 2021), both as vulnerable ecosystems requiring protection and as resources that can actively contribute to recovery, for instance when recognised as Nature-Based Solutions (Ferreira et al., 2023). To be effective, these models must integrate socio-economic and policy constraints with outputs from physical models; however, linking management objectives with underlying physical processes remains a significant challenge. A key gap in this urgently needed area of research is therefore the development of fully integrated, cross-sectoral frameworks, which are still lacking despite parallel advances in physical and socio-economic modelling. Among the most common decision-support applications are water allocation models (Hatamkhani et al., 2022), primarily developed for arid and

semi-arid regions or for highly anthropized basins where wetland persistence depends directly on the allocation of environmental flows. These models increasingly incorporate participatory approaches, as their primary purpose is to address conflicts between the conservation of natural areas and the use of water for agriculture, particularly under the critical conditions imposed by climate change (Mirzaei et al., 2021). Optimisation algorithms, which have long been applied in water resource allocation, have more recently been employed in studies focusing on wetlands (Hatamkhani and Moridi, 2023). However, a persistent challenge is the integration of wetland-specific information, both hydrological data and qualitative, context-dependent knowledge, into these simplified modelling frameworks.

In some cases, decision support modelling does not address, as previously noted, how to inform policy regarding the amount of water that may be diverted from wetlands for agricultural use, or which wetland should be prioritised for anthropogenic exploitation. Rather, the focus shifts, subtly but significantly, towards supporting wetland restoration planning in both natural and urban contexts (Consorte Widis et al., 2015). Decision support tools are also employed in short-term emergency management, for instance to predict whether and how accidental pollutant discharges into rivers may affect downstream wetlands (Bi et al., 2011). In all such applications, model outputs must be transparent and readily interpretable for stakeholders. This requirement explains why integration of these approaches has advanced only gradually (Guo et al., 2024), as the primary limitation remains the communicability of results, a challenge that has only partially been mitigated by the recent introduction of machine learning methods, which themselves often rely on ‘black box’ approaches. To address this gap, decision support frameworks frequently employ simple, multidisciplinary indicators to convey results effectively (Zhao et al., 2016).

The following section examines the major purpose of wetland modelling considered in this study: the representation of the water balance and hydrological regime. This is one of the most fundamental and long-established applications of wetland modelling and indeed provides the basis upon which many other models rely. Wetland water balance is presented here as the starting point for this work, precisely in light of both

the limitations and the potential of existing approaches, and to introduce the objectives and capabilities of the WetMAT model, which will be described in detail in the following sections and chapters.

Hydrological balance models address the research questions related to the quantification of water storage capacity within wetlands (in terms of surface, volume and depth), the calculation of hydroperiods, and the identification of interactions between wetlands and adjacent water bodies. They are also employed in combination with more specialised modelling approaches, for instance, those focusing on flooding, ecological dynamics, or pollutant transport.

Hydrological models for water balance are generally conceptual, and the components considered relate to precipitation, evapotranspiration, inflows and outflows from water bodies, as well as exchanges with groundwater. Models such as MIKE SHE or SWAT are the most commonly used in this case, and when wetlands are isolated, it is clearly easier than for wetlands that are part of a larger hydrological context. The outputs of hydrological modelling provide information on annual water surplus or deficit, length of the hydroperiod, levels and extent of the wetland, as well as drainage frequencies in temporary wetlands. Detailed guides on hydrological modelling, such as the Wetland water balance modelling guidance document by the Toronto and Region Conservation Authority (2020), offer insights into the actions to be taken depending on the wetland of interest, as well as the type of model to be used.

1.4 Challenges and limitations in wetland modelling

Despite the purposes for which they are used and the remarkable applications described, wetland modelling still presents persistent challenges that limit its accuracy or applicability.

Below are the main limitations that these types of models generally present, and how hydrological science is working to resolve them, in order to conclude the summary framework within which the WetMAT model fits and attempts to add a piece to the current developments in the literature.

A fundamental challenge common to all wetland models is the scarcity of measured data. In situ observations are often difficult to obtain, as wetlands are frequently ungauged systems. This difficulty arises partly from the fact that wetlands are natural areas and partly from the practical challenges associated with installing and maintaining monitoring equipment. Factors such as soft soils, flooding, and restricted accessibility for operators often lead to data gaps, resulting in discontinuous and noisy time series. Even tracer experiments are more complex than in pipes or channels, as wetlands constitute uncontrolled and multidimensional environments. Flow data are limited, while water quality data are even less available. The lack of gauging stations is particularly pronounced in developing regions and in temporary wetlands, whose ecological importance has only recently been acknowledged. Remote sensing has partly alleviated this issue (Kissel et al., 2020), although such observations remain indirect and still require ground-based measurements for calibration and validation. Innovative strategies have therefore been proposed, including the use of environmental tracers (e.g. isotopes in water flows) and alternative calibration methods such as participatory approaches (Ritzema et al., 2010) or historical documentary sources. Nevertheless, data scarcity continues to represent a major source of uncertainty in wetland modelling.

The heterogeneity of wetlands in terms of vegetation and soil properties, which vary markedly from point to point, results in uneven water distribution and makes accurate representation challenging. To address this, gridded representations are often employed to distinguish areas that remain permanently saturated from those that are only occasionally inundated. High-resolution LiDAR surveys can reveal depressions and natural features that influence internal flow pathways, although dense vegetation frequently obscures such details. Satellite imagery can also be employed for this purpose, although vegetation frequently obscures accurate detection. Synthetic Aperture Radar (SAR) provides clear advantages in this context, yet it often lacks the spatial resolution or temporal frequency required for detailed analysis. These remote sensing tools are generally effective in detecting temporary flooding events or major changes in wetland extent, but they are less reliable in capturing subtle or small-scale varia-

tions. This limitation is typically addressed through conceptual simplifications or parameter calibration. Applying uniform parameter values across an entire wetland, particularly at larger spatial scales, produces unrealistic outcomes; however, explicitly accounting for spatial variability substantially increases model complexity and, with finer grid resolutions, also introduces computational constraints.

Challenges in representing vegetation affect not only the modelling of flood dynamics but also the simulation of processes that directly influence hydrological flows. For instance, the seasonal or multi-annual expansion of invasive aquatic plants can substantially alter system behaviour. Such changes directly impact evapotranspiration estimates, which even simplified balance models cannot disregard. Although integrated ecohydrological models are capable of addressing these interactions, their complexity has so far limited their widespread application.

For similar reasons, the integration of hydrological and ecological models, an area currently receiving considerable attention, is inherently complex, as their outputs operate on different temporal scales and orders of magnitude. Fully coupled models remain rare, primarily because of over-parameterisation, while many ecohydrological models are necessarily simplified. As a result, they often fail to capture system behaviour comprehensively, particularly under extreme conditions.

Innovative approaches to address some of the aforementioned limitations often rely on high-resolution modelling. However, this strategy is not always advisable, as it is computationally demanding and therefore unsuitable for long-term simulations or extensive sensitivity analyses. With very fine grids, some simulations may require several days to complete, making scenario implementation or Monte Carlo analyses for uncertainty quantification impractical. In contexts requiring multiple iterations, computational burden becomes a major limitation, further compounded by the challenges of managing large datasets and handling big data, which demand advanced technical expertise beyond hydrological knowledge alone. For these reasons, conceptual models are frequently preferred over fully distributed ones, particularly in the initial stages of research.

Given the inherent complexity and heterogeneity of wetland systems, model transferability remains a persistent limitation, even across sites of comparable scale. For example, a nutrient removal rate calibrated in one wetland may not be valid in another with different hydrological, thermal, or vegetative conditions. Although initiatives such as the Ramsar Wetland Modelling Toolkit have sought to provide standardised protocols and best practices, their practical impact is constrained by the highly site-specific nature of wetland dynamics. This limited transferability significantly reduces the generalisability of model outcomes and represents a major barrier to their broader application in wetland management and conservation policy.

Concluding, recent advances are reshaping wetland modelling by addressing long-standing challenges such as data scarcity, model complexity, and integration across disciplines. Remote sensing and IoT technologies are improving data availability, while drones and sensor networks provide high-resolution and near real-time monitoring. Machine learning and hybrid approaches enhance predictive capacity and help manage uncertainty, complemented by high-performance and cloud computing that enable large ensembles and uncertainty analyses. Modular frameworks and coupled land-surface–flood models, despite their limitations facilitate integration of hydrology, ecology, and climate processes, while sensor-based calibration and feedback control are envisaged for constructed wetlands. Emerging approaches also include socio-economic integration (e.g. agent-based models) and open data platforms that promote collaboration and reproducibility.

All these innovative solutions and tools, developed to overcome the limitations of existing models, provide valuable support to ongoing modelling efforts. Nevertheless, in the hydrological domain, it remains essential to preserve the structural simplicity of the model, as uncertainty is inherently high in such environments and the primary objective is to explain the underlying system mechanisms using the limited resources typically available.

1.5 Overview on *WetMAT* model

Within the context described in the previous state of the art, the following section introduces the rationale for this work and outlines the general features that characterize the *WetMAT* hydrological model compared with the existing approaches.

The growth of research on wetlands, as can be seen from the analysed literature, has been driven primarily by the need to manage these ecosystems consciously and sustainably, not only with the aim of preserving them but also, and increasingly, to harness their potential in addressing anthropogenic challenges such as carbon sequestration and water purification.

Although many of the models developed for the purposes outlined above incorporate hydrological components to some extent, or cannot disregard them entirely, these aspects are often treated only marginally, for instance in empirical models designed with objectives other than hydrological outputs, or do not represent the primary focus of the modelling framework.

Given the relatively scarce applications, there is a need for models that move beyond hydrological processes to explicitly account for ecological water needs within wetlands. In the context of holistic management frameworks, such as the Water–Energy–Food–Ecosystem (WEFE) Nexus, wetlands with their ecosystems are increasingly recognised as legitimate water-dependent systems that require explicit allocation of water resources since maintaining their ecological functions depends on meeting specific hydrological requirements (Acreman et al., 2014). These requirements, defined in terms of quantity, timing, and quality, are therefore comparable to those of human water uses, including industrial, domestic, and agricultural sectors (Albrecht et al., 2018).

This study, inspired by simple hydrological models specifically implemented for wetlands, used for example to evaluate the effect of climate change (Lee et al., 2020), aims to develop a generalised model for wetlands, designed to perform hydrological balance analyses in order to evaluate their water requirements. The model, named *WetMAT* (Wetland Management Tool), has been conceived as a flexible but also simple framework to support such assessments, following the idea of the classical hy-

drological model for wetlands, with the general water balance framework illustrated in Figure 4 (Hare et al., 2022).

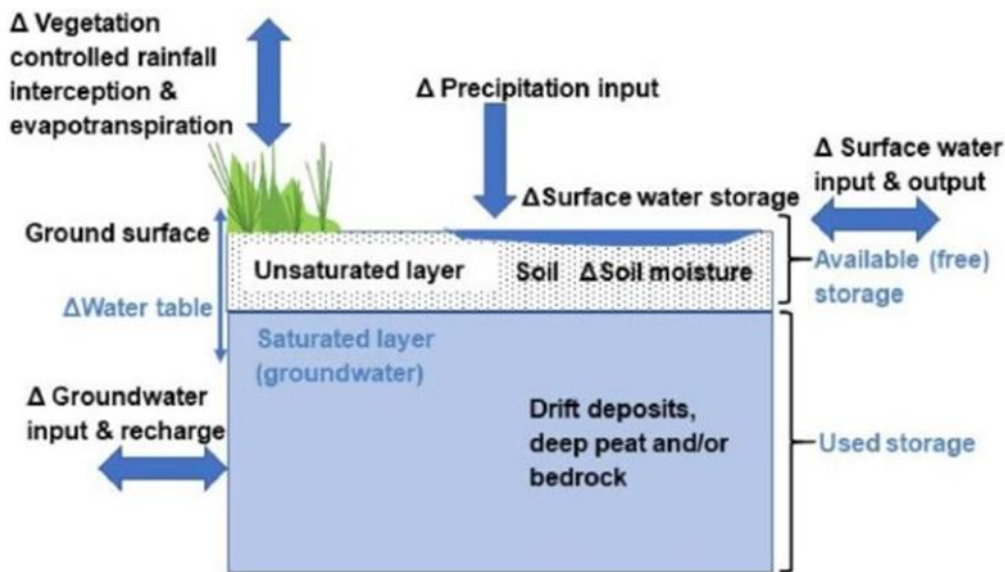


Fig. 4 – Conceptual scheme of a classical wetland hydrological balance, with an indication of the main variables that act on a wetland system and the main processes that the model must reproduce, Source: Hare et al., (2022)

As previously noted, wetlands can be classified into different types according to their morphological setting. Among these, coastal wetlands, whether located at river mouths or fed by springs, represent particularly fragile systems, being subject to pressures from both terrestrial and marine environments. The WetMAT model was developed in response to the need to study the water requirements of such environments, which are not only complex but also central to numerous protected areas, particularly in the Mediterranean region. Here, acute conflicts frequently arise between wetland conservation and agricultural demands for water resources. These tensions are further intensified by climate change, with rising temperatures and declining precipitation posing major challenges. In arid and semi-arid regions, in particular, pro-

longed droughts have become increasingly common, further threatening the ecological balance of these vulnerable systems.

The requirements derived from simplified hydrological modelling can serve as inputs for decision support when integrated with other frameworks, yet hydrological modelling remains fundamental. Accordingly, WetMAT does not incorporate ecological or biogeochemical modules, as is often seen in the literature. Instead, with a focus on conservation and protection, it is specifically designed to represent the basic hydrological functioning of the system.

The WetMAT model, by representing simple yet fundamental processes in a primarily conceptual form, is designed to address the issue of environmental water requirements from a quantitative perspective. Simplicity is a necessary feature, to minimize model parameterization and allow model validation with limited observations available for wetlands, which are historically regarded as minor and overlooked water bodies, until the recent attention on conservation.

This simplicity is reflected first of all in the geomorphology, within which the natural system is schematized using a few effective comparison parameters, such as the basin area, the drainage density, depth-volume-area function of the topographic depression and average section of channel network.

Depending on the reality to be represented, the model will schematize even complex forms with fundamental features which regulate the hydrological balance, which can be summarized through Equation 1:

$$\frac{dV_w}{dt} = P + I_w + G_w - O_w - D_w - ET_w \quad (1)$$

where $\frac{dV_w}{dt}$ is the volume of water that floods the wetland during each timestep, P is the daily inflow from precipitation, I_w is the wetland inflow from rivers and surface waters and G_w is the wetland exchange with groundwater, which can be negative or positive depending on type of exchange with bordering aquifers. O_w represent outflows from the wetland, i.e. towards downstream rivers or coastal discharge. ET_w is the term for daily evapotranspiration and D_w , lastly represents the lateral drainage of

channel network that constitute the system, which allows us to represent the internal drainage exchange between land patches and wetland elements represented by ponds and drainage channels.

The WetMAT model is structured on a daily time step, which represents an appropriate compromise between model parsimony and the level of detail required to characterise hydrological regimes. This resolution is also appropriate for assessing the water requirements of temporary wetlands, whose areal extent and hydrological response vary markedly over the course of the year. A finer temporal resolution would significantly increase data demands and computational complexity, whereas coarser resolutions would risk overlooking critical short-term dynamics relevant to wetland inundation and persistence.

Irrespective of the equations applied in specific case studies, the fundamental feature of WetMAT is its structure, which is organised into three blocks, three interconnected water balance modules that represent specific physical processes with mutual water exchange between each other. First, a Soil Water Balance is lumped over the entire wetland domain: when the soil is above field capacity, a "drainable water flux" is generated, which is the input of the Channel Water Balance lumped over the entire drainage network. The channels are the schematical representation of the most dynamical component of the wetland, which are the first to be inundated during flooding events and the last to drain. When the channels reach a certain threshold, the excess water, namely the "channel overflow flux", starts inundating the wetland. The Wetland Water Balance lastly describes the wetland flooding phenomenon and allows the computation of key variables such as the flooded area, depth, volume, and hydroperiod.

Figure 5 summarizes in a schematic and conceptual way the sequence of the three balance components with an indication of the most relevant outputs that the model offers.

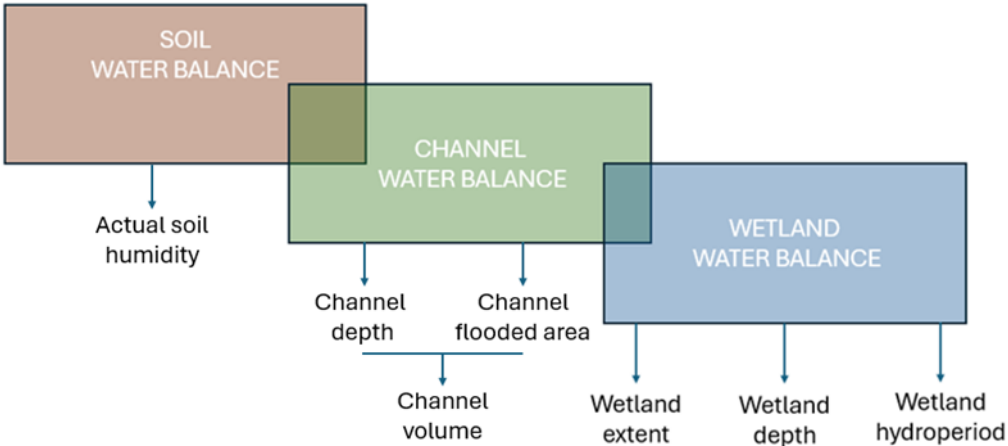


Fig. 5 – Generic schematization of the WetMAT model and its main outputs

The following chapters will start from this concept and will in-depth examine the implementation of these general characteristics, together with additional specific features, across case studies of varying complexity.

2 LARGE-SCALE APPLICATION OF THE WETMAT MODEL: THE DOÑANA MARSHLAND CASE STUDY

This chapter presents the development of the WetMAT model through its application to the Doñana Marshlands (SW Spain). This case study provided favourable conditions for the model's initial implementation, owing both to the large spatial scale afforded by the system's extent and to the simplifications that could be applied in the hydrological representation. At the same time, Doñana was selected for its emblematic status, reflecting both its international ecological importance and the well-documented conflicts over water resource use within the area, thereby making the WetMAT modelling particularly suitable for its primary purpose of identifying environmental water requirements to support restoration efforts.

In this chapter, the framework of the case study will be outlined in detail, together with the case-specific adaptations of WetMAT. A sensitivity analysis of the model will be conducted, and the main outputs it generates will be presented and discussed. These outputs aim to support the identification of the wetland's water requirements, in order to guide possible restoration actions. In this context, the introduction of a synthetic hydrological response index, named Inundation Persistence Index (IPI), was explored. The index was conceived specifically for temporary wetlands, with the aim of identifying drought periods and understanding water needs under such conditions, based on key hydrological features and regimes of the system.

2.1 *The case study*

The Doñana region (37°N, 6°W) is an area overlooking the Atlantic Ocean bordered by the estuary of the Guadalquivir River to the East and the estuary of the Tinto River to the West. Internally extends to the valley of the Guadiamar River (see Figure 6).

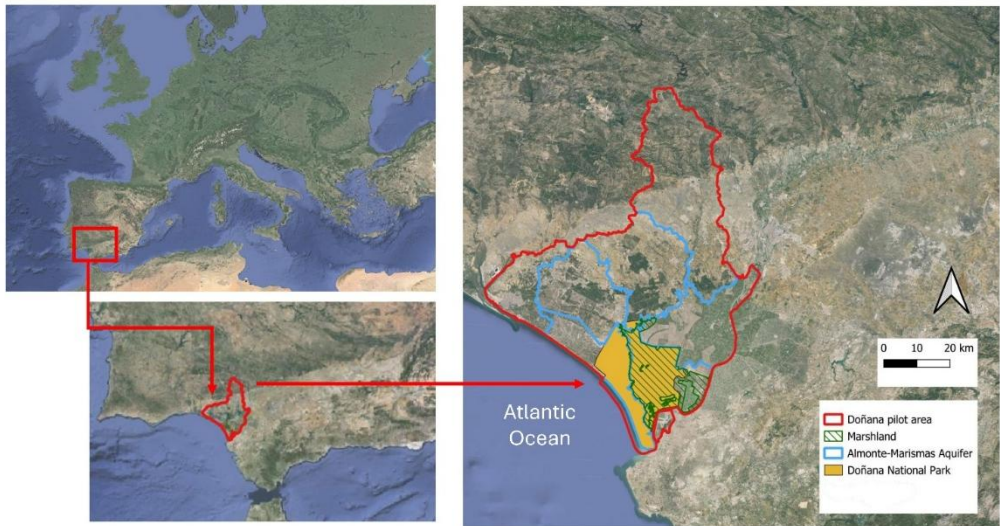


Fig. 6 – Overview of the study area and its major characteristics (aquifers, marshland, National Park) (elaboration from © Google Earth 2024)

Located in the SW of Spain, Doñana hosts the largest wetland in the Country, as well as one of the largest in Europe. Such wetland in addition to offering scenarios of considerable aesthetic value, has a particular ecological importance: given its crucial position between the European and the African continent is an important migratory stage and crossing point for birds and in fact here are 75% of the birds present throughout Europe (Fernandez-Delgado, 1997). This centrality is further supported by the findings of Deboelpaep et al. (2022), who demonstrate that along the East Atlantic Flyway a small number of large coastal wetlands sustain most of the functional connectivity for waterbirds. Within this network, Doñana's extensive marshes occupy a strategic position, acting as a major node on the principal migratory pathways across Western Europe. From a demographic point of view, in the region of Doñana there are 12 municipalities that are part of three provinces and that collect about 200.000 inhabitants. The main economic activities of the area, as will be observed later, concern tourism and especially agriculture (Palomo et al., 2011).

Regarding climate, Doñana is exposed to a sub-humid Mediterranean climate, characterized by rainy autumns and winters, hot and dry summers, mild winters. The aver-

age annual rainfall is 540 mm/yr, but this is not really indicative. Because of the Atlantic influence this area is subject to a considerable variability of precipitation that does not show any trend (Serrano, 2016). Moreover, like many Mediterranean areas, climate change is estimated to be particularly aggressive here, and climatic projections show a gradual increase in temperatures that could affect these water dependent ecosystems.

The natural context of Doñana is characterized by the presence of different ecosystem units: marshes which is the predominant one, sand dunes featured by temporary ponds, coastal systems and estuary (Zorrilla-Miras et al., 2014). The topology of the marshland, consists of a flat zone characterized by slight depressions ("lucios") and equally slight rises ("vetas") that even during periods of major flooding of the area are not flooded. The marshland of Doñana has, at least originally, a complex hydrology given by several streams that flew into it and that participated in its flooding in addition to the contributions of rain. Among these the most relevant were Guadiamar River, El Partido stream and La Rocina stream (Serrano et al., 2006). Initially, the natural marshland covered an area of 1,500 km², but over time it has been reduced by approximately 80%, now spanning around 300 km². This reduction is largely the result of various anthropogenic modifications, particularly those related to agricultural development during the 20th century (Leiva-Piedra et al, 2024; Green et al, 2024).

Also the underground hydrology is complex, as due to the deltaic origin of the area it is possible to find a confined aquifer that mainly lays under the marshland and one, unconfined and fed by rainfall, in the coastal zone (Susó and Llamas, 1993). These aquifers, commonly known as "Almonte-Marismas" unit, have been subdivided into five different units for administrative reasons as well as for geological ones. Some of these, according to different monitoring systems, are in poor quantitative and/or qualitative status (Confederación Hidrográfica del Guadalquivir, CHG, 2022).

According to its geology, under the marshland lies a clay layer that keeps it isolated from direct exchanges with the groundwater below (Naranjo-Fernández et al, 2018).

As an area where nature and society have been living together for centuries, there are numerous safeguards that concern Doñana, probably the most significant European

wetland. 1969 saw the establishment of Doñana National Park, a strictly protected area. In 1989 an additional area, which expanded the boundaries of the National Park although with more permitted activities was established: the Doñana Natural Park. Both Parks were united in 2005, being part of a single Natural Protected Area. Doñana has also received international protection awards: in 1980 it was recognized as International Biosphere Reserve, in 1982 the wetland was officially recognized by the Ramsar Convention for wetlands and in 1995 by UNESCO as World Heritage Site. In 1990, however, the site of Doñana has also returned to the Montreaux Record of Ramsar Sites Under Treat: as will be explained below there are many anthropic actions that over time and still endanger the conservation of this natural reserve (Martín-López et al., 2011).

The territory of Doñana, from the middle of the last century and precisely from the '60s when the Irrigation Area of the Lower Guadalquivir was established with significant changes also in the landscape, has been affected by the impacts caused by the agricultural sector. Initially these transformations were also encouraged by the Spanish government and to make the area more profitable it has undergone profound changes in its hydrology, with the reduction of the marshland through draining. Waterways have been regimented, for example by disconnecting the Guadiamar River from the marshland itself through the construction of "Entremuros", an artificial embankment that let the Guadiamar river to flow directly into the ocean while before it was the wetland's main source of supply. Although attempts have been made to restore the natural waterways, their effects on the wetland hydrological regime have been limited, and the marshland is effectively no longer supplied by the Guadiamar River or its tributaries. (Baena Escudero & Guerrero Amador, 2006).

Agriculture near the Doñana's wetland today is mainly oriented to berry production: this area is one of the main exporters throughout Europe. The product of such highly hydro-demanding crops involves the pumping from groundwater of large amounts of water, sometimes even from illegal wells and therefore difficult to estimate: piezometric analyses in the area show significant drops in the water table and negative effects were claimed (Díaz-Paniagua & Aragonés, 2015) on the natural system for example

the discharge reduction of the small streams that feed the marshland as well as the springs on the edge of it (ecotones).

Doñana region is also threatened from others anthropogenic factors: tourism is also exerting pressure on water resources of the area. The coastal tourist centre of Matalascañas, with its increasing water pumping from the aquifer concentrated especially in the summer season (over 100,000 people each year), was identified as a contributing factor to the significant lowering of the coastal aquifer leading to the reduction or even disappearance of the dune ponds near the village (Serrano & Serrano, 1996; Sousa et al., 2009).

Industry has also contributed to the negative anthropogenic pressures on the area. In 1998, the accidental collapse of the Aznalcollar dam forced local authorities to further isolate the wetland from the Guadiamar River to preserve it from the large volumes of toxic sludge that would have invested it.

Figure 7 illustrates the successive actions of protection and losses of Doñana's marshland throughout the 20th century, primarily due to agricultural expansion. Most of the conservation actions implemented during this time did not translate into tangible restauration of the marshland but rather served as formal acknowledgments of its ecological value and the necessity to safeguard it. Currently, nearly 80% of the original marshland has been lost to anthropogenic activities, while the remaining portion is under active protection.

Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes

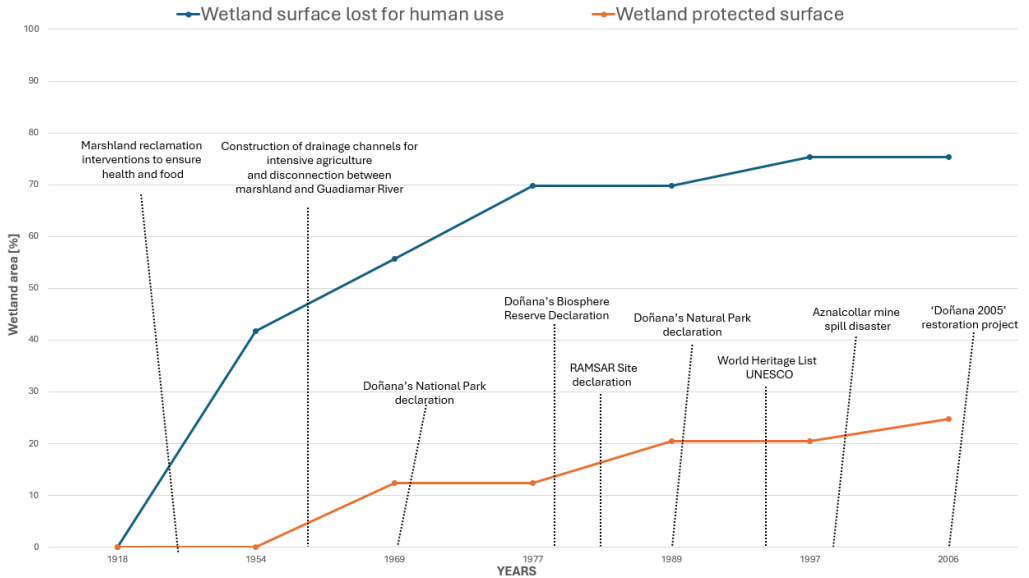


Fig. 7 – History of conservation and losses actions in the Doñana's marshland during the 20th century by the percentage of wetland area protected or exploited for human uses

Doñana area represents a complex water-dependent ecosystem, rich in anthropic pressures on an environment in which water dynamics constitute the key structuring element in the system. In this case, a hydro-ecological model for the estimation of environmental needs not only constitutes a research challenge but is necessary for the management of resources also considering the threat of climate change in that area (Gil Gil & Schmidt, 2024). Doñana, although lacking long-term observation network to characterize the dominant hydrological fluxes, is a globally recognized study site for several scientific disciplines.

Figure 8, based on official data from the 2022–2027 Hydrological Plan of the Guadalquivir Hydrographic Demarcation drafted by Confederación Hidrográfica del Guadalquivir (CHG), illustrates a segment of the ongoing exploitation of the area through a quantitative analysis of groundwater use across different human sectors. Nearly all groundwater bodies of the area are classified as being in poor quantitative and qualitative status (CHG, 2024), as also indicated by the Groundwater Exploitation Index GEI (Portoghese et al., 2013), which relates the natural recharge of the aquifer

to the volume of abstractions. According to European reference thresholds (European Environment Agency, 2025), when the exploitation indexes exceed 40%, the system is considered to be under severe water stress and subject to unsustainable resource use. In this case, as indicated by the GEI in Figure 8, such conditions are observed in three instances, as confirmed by CHG estimates.

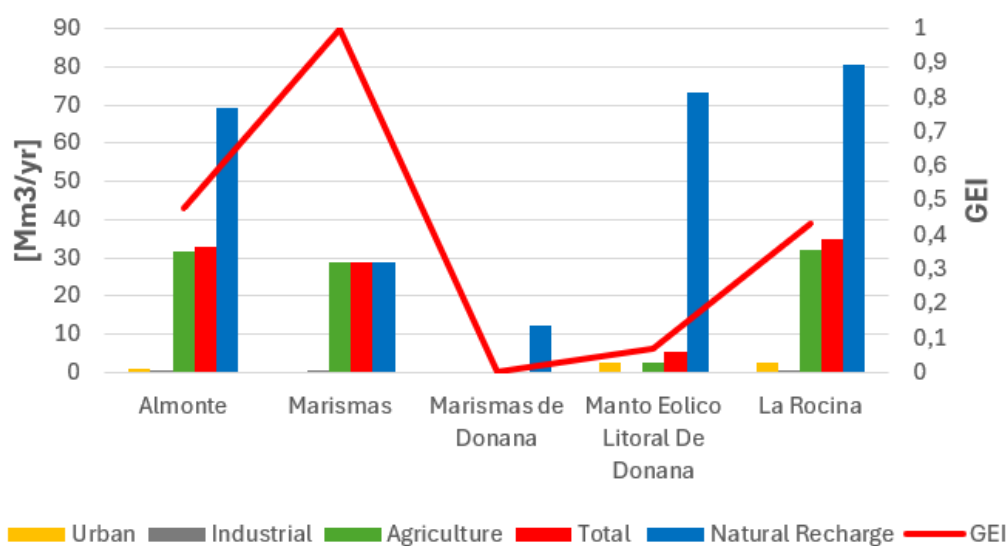


Fig. 8 – Representation of aquifer exploitation in the Doñana area and calculation of the Groundwater Exploitation Index (GEI).

Being Doñana subject to different anthropogenic pressures and therefore with many stakeholders directly or indirectly (as users of the products, not just as producers) involved in, Doñana constitutes an interesting case study for its characteristics but also for this strong and not negligible contrast between wetland conservation and agricultural development.

As a preliminary analysis within the framework of a water resource allocation model, the system was simulated using the WEAP (Water Evaluation and Adaptation Planning) software (Yates et al., 2005). This tool allows for a simplified yet structured representation of real systems through the use of nodes (representing water supply and demand) interconnected by links. Both nodes and links were customised for agricul-

Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes

tural demand nodes (i.e. characteristics of individual crops with their areas), natural 'runoff/infiltration' connection links and artificial 'transmission links'. Each user can also be assigned a preference order, enabling the model to simulate future scenarios and identify which sectors are most likely to experience water shortages, as well as potential strategies to ensure adequate water allocation. Figure 9 shows the WEAP schematic implemented for the Doñana region, with the various demand nodes grouped by aquifer, as reported in official documents.

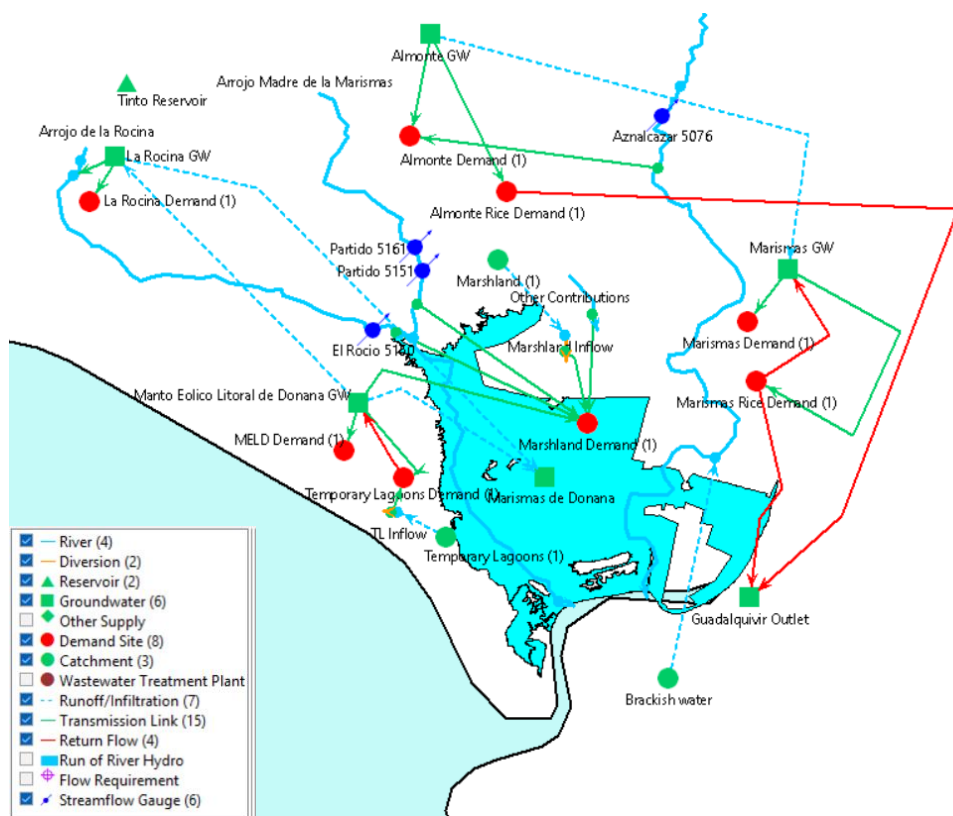


Fig. 9 –Schematic representation of Doñana case study through WEAP software for water accounting

However, the initial analyses revealed that as a tool designed for water resource allocation planning WEAP requires clearly defined input data for environmental water demands, comparable in precision to those used for human needs. Only once these

inputs are established such tool can effectively support the development of water management scenarios. For a complex system such as a wetland, and particularly for a temporary one like Doñana, estimating environmental water requirements necessitates suitable modelling tools. This need motivated the development of the WetMAT model, whose characteristics are described in the following section for the Doñana case study. The model is specifically designed to capture the typical hydrological regime of wetlands through water balance analysis, thereby providing the essential environmental water demand required for stabilizing water-dependent ecosystems even under drought conditions.

2.2 Model configuration and main characteristics of WetMAT model in the Doñana case study

As outlined in the case study description, the Doñana wetland is characterised as a temporary, or seasonal flooding, which have a lower hydroperiod compared to permanent aquatic systems (Calhoun et al., 2017). Hydroperiod, as a variable related to the timing, duration, and frequency of inundation, plays a crucial role in shaping the ecological structure and function of wetlands by influencing the distribution of plant and animal species (Mitsch & Gosselink, 2007). For this purpose, the tool should be capable to dynamically describe the strong hydrological variability that affects the area throughout the year.

Wetland flooding, which is mainly due to rainfall in the case of Doñana, as it is almost completely disconnected from surface water and the aquifer, follows the annual course of the precipitation. So, there is a wet season, generally starting in October and ending in late Spring for the Doñana wetland, and a dry season, generally from April to September, when the evapotranspiration process is predominant over precipitation (Fernandez-Carrillo et al., 2019). At the beginning of the wet season, the first rainfall contributes to the imbibition of the soil. Once the soil is saturated, ponding firstly occurs within the most depressed areas of wetland. During the Winter season, with the increase of frequency and intensity of rainfall, the most significant flooding

occurs. As the soil is saturated and the most depressed areas filled, wetland starts to flood almost uniformly. During the Spring season, when rainfall events start reducing and temperatures rise, flooded areas reduce, and the small soil depressions start to drain out. During Summer the contribution of precipitation is completely missing while evapotranspiration reaches its annual peak: the drying process of the marshland thus continues and involves also the soil. The whole process is cyclical, although it varies from year to year as the variability of rainfall in the study area is high. Figure 10 shows the process of marshland flooding and emptying during the hydrological year. It also includes a reference to the main drivers considered in the modelling, i.e. the precipitation P and the evapotranspiration ET .

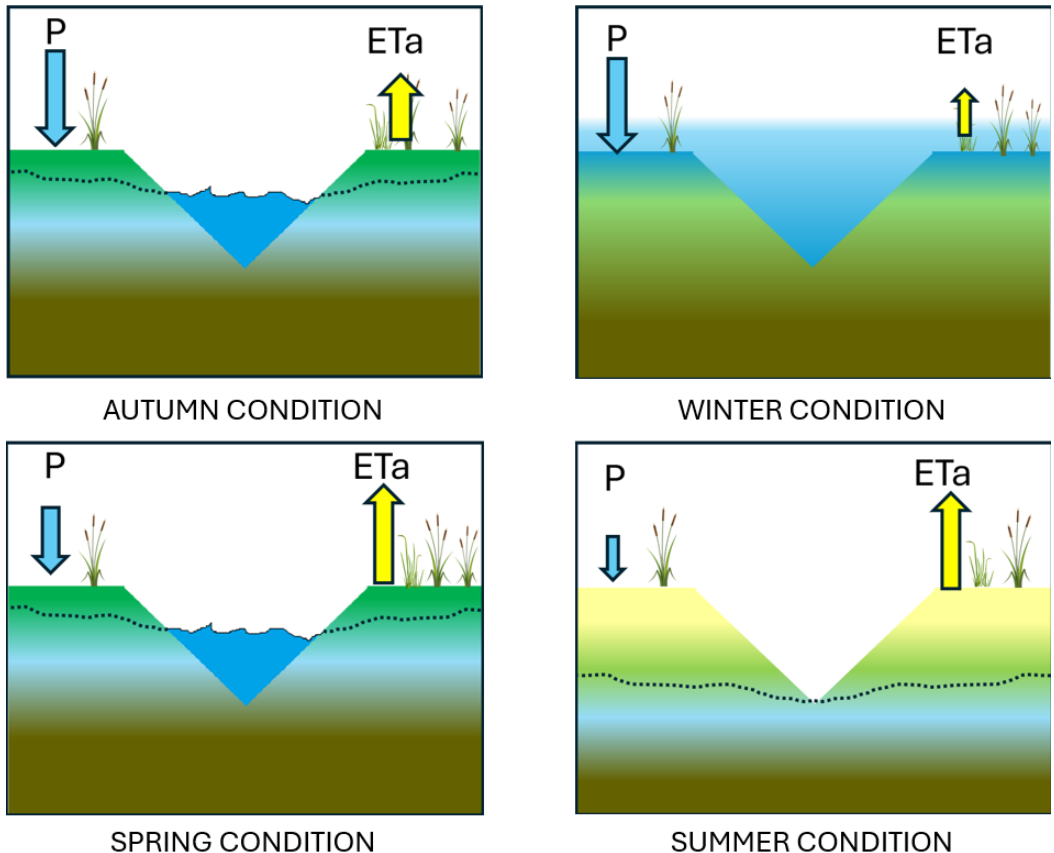


Fig. 10 – Conceptual representation of the flooding and drying processes of the temporary wetland, reproduced by the WetMAT model. The figure illustrates also the main processes of precipitation and

evapotranspiration, with the size of the arrows qualitatively representing the relative magnitude of each phenomenon

The processes schematically represented in Figure 10 are reproduced by the WetMAT model at daily scale through the three interconnected modules as shown in the flowchart of Figure 11, representing the major sub-domains and respective water flows. The functioning of the three modules and the related inflows and outflows has already been described previously, and Figure 11 fully identifies the natural input variables and outputs of each individual water balance. While evapotranspiration is common to all the three water balance modules, additional phenomena such as lateral drainage through channels and vertical losses contribute to the marshland draining process.

Despite the three sub-models the WetMAT model structure considers a single spatially lumped unit where the marshland is considered as a single cell on which the various processes act. This leads to approximations, which, however, on a large scale, as in this case study, are reasonably tolerable.

Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes

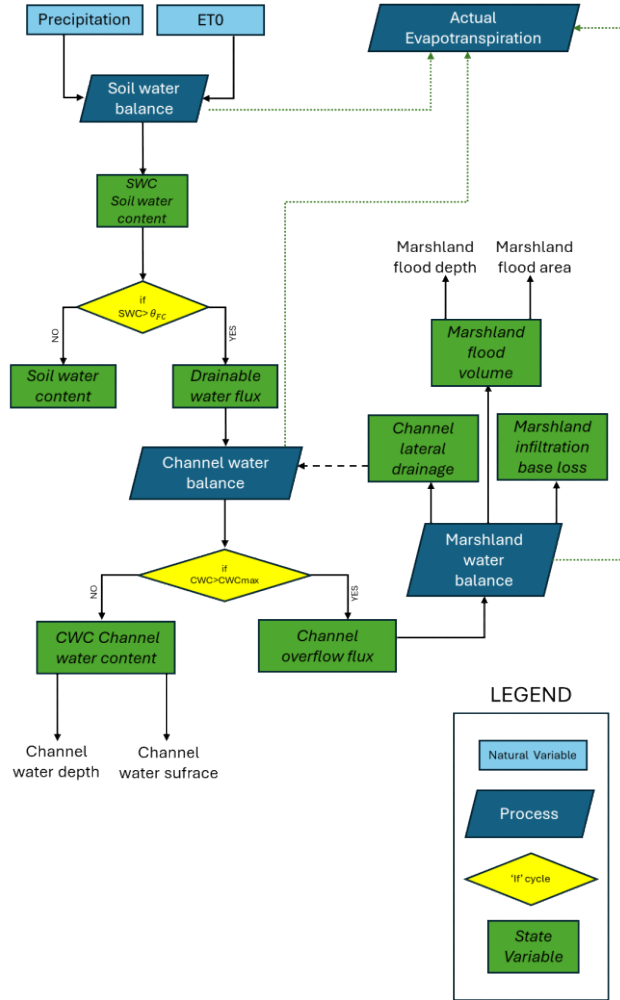


Fig. 11 – WetMAT model flowchart for Doñana case study

Going further into details, the WetMAT model structure relies on nine parameters in order to characterize the system in terms of water storage and drainage features. Some parameters depend on soil hydrologic parameters and land cover of the area, i.e. the soil water content at wilting point and field capacity (θ_{WP} , θ_{FC}), the lateral drainage factor (Df), the subsoil hydraulic conductivity (K) and the root zone depth (Dr). Other model parameters refer to system hydrogeomorphology i.e., marshland area (Am), slope of channels banks, maximum channel water depth (Hp), number of

channels (n). Once these parameters are set (through direct survey and/or calibration), other depending on parameters are calculated. These include the channel maximum width, the marshland maximum soil water storage, the single channel length, the total channel length, the channel maximum water storage and the drainage density. Table 1 summarises the characteristics of the parameters and quantities described. In the ‘Type’ column, ‘H’ denotes parameters related to hydraulic characteristics, while ‘G’ refers to hydro-geomorphological characteristics.

Table 1 – Parameters and derived quantities used by WetMAT model

PARAMETER	DESCRIPTION	UNIT OF MEASURE	TYPE
ϑ_{WP}	Soil water content at wilting point	mm m ⁻¹	H
ϑ_{FC}	Soil water content at field capacity	mm m ⁻¹	H
Df	Lateral drainage factor	m s ⁻¹	H
Slope of channels banks		deg	G
K	Subsoil hydraulic conductivity	m s ⁻¹	H
Dr	Root zone depth	m	H
Am	Marshland area	km ²	G
Hp	Maximum channel water depth	m	G
n	Number of channels	-	G
DERIVED QUANTITIES			
Linear channel maximum width		m	G
Marshland max soil water storage		m ³	G
Single channel length		m	G
Total channel length		m	G
Channel max water storage		m ³	G
Drainage density		km km ⁻²	G
Channel volume/ Soil water storage		%	H-G

Although Equation 1 expresses the foundation of the model, containing all the essential components included in WetMAT, as in any hydrological model aiming to represent the water balance of a wetland, the specific equations defining each of the three modules are presented below. The Soil Water Balance is mainly based on the calculation of the soil water content SWC according to Equation 2:

$$SWC = \max (\vartheta_{WP}; \vartheta_{t-1} + P_{t-1} - ET_{t-1})$$

(2)

where P is the precipitation in the previous temporal step and the actual evapotranspiration ET of the previous temporal step is calculated as a function of the potential evapotranspiration ET_0 using the Hargreaves-Samani formula (Hargreaves and Samani, 1985), chosen for its suitability to parsimonious modelling frameworks and its limited data requirements related solely on air temperature, and the soil water extraction according to the following Equation 3:

$$ET = ET_0 * \left(\frac{\vartheta - \vartheta_{WP}}{\vartheta_{FC} - \vartheta_{WP}} \right) \quad (3)$$

where ϑ is the actual water content in the soil [mm m^{-1}], ϑ_{WP} is the soil water content at wilting point and ϑ_{FC} is the soil water content at field capacity [mm m^{-1}]. In case $SWC > \vartheta_{FC}$ the soil produces a surplus, called Drainable Water Flux, which feeds the second component of the model i.e. the Channel Water Balance. The Channel Water Balance is described by Equation 4:

$$CWC = \max(0; CWC_{t-1} + \min(DWF; CWC_{\max} - CWC_{t-1}) - ET * Ap_{t-1} + P * Ap_{t-1}) \quad (4)$$

where CWC [m^3], i.e. the Channel Water Content, is compared to its maximum value $CWC_{\max}$ [m^3], depending on the geometric characteristics of the system. DWF [$\text{m}^3 \text{day}^{-1}$] is the Drainable Water Flux, i.e. the surplus of water which cannot be absorbed by the soil in the Soil Water Balance. Ap is the Channel Water Surface in the previous time step. Unlike Soil Water Balance where soil characteristics are predominant, the Channel Water Balance mainly depend on system geomorphology. The water content in the channels at the previous temporal step and the available Drainable Water Flux are considered. In addition, Equation 4 considers the balance of direct evapotranspiration (ET) and precipitation (P) on the water surface of the channel. The Marshland Water Balance is mainly expressed through the variable

Marshland Flood Volume (MFV) [m^3], represented by the following Equation 5. Again, the Marshland Water Balance is triggered by an “overflow flux” from channel overflow, responding to the condition “ $\text{CWC} > \text{CWC}_{\max}$ ”.

$$\text{MFV} = \max(0; \text{MFV}_{t-1} + \text{COF} + \text{MFA}_{t-1} * P - \text{MFA}_{t-1} * \text{ET} - \text{CLD} - \text{MIL}) \quad (5)$$

In Equation 5 above, COF [$\text{m}^3 \text{ day}^{-1}$] is the “Channel Overflow Flux”, the amount of water flowing from channels once they have reached the $\text{CWC}_{\max}$. MFA stands for “Marshland Flooded Area” and is estimated considering the area of the marshland A_m and the Marshland Flood Depth MFD [m] through an empirical formulation (Equation 6) developed in this study. The equation is simplified but based on real field measurements and observational data reported in previous studies (e.g., Leiva-Piedra, 2024), which adopted more detailed approaches.

$$\text{MFA} = A_m * \text{MFD}^{0.2} \quad (6)$$

Figure 12a) provides a graphical representation of the soil evapotranspiration loss of the entire marshland described in Equation 3, while Figure 12b) depicts the flooding function expressed by Equation 6, which relates the flooded area to the maximum flooding depth attained.

The Channel Lateral Discharge (CLD) [$\text{m}^3 \text{ day}^{-1}$] and the Marshland Infiltration Loss (MIL) [$\text{m}^3 \text{ day}^{-1}$] represent the losses from the marshland. Specifically, CLD refers to the lateral drainage of the channels, taking into account their hydrogeomorphological characteristics. Similarly, MIL addresses the vertical losses from the floods, assuming a clayey soil and therefore a low drainage rate from the marshland bottom. Equations 7 and 8 show how CLD and MIL are determined respectively:

$$\text{CLD} = (\text{MFA}_{t-1} * dd) * \max(0; H_{p_{\max}} - H_{p_{t-1}}) * Df \quad (7)$$

$$MIL = MFA_{t-1} * K \quad (8)$$

where dd is the drainage density [$km\ km^{-2}$], a derived parameter that defines the ratio between the total channel length and the system's surface area; Hp_{max} [m] is the maximum channel water depth and Hp_{t-1} [m] is the channel water depth in the previous temporal step; Df [$m\ s^{-1}$] is the lateral drainage factor. In Equation 8 K [$m\ s^{-1}$] indicates the marshland subsoil hydraulic conductivity. Based on the set of Equations above, the model aims to reproduce in a simplified yet consistent way the marshland flooding and drying processes. The variable MFA [km^2] is calculated on a daily basis, thus reconstructing the temporal evolution of the wetted area even when non measurements are available from remote sensing. Through the analysis of the MFA output is also possible to calculate how many days per year flooding of the marshland is flooded (hydroperiod), so obtaining temporal information about the flooding and draining processes in the system.

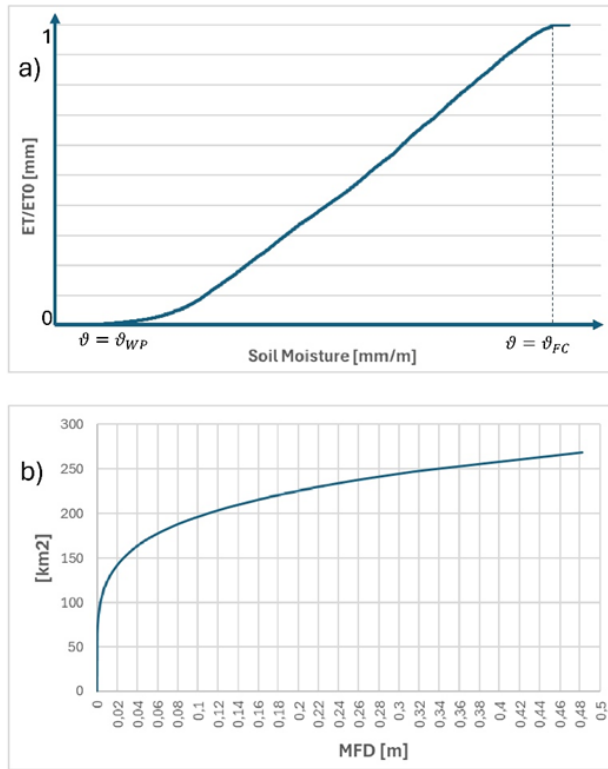


Fig. 12 – Graphical representation of main processes described through WetMAT modules

The geometry of the system is significantly simplified in the model. The Doñana wetland is a flat area with small depressions, which is modelled in WetMAT as a square area (with the actual area of 311 km^2) and a series of channels. The average distance between channels is assumed over the whole wetland area, with each channel having a triangular shape with a side slope of 45° . Figure 13 shows the WetMAT scheme for the Doñana case study as described above. Referring to Equation 1 both I_W and G_W are negligible for the case study since the wetland, because of modifications of the hydraulic system, can be actually considered disconnected from inflow and direct recharge systems from groundwater. For the purpose of the present case study, a vertical leakage rate is only assumed, which depends on the current flooded

area and the existing estimates of the vertical hydraulic conductivity of the subsoil. As it is mainly constituted by clay, its contribution is negligible compared to ET_w , the daily evapotranspiration, which constitutes the main component of the balance. The WetMAT simplified water balance for the Doñana case study, adapted from Equation 1 is hence expressed by Equation 9:

$$\frac{dV_w}{dt} = P - O_w - D_w - ET_w \quad (9)$$

where O_w , the wetland outflow is in this case constituted by the Marshland Infiltration Loss (MIL) component, representative of vertical infiltration losses while the generic term D_w has been defined as Channel Lateral Discharge (CLD), representative of marshland lateral drainage.

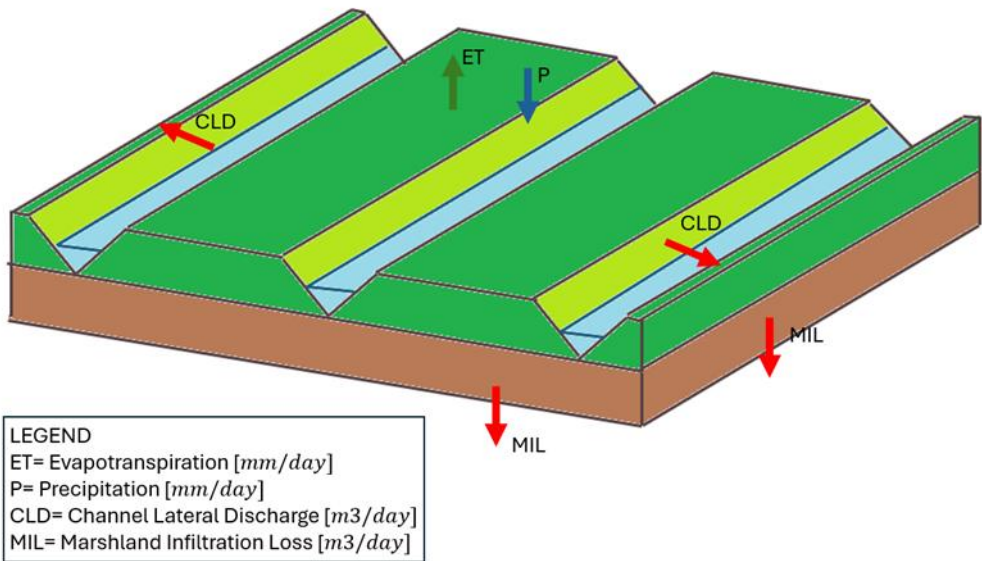


Fig. 13 –WetMAT model scheme for the Doñana case study, with evapotranspiration (ET) and precipitation (P) as main fluxes, deep infiltration losses due (MIL) and the Channel Lateral Discharge (CLD) affecting the marshland drying process

Table 2 shows the ranges of values recommended in the literature for each parameter. According to the geology of the marshland, with a superficial layer of clays and a deeper layer of alluvial sands and gravels (García Novo and Marín Cabrera, 2006), for the ϑ_{WP} a range of 100-300 mm m^{-1} was considered while ϑ_{FC} lies likely in a range between 300 and 500 mm m^{-1} . The lateral drainage factor (Df) should reasonably be assigned values between 10^{-6} and 10^{-3} m s^{-1} according to the average properties of a marshland's system. For the marshland subsoil hydraulic conductivity (K), depending on the type of soil (Shackelford, 2013) a spectrum of values ranging between $2 * 10^{-10} \text{ m s}^{-1}$ and $2 * 10^{-7} \text{ m s}^{-1}$ has been set. Considering the presence of a clay soil and the vegetation that composes the marshland, the root length is not very deep and therefore the root zone depth (Dr) is estimated to be from 0.5 to 1.5 m. Other parameters, depending exclusively on the geometry of the model, were chosen as follows: Hp varies between 1 and 3 m while the number of linear channels was varied from 2 to 11. The marshland area Am , assumed square for simplicity, was not varied as it is known from technical reports (Sánchez Navarro et al., 2009), literature sources (Paredes Losada, 2020; Aldaya et al., 2010) and available data (such as soil use maps). Similarly, the slope of channels banks is assumed to be 45° for modelling simplicity. Table 2 provides full details on the parameters, and includes the “default values”, i.e. reference values for conducting the sensitivity analysis, discussed in next paragraph.

Table 2 – Parameters' ranges and default values for Doñana case study

PARAMETER	UNIT OF MEASURE	ESTIMATED RANGE	DEFAULT VALUE
ϑ_{WP}	mm m^{-1}	100 - 300	200
ϑ_{FC}	mm m^{-1}	300 - 500	330
Df	m s^{-1}	10^{-6} - 10^{-3}	10^{-5}
Slope of banks	deg	45	45
K	m s^{-1}	$2 * 10^{-10}$ - $2 * 10^{-7}$	$2 * 10^{-9}$
Dr	m	0,5 – 1,5	1
Am	km^2	311	311
Hp	m	1 – 3,5	3
n	-	2 - 11	5

2.3 WetMAT sensitivity analysis

This paragraph explores the sensitivity of each parameter in relation to the various dynamics simulated by the model. Through sensitivity analysis, we assess to what extent each parameter influences the different processes and identify which parameters have the most significant impact on each dynamic.

This sensitivity analysis is fundamental as it supports the calibration of parameters, detailed afterwards. It has allowed breaking down the model into components that can be analysed individually, to increase awareness of model limits and sources of uncertainty. We refer, in the following, to three key output variables, namely: the maximum annual flooded area (max MFA) (km^2) of the marshland, the mean hydroperiod (days year^{-1}) and the average annual flooded surface of channels Ap (m^2) variable, which draws attention to the Channel Water Balance. A univariate sensitivity analysis was performed, varying each parameter within its range using regular intervals. The other parameters were set to their default value (see Table 2). The choice to adopt a univariate sensitivity analysis is motivated by the conceptual and parsimonious nature of WetMAT, which is designed to support process understanding through

a limited number of parameters. A one-at-a-time approach enables a clear interpretation of the role of each parameter on the model outputs, while keeping computational requirements and data needs low, in line with the data-scarce conditions that typically characterise wetland environments. The marshland model has been run using precipitation (mm) and temperature (°C) daily data in the period from 1979 to 2018, recorded in the “Palacio de Doñana” meteorological station by EBD-CSIC. From these data the daily ET (mm) has been obtained using Hargreaves-Samani approach (Hargreaves & Samani, 1985).

Figures 14, 15 and 16 represent the sensitivity of the selected output variables to each input variable, referring to the 25th, 50th and 75th percentile. More specifically, Figure 14 shows the impact of all parameters on the maximum annual flooded area of the marshland, normalized by the total marshland area.

Parameters related to soil moisture thresholds, ϑ_{WP} and ϑ_{FC} , certainly have an interesting trend for low values in the chosen ranges. The ($X > 0.25$) percentile shows a significant “threshold” trend in both cases (for values placed in the middle of the chosen range) and in the case of ϑ_{FC} this behaviour is also consolidated for the median. The ϑ_{WP} parameter is directly proportional to the increase of the flooded area while for high values of ϑ_{FC} the flooded area decreases as is normal to expect from a soil that can absorb greater amounts of water. Moreover, the great difference between three percentiles can be, in this case, attributed to the climatic variability of the area in terms of precipitation and evapotranspiration.

The model is quite insensitive to the variation of Df, Hp and n, which are the main geometric parameters of the model. This is not surprising, as the annual maximum flooding occurs when the channels are already full and the area above is uniformly flooded. A limited sensitivity to K and Dr is also observed: as expected, when the hydraulic conductivity increases, the maximum flooded area decreases. The root zone is relevant, as it regulates the emptying phase.

Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes

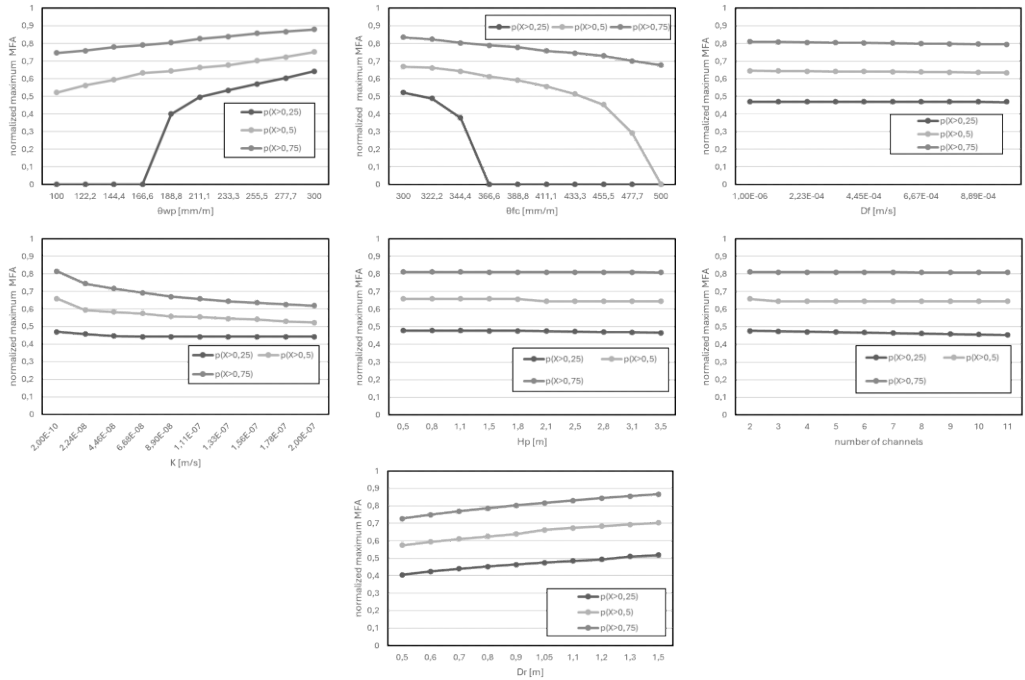


Fig. 14 –Sensitivity analysis for model parameters in the evaluation of maximum MFA variable

Figure 15 proposes the analysis of the hydroperiod behaviour, normalized by the number of days per year. This provides a different perspective, as it focuses on environmental needs over time rather than in space, indicating how many days per year the marshland is flooded. The model shows again a relevant sensitivity to soil moisture parameters (ϑ_{WP} and ϑ_{FC}). Compared to Figure 14, a lower dispersion of data can be noticed. The model sensitivity to H_p , n and D_r is almost negligible, thus highlighting a limited relevance of geometric characteristics. A considerable dispersion of data for a variable number of channels is evident: this occurs because the change of geometry contributes to create a different system in every simulation, thus with a different response to daily climatic outputs.

Interestingly, D_f and K have a limited impact on the dispersion of values, but a high sensitivity (in particular for K) for low values within the selected range.

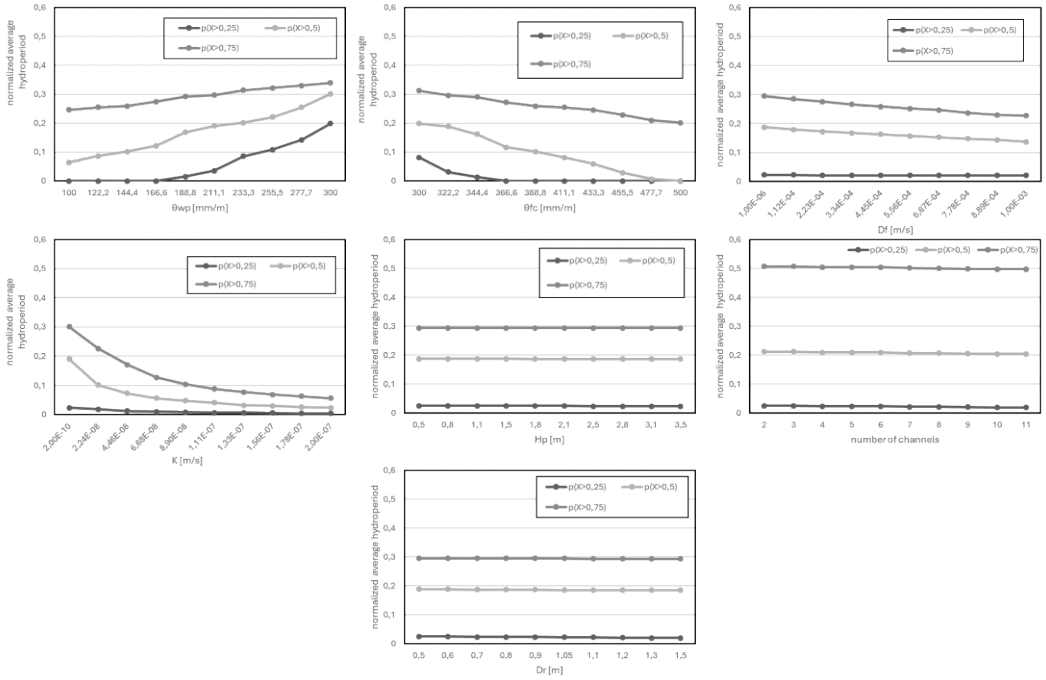


Fig. 15 –Sensitivity analysis for model parameters in the evaluation of mean hydroperiod

Figure 16 shows the sensitivity of the normalized average channel area, which is relevant to describe the process channel flooding and drying. The parameter is normalized by its maximum value found in simulations performed ($1,200,000 \text{ m}^2$).

Also in this case, the moisture related parameters (i.e., ϑ_{WP} and ϑ_{FC}) are highly relevant, yet with a minor data dispersion compared to Figures 14 and 15. Figure 16 also shows that the channel wetted area is insensitive to Df , K and Dr . Conversely, Hp and n play a central role in the process of flooding and drying of channels. Drainage velocity, both lateral and vertical, does not exert a dominant influence on the model results. In contrast, geometric parameters have a stronger impact: variations in Hp produce changes in the output from slightly above zero to about 0.4, while changes in the number of channels result in values ranging from approximately 0.15 to 0.8. The low data dispersion indicates consistent behaviour across simulations, with no evident outliers. Overall, the relationship between these geometric variables and the model outputs is directly proportional.

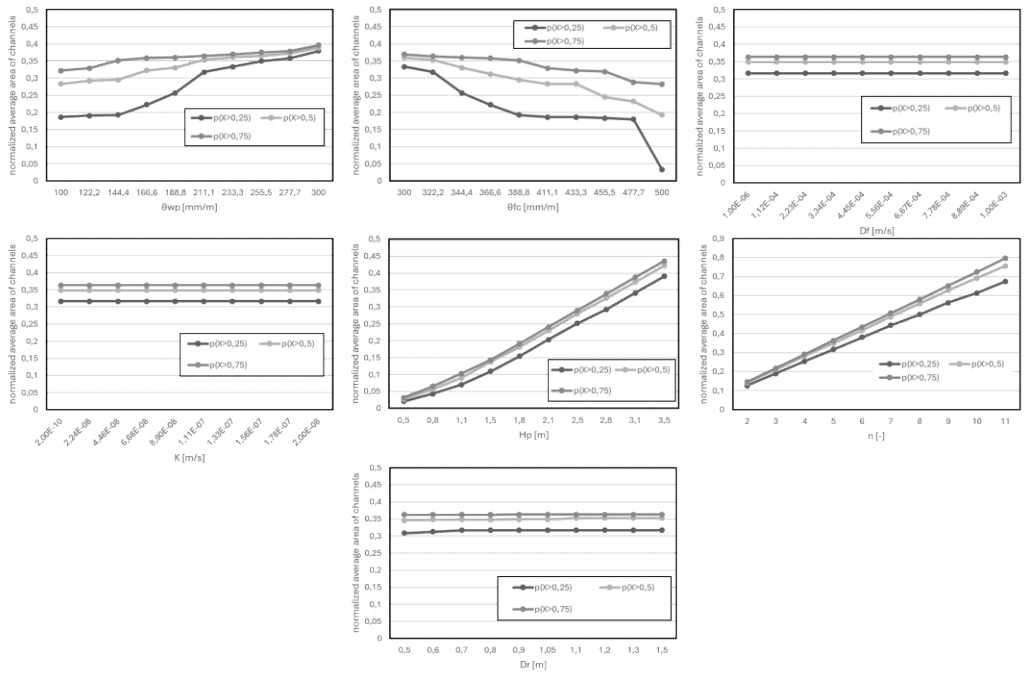


Fig. 16 –Sensitivity analysis for model parameters in the evaluation of average flooded area of channels

The summary of the results of the sensitivity analysis is qualitatively proposed in Table 3. Basically, the selected outputs show a high sensitivity to parameters concerning the soil moisture, so a careful calibration is needed. Conversely the lateral drainage factor shows a very limited impact. Concerning the other parameters, each output shows a different sensitivity. Interestingly, n and H_p (geometry parameters) in the first two dynamics have no sensitivity, but the average channel area is really dependent on them.

Noteworthy is also the case of the depth of the root zone, which affects only the maximum flooded area even if it is located below the system so it might expect to influence the other processes.

Table 3 – Qualitative results of sensitivity analysis for all processes simulated with WetMAT

PARAMETER	ANNUAL MAX MFA	MEAN HYDROPERIOD	AVERAGE CHANNEL AREA
ϑ_{WP}	High	High	High
ϑ_{FC}	High	High	High
Df	Low	Medium	No
K	Medium	High	No
Hp	No	No	High
n	No	Low	High
Dr	Medium	No	No

2.4 WetMAT calibration and validation

The present section deals with model calibration and validation, which has been performed comparing model outputs (annual maximum flooded area MFA and average hydroperiod of the marshland) with observed data. Regarding the MFA, reference was made to observed data from the 1980-81 hydrological year to 2017-18 (Green et al., 2024). The observed data for the hydroperiod were instead extrapolated from a study based on satellite images processing in the period 1980-2014 (Díaz-Delgado et al., 2016). Regarding the latter, it is worth to highlight a difference in how the variable "hydroperiod" is calculated: while in the observed series it is calculated based on the pixels assumed flooded in the processed satellite images and counting the number of days per year during which the pixels are flooded, from which a mean value for the entire area is then deduced, WetMAT model does not provide a spatial representation of the wetland, which is therefore considered as a single pond and the hydroperiod variable consists of an estimation of the days when the model has a non-zero flooded area value.

Figure 17 summarizes the main steps in the calibration and validation processes of the WetMAT model, described in further details afterwards.

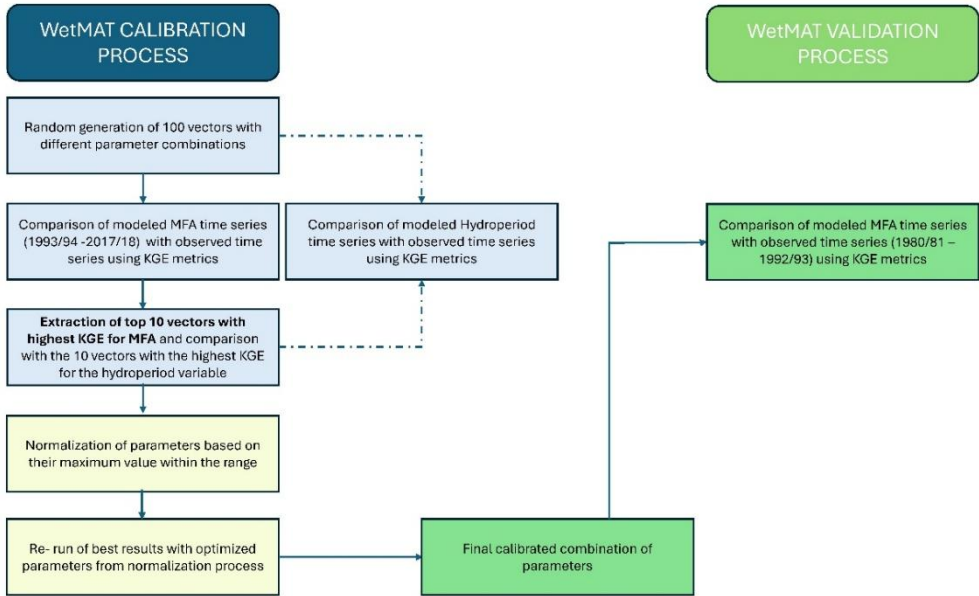


Fig. 17 – Main steps involved in the calibration and validation processes of the WetMAT model

The calibration process started with the random generation and combination of input parameters (except the marshland area and the slope of the channels that were kept constant) in 100 simulations. Results were then compared with observation using the Kling–Gupta Efficiency (KGE) metric (Gupta et al., 2009; Liu, 2020), which is one of the most used measures in hydrological modelling. The KGE is a performance metric that combines three fundamental aspects of the comparison between simulated and observed data: correlation, bias, and relative variability. Compared to other commonly used indices, KGE enables a balanced evaluation of model behaviour, preventing strong performance in one component from concealing deficiencies in others. For this reason, it was selected as metric for the calibration and validation phases in this work, as it provides a comprehensive and robust assessment of model reliability. The historical time series used for the calibration phase spans from the 1993-94 hydrological year to the 2017-18 hydrological year for the MFA variable. The hydroperiod time series, used solely as support for the validation process, covers the hydrological years from 1980-81 to 2013-14. The KGE values for the MFA vary significantly be-

tween 0.13 and 0.82. The KGE values for the hydroperiod are instead below 0.54 but results are consistent with those of the maximum annual flooded area, as 70% of the ten simulations with highest KGE values for the MFA coincide (although not in the same order) with the ten simulations having the highest KGE values for the hydroperiod. These results are valuable, particularly considering the limited availability of historical time series for comparison, as the only datasets available for the maximum MFA and hydroperiod variables are those used in this study, with no other relevant literature providing comparable datasets. An additional analysis was then performed to further support model calibration: the ten high-ranked simulations in terms of KGE for the maximum flooded area were analysed, normalizing the parameters with respect to the maximum value in the range. Most of the parameters chosen within these simulations were in the middle of their range; however, the values of ϑ_{WP} and K fell respectively in the upper and lower bounds of their ranges. The ten best random simulations have been therefore re-run, replacing the ϑ_{WP} and K values with extreme values. As a result, higher values of KGE (0.85) were obtained for the maximum flooded areas. The set of calibrated parameters is reported in Table 4 and used for WetMAT simulations. Figure 18 a) shows the comparison between the observed and calculated values of the maximum MFA.

The validation phase of the WetMAT model involved comparing historical time series of the maximum MFA variable, specifically the series generated by the WetMAT model with that from (Green et al., 2024), covering the period from the 1980-81 to the 1992-93 hydrological years. While the relatively short length of the time series represents a limitation, the obtained KGE value of 0.73 suggests a satisfactory model performance. Figure 18 b) shows results of the validation process.

Table 4 – Calibrated vector for WetMAT model

ϑ_{WP} [mm m ⁻¹]	ϑ_{FC} [mm m ⁻¹]	Df [m s ⁻¹]	K [m s ⁻¹]	Dr [m]	Hp [m]	n [–]
300	426	$8,14 \cdot 10^{-4}$	$2 \cdot 10^{-10}$	1,41	1,25	5

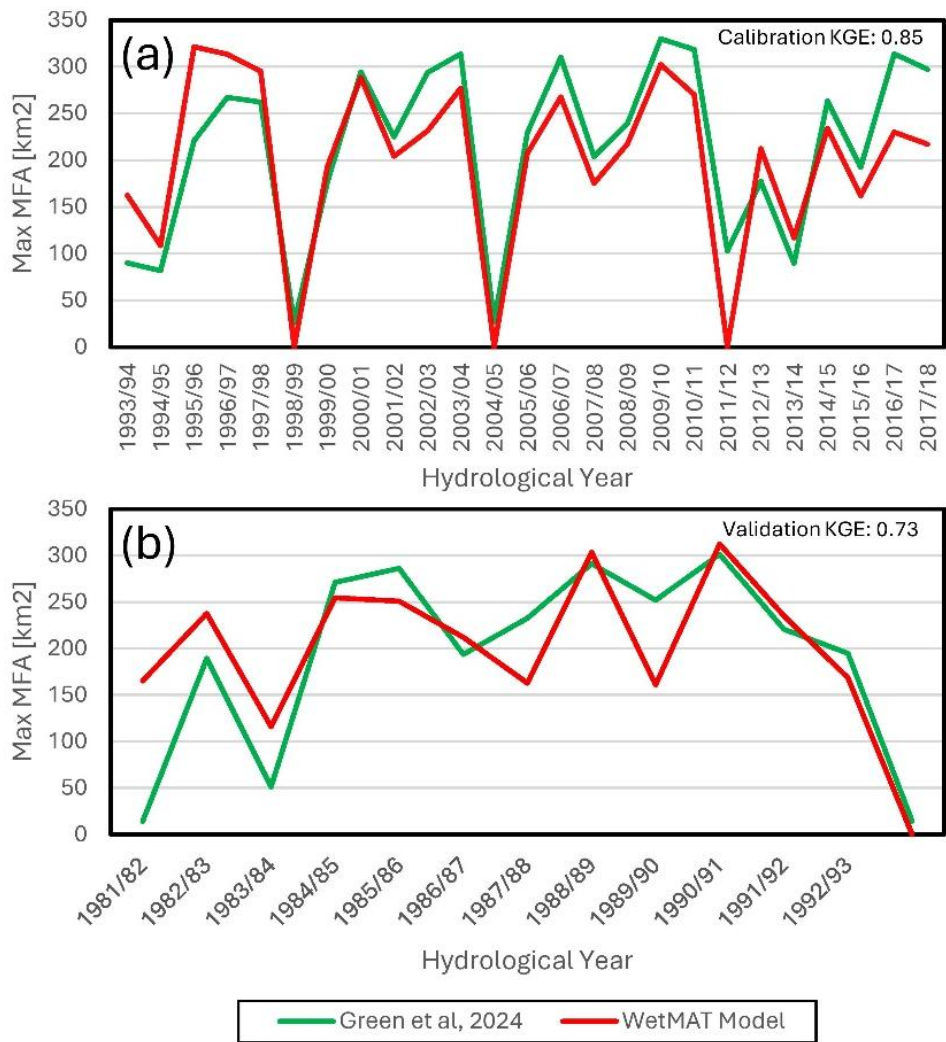


Fig. 18 – Panel (a) shows the WetMAT model calibration according to the “maximum MFA” variable. Panel (b) shows the WetMAT validation according to the “maximum MFA” variable

2.5 WetMAT outputs

This section aims to describe the main outputs that WetMAT can generate. Among these, particular attention is given to those used during the calibration and val-

idation processes, as they were considered the most relevant for the research objectives. Corresponding applications are presented and discussed below. First, the model produces outputs at a daily time step, which, as previously discussed, provides sufficient temporal resolution to capture the dynamics of a temporary wetland. Although the applications presented here primarily focus on the estimation of the Marshland Flooded Area (MFA) and the hydroperiod, thereby representing the wetland as a whole, it is also possible to analyse specific components such as daily soil moisture dynamics, expressed as Soil Water Content (SWC) [mm], or channel flooding, expressed as Channel Water Content (CWC) [m³/day]. Figure 19 presents a long-term plot of these outputs, illustrating the model's behaviour: when the SWC output reaches its peak, channel activation becomes evident through the corresponding increase in channel filling. Furthermore, once the channels reach their maximum capacity, the figure also displays the associated MFA response.

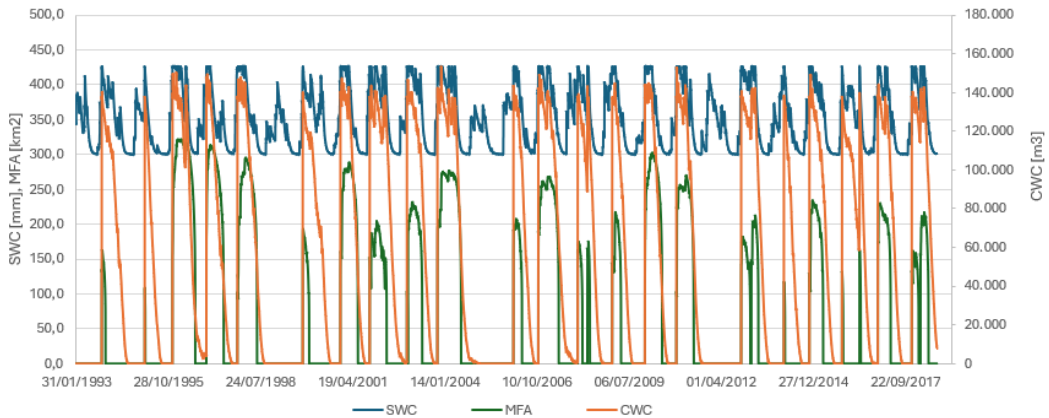


Fig. 19 – Time representation of WetMAT soil balance (SWC, mm), channel balance (CWC, m³/day) and marshland balance (MFA, km²) outputs

Another output of particular interest is the temporal evolution of water depth during marshland flooding. Figure 20 presents this variable over time, in relation to the Marshland Flood Area (MFA). As the model adopts a lumped configuration, treating the marshland as a single unit, it does not resolve spatial variability in depth;

therefore, the Marshland Flood Depth (MFD) values should be interpreted as spatial averages over the entire area.

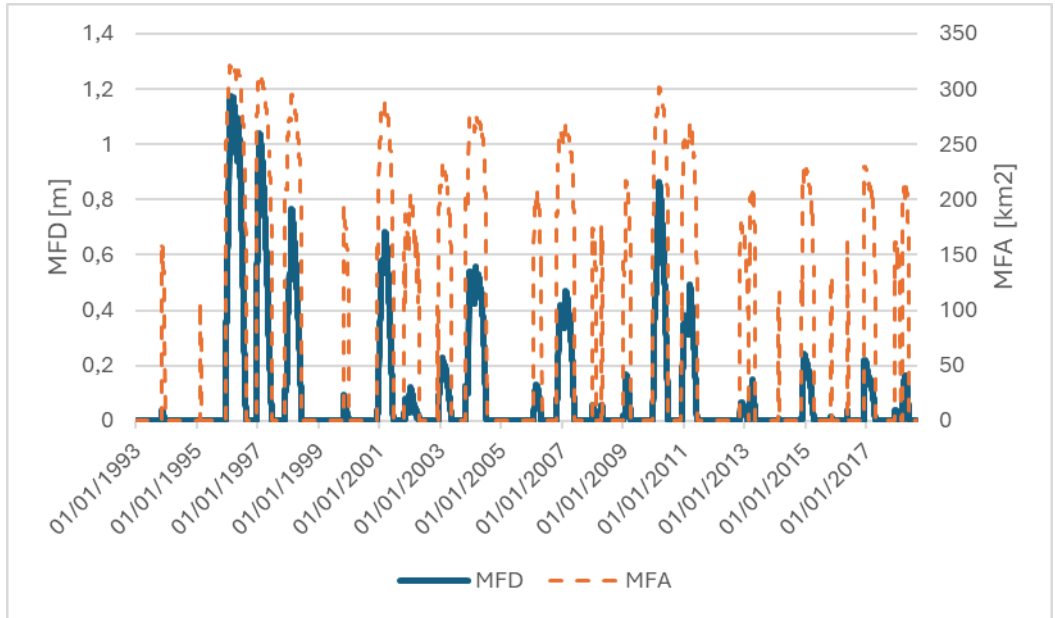


Fig. 20 – Time representation of WetMAT outputs related to flooding depth (MFD) and flooding area (MFA)

The MFA output allows the calculation of the hydroperiod which is fundamental for the purposes of the model. Some examples are shown in Fig. 21 where the recorded rainfall is represented as well. More specifically, Fig. 21 a) shows the hydrological year 2005-06, that can be considered a dry year, as the precipitation value (468.3 mm) is close to 25th percentile of the analysed series (440.25 mm). Figure 21 b) instead refers to the hydrological year 2006-07, i.e. a wet year, as the precipitation value (716.9 mm) is the closest to the 75th percentile in the analysed series (1992-93, 2017-18). Figure 21 c) then represents the output of the model over an average hydrological year (2014-15), characterized by a total annual rainfall of 531.85 mm. The flooding period obtained through WetMAT generally begins in autumn and ends in late spring and, despite a high variability, it is consistent with literature evidence (Bustamante et al., 2009).

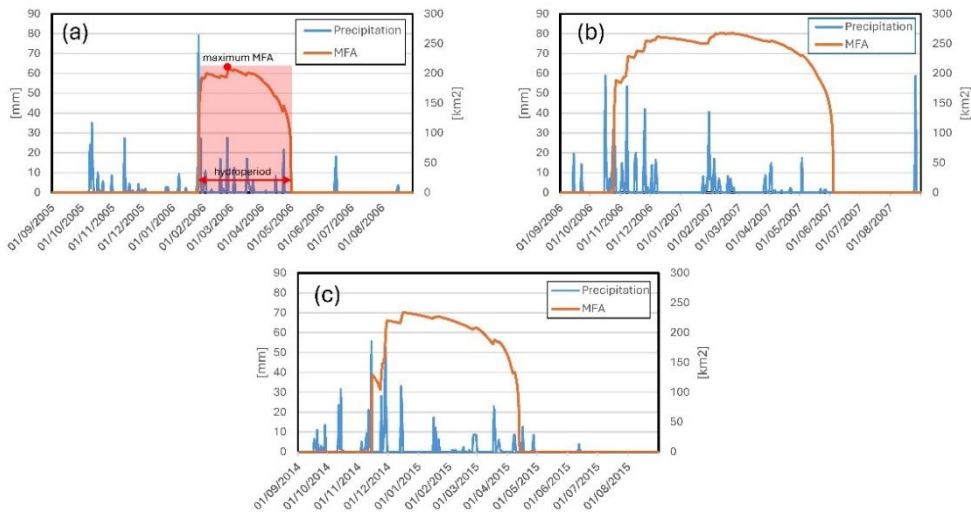


Fig. 21 – Panel (a) shows the daily plot of flooded areas in hydrologic year 2005-2006 taken as representative of dry year. Panel (b) shows the daily plot of flooded areas in hydrologic year 2006-2007 taken as representative of wet year. Panel (c) represents flooded areas in hydrologic year 2014-2015, taken as representative of an average hydrologic year. All panels show daily precipitation during the hydrologic year

As previously discussed, the MFA variable facilitates both spatial and temporal analysis of flooding dynamics in the marshland. While the hydroperiod and maximum annual MFA are somehow correlated, there is no evidence of mutual dependency, and therefore is essential to consider both variables when assessing marshland hydrological state from year to year. The importance of incorporating both temporal and spatial characteristics in hydro-ecological studies has been widely acknowledged (Peng et al., 2022; Coleman et al., 2015), particularly in the context of temporary wetlands (Rawat et al., 2025). In this study, therefore, a composite variable, hereafter referred to as the Inundation Persistence Index (IPI), is introduced, effectively integrating the concepts of maximum MFA and hydroperiod. IPI is designed to quantitatively highlight both aspects that, from the hydrological point of view, reflect the state of a temporary wetland throughout the year. This composite variable is given by the product between maximum MFA and hydroperiod and can be easily graphically represented (see Figure 21 a). The relationship between the IPI variable and the annual rainfall (pro-

posed in Figure 22) shows that IPI is equal to or very close to zero in all years with precipitation below around 400 mm/yr. Figure 22 shows also that if the annual precipitation goes above the 400 mm there is non-linear correlation between the two variables.

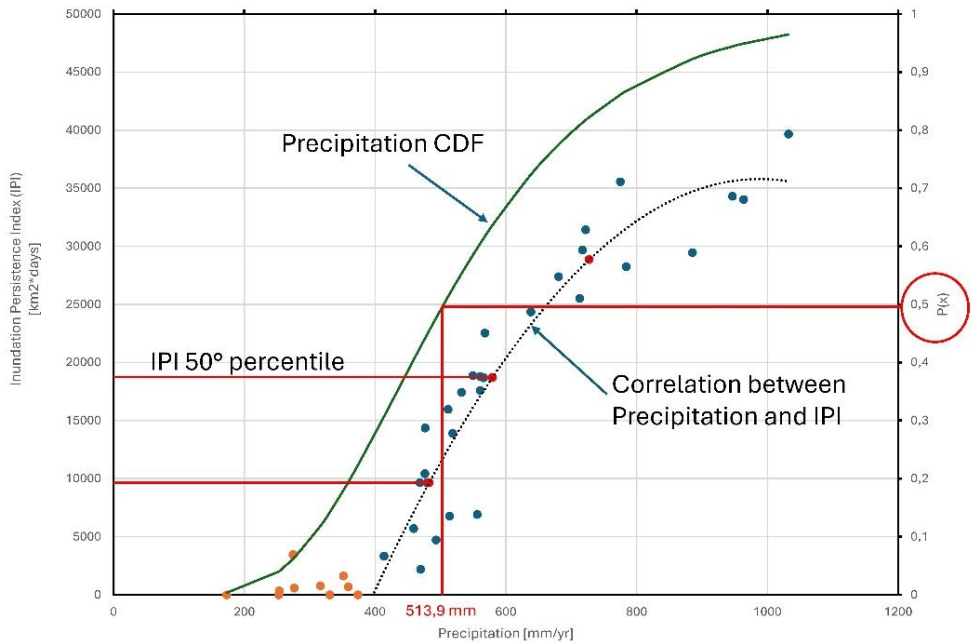


Fig. 22 –Precipitation log-normal CDF and correlation between Precipitation and Inundation Persistence Index (IPI)

Figure 22 also proposes a direct comparison between the cumulative density function (CDF) of precipitation and the curve relating the IPI to annual precipitation. Interestingly, there is no direct correlation between the 50th percentile of both series. This means that even in an average year (in terms of rainfall) represented by a rainfall of $513,9 \text{ mm year}^{-1}$ a reduced marshland flooding is likely to occur with an IPI below its 50th percentile. This hydrological condition of the marshland (reflected by $\text{IPI} < 20,000 \text{ km}^2 \cdot \text{days}$) has been therefore assumed as a quantitative measure of the water-related ecosystem state for which a direct connection with its water requirement was established. In other terms, the marshland condition when IPI is below its aver-

age has been adopted to characterize the water deficit affecting the ecosystem in a given year.

To quantify the amount of water needed by the marshland to reach its average flooding conditions, the correlation between annual precipitation and IPI and then between the IPI and the annual water inputs to the marshland, namely the Cumulate Inflow (Mm^3), are determined. This correlation is shown in the abacus of Figure 23, developed using both a second-degree equation ($R^2 = 0.89$) and a third-degree equation ($R^2 = 0.95$). Those equations could be adopted to estimate, on a yearly basis, how much water is needed to stabilize the marshland flooding.

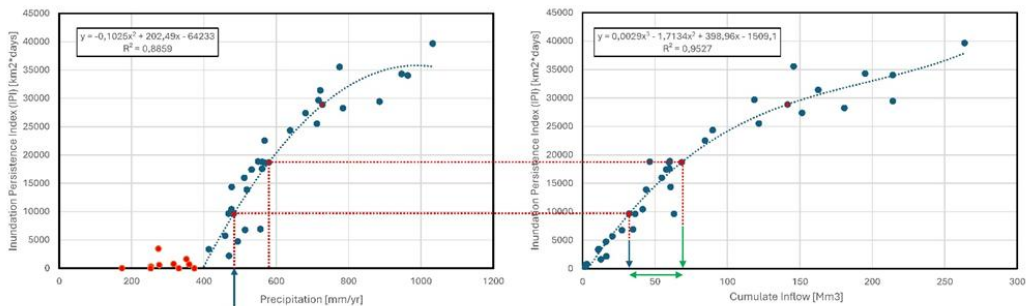


Fig. 23 – Determination of annual water inputs to the marshland

Using the abacus shown in Figure 23, it is possible, given the value of annual rainfall for each hydrological year, to estimate the cumulative inflow to the marshland based on WetMAT results. This approach makes it possible to identify dry years for the wetland according to the volume of cumulate inflow and to quantify the additional water input required to reach average flooding conditions.

Figure 24 shows, based on annual rainfall values from the 1979–2018 historical series, the external water volumes to be added to the system under different precipitation conditions, estimated through equations of the curves in Figure 23. As indicated in Figure 23, a cumulative inflow of 68.4 Mm^3 is required for the wetland to reach average flooding conditions. On average, an additional 19 Mm^3 is needed, corresponding to 27% of the total.

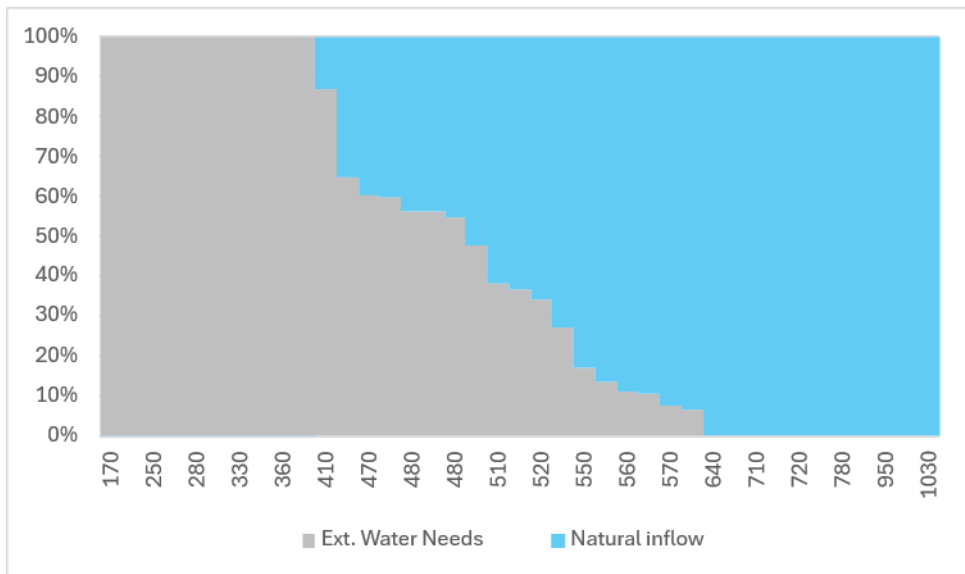


Fig. 24 – Representation of the wetland's water requirements to achieve the established average flooding conditions, starting from annual precipitation values.

To provide a temporal perspective on the issue, an estimate of return periods (in years) was made, i.e., the average interval at which a given amount of external water must be supplied to reach average inundation. Figure 25 shows that approximately every two years, 25 Mm³ must be added to the wetland to ensure adequate flooding, a volume already higher than the average requirement shown in Figure 24. It is also worth noting that the maximum observed external water requirement has a return period of four years, highlighting the strong climatic variability and, consequently, the complex hydrological dynamics of the area.

This frequency-based approach to evaluating external water needs for ecological and hydrological support has already been applied in recent studies (Yu et al., 2021), including in geographic contexts closely related to the present case study, such as Las Tablas de Daimiel (Canto-Perello et al., 2021). These applications underscore the necessity of developing indices such as the IPI in order to compare issues in similar contexts.

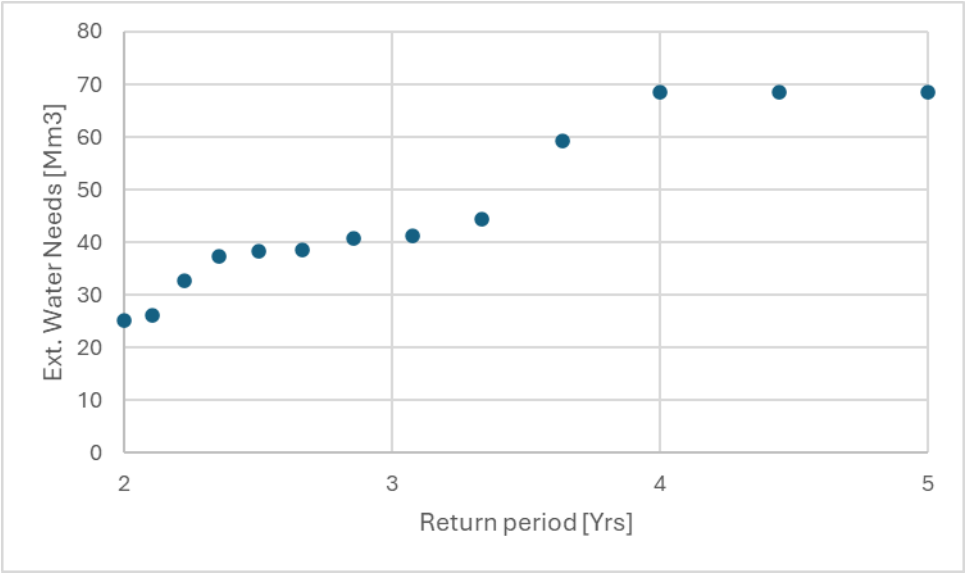


Fig. 25 – Relation between return period (years) and the external water volume (Mm³) required to re-store average flooding conditions in the wetland

3 ADAPTING THE WETMAT MODELLING FRAME- WORK TO A SMALLER SCALE: THE TORRE GUACETO WET- LAND

While the previous chapter presented the implementation of the WetMAT model on a large-scale case study, yet with several modelling simplifications, the present chapter details an application to a small-scale wetland, i.e. the Torre Guaceto wetland (SE Italy). The relevance of the case study is mainly related to the conflict between agricultural activities and environmental conservation, and related impacts on water resources. Torre Guaceto area is approximately 300 times smaller than the Doñana wetland, so it requires a higher level of detail in the representation of hydrological dynamics, which are further complicated by the interaction between surface water and groundwater. Following an introduction to the study area and its main challenges, this chapter describes the configuration of WetMAT for this case, the modelling procedures adopted, and the calibration and validation processes. The main model outputs are presented afterwards and used to explore potential scenarios for the sustainable management of the wetland.

3.1 Overview on the case study

The Torre Guaceto wetland is part of the Torre Guaceto State Nature Reserve, located along the Adriatic coast of Apulia region, a few kilometres north of Brindisi in southern Italy. The Nature Reserve covers an area of approximately 11 km² and comprises several ecosystems, including the wetland covering about 1.2 km². The protected area also includes a marine area of 22 km², designated in 1991 by a decree of the Italian Ministry of the Environment. Although it lies near the mouth of the Canale Reale, the wetland has no direct hydraulic connection with it, as the two systems are separated by artificial embankments. The wetland is primarily fed by rainfall and freshwater springs and can therefore be classified as a groundwater-fed system. It is classified as a permanent wetland, since neither complete drainage nor substantial

reductions in flooded area are observed throughout the year. Figure 26, from Passarella et al. (2025), provides an overview of the study area.

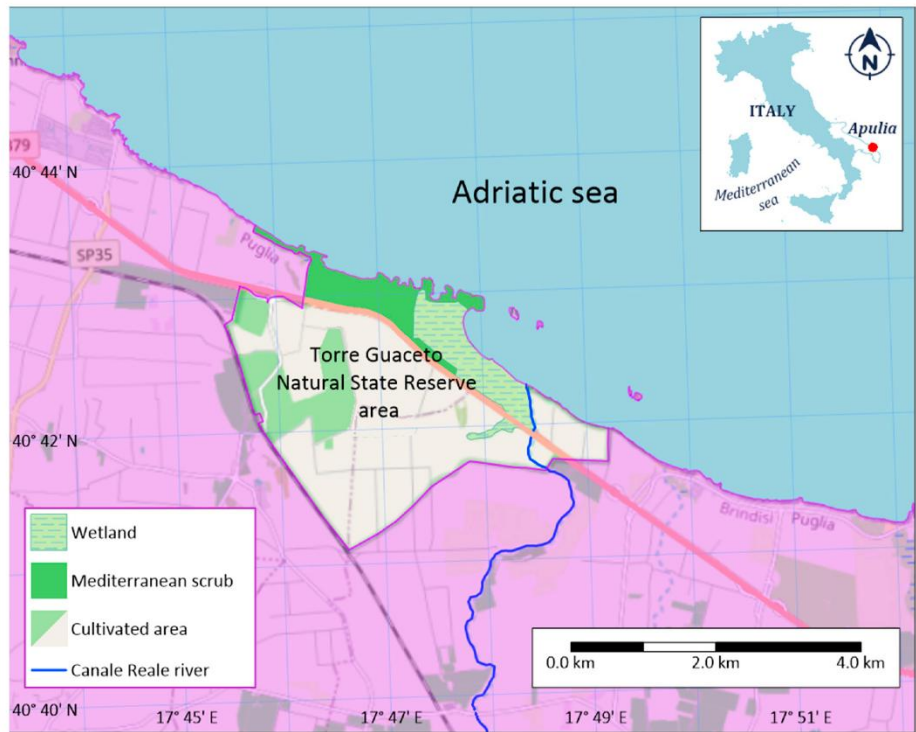


Fig. 26 – Map of Torre Guaceto area. Source Passarella et al. (2025)

From a hydrogeological perspective, the main aquifer system feeding the wetland ponds is a deep carbonate aquifer hosted in karstified and fractured Cretaceous formations, primarily recharged by precipitation. Due to its coastal location, the extensive groundwater abstraction for agricultural use, and the additional pressure exerted by climate change through changes in the temperature and precipitation regime or through phenomena such as rising sea levels (Doglioni e Simeone, 2019; Passarella et al., 2020), the aquifer is increasingly affected by marine intrusion, leading to a progressive inland migration of the saltwater wedge. This salinization process severely impacts the wetland ponds, which, as previously mentioned, are hydraulically connected to the groundwater system.

The conceptual framework presented in Figure 27 (Passarella et al., 2025) illustrates that the salinization process establishes a self-reinforcing feedback loop affecting both the aquifer and the wetland. In addition to the previously mentioned role of marine intrusion, soils irrigated with brackish groundwater release salts that percolate downward with rainfall infiltration. Furthermore, the contribution of marine aerosols exacerbates the overall salinization dynamics.

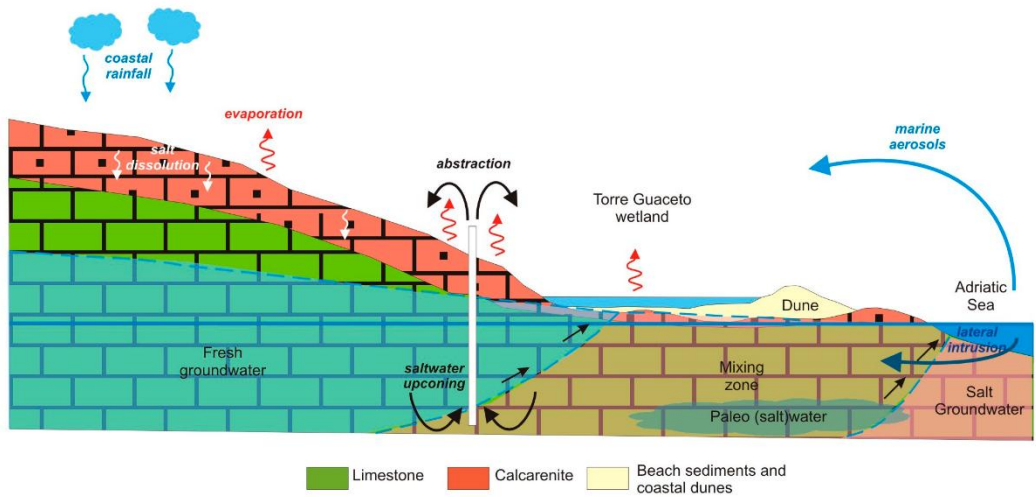


Fig. 27 – Conceptual representation of main salinization dynamics on the area. Source: (Passarella et al., 2025)

The protected area comprises both zones of high natural value and others characterised by an agricultural landscape. The natural section includes diverse habitat types, such as rocky and sandy coastal environments, forest habitats, Mediterranean scrub, and wetlands (Marzano et al., 2012). From an ecological standpoint, the Torre Guaceto wetland is characterized by high biodiversity and provides several essential ecosystem services including water purification, carbon sequestration, flood regulation, and ecological support for numerous species, such as flamingos and herons. Moreover, it serves as an important stopover site along major migratory bird routes. For such reason, the area has been subject to multiple protection measures. In 1981 it was designated as a Wetland of International Importance under the Ramsar Convention. In 2000, the terrestrial protected area (State Nature Reserve) was officially es-

established, and in the same year a management consortium was formed by the neighbouring municipalities of Carovigno and Brindisi, in partnership with WWF Italy. This development, however, triggered conflicts and debates with local landowners, in view of potential restrictions on land use (Russi, 2020). Torre Guaceto has received attention also in recent years: for instance, it was included in the National List of Protected Areas in 2010, and in 2013 the Management Plan and the Implementing Regulation for the Nature Reserve were formally approved.

In the areas surrounding the Torre Guaceto wetland, agricultural activities have long been present, traditionally focusing on olive cultivation, vegetable crops such as artichokes and tomatoes, and vineyards. The main source of conflict with the natural environment is not related to land use, which is largely regulated and protected. It rather relates to groundwater, and particularly its quality, which is threatened by agricultural pollution and salinization. Salinization, in turn, also affects agriculture itself, reducing both crop yields and soil fertility.

Further pressure on the aquifer state occurred in recent years, after the spread of *Xylella fastidiosa*, a bacterium that devastated olive groves across southern Apulia, leading to the death of thousands of century-old olive trees. As a response, infected olive trees have been replaced by annual cash crops, which are typically characterised by a significantly higher water demand.

Considering the value of the natural landscapes and the high biodiversity, the area is also an important ecotourism destination. Activities such as birdwatching contribute to funding conservation initiatives and research, both in marine and coastal ecosystems and within the wetland itself. Nevertheless, while agricultural abstraction remains the primary anthropogenic threat, concerns have also been raised about groundwater use to support seasonal demand related to tourist fluxes, which could exacerbate existing pressures. For all these reasons, effective hydrological modelling tools are needed for understanding system state and for developing sustainable water management scenarios across multiple sectors.

3.2 *WetMAT configuration and main features in Torre Guaceto case study*

As shown in the flowchart presented in Figure 28, the fundamental process underlying WetMAT (already detailed in Section 1.5), based on three interconnected modules for the representation of the hydrological balance, remains essentially the same, and the model relies on the same climatic input variables. An additional input variable, namely “groundwater recharge”, has been added in the Torre Guaceto case study since the aquifer contributes both to the soil and the wetland dynamics. In this regard, as Torre Guaceto is a permanent wetland, an additional difference from the configuration shown in Figure 11 concerns the third module, which is referred to as the ‘pond balance’.

Regarding the Channel Water Balance, WetMAT includes in this case both a ‘sea discharge’ and a ‘pond discharge’. This distinction, which will be clarified afterwards, highlights that in a small-scale and hydrologically complex context such as Torre Guaceto, WetMAT evolves from a strictly lumped structure to a semi-distributed configuration. In this framework, the soil and channel modules remain unchanged.

Figure 28 highlights also the most significant WetMAT outputs i.e. the pond level and pond salinity, for model calibration, validation, and scenario analysis development. Section 3.4 will detail additional outputs that can be derived from the model.

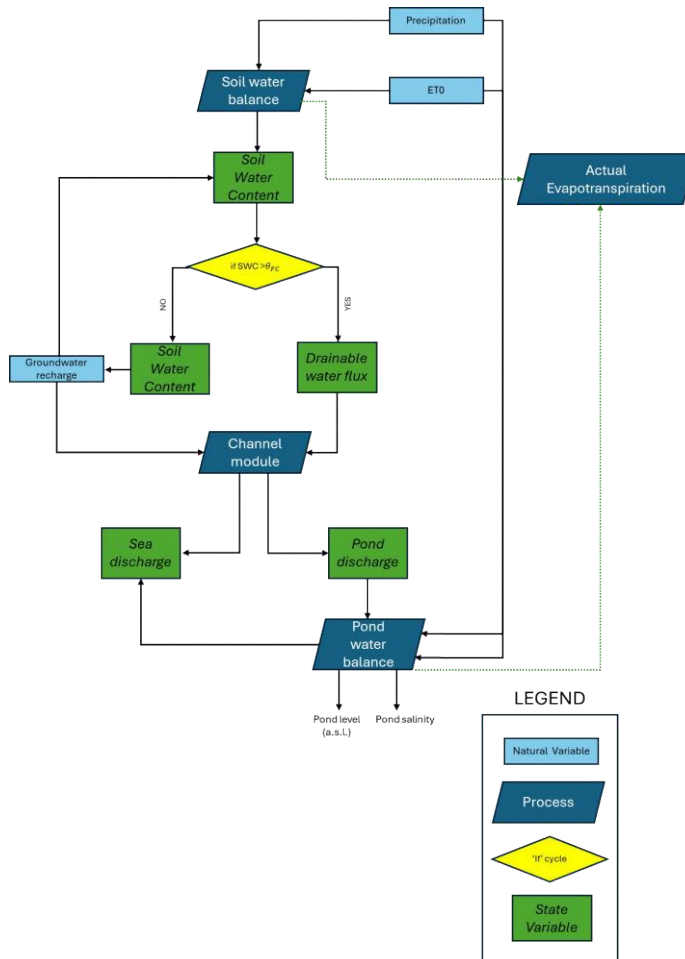


Fig. 28 –Flowchart of WetMAT model in Torre Guaceto case study

To better understand the modelling assumptions, Figure 29 presents a Digital Terrain Model (DTM) of the study area, taken from the Regional Wildfire Prevention Plan of the Torre Guaceto State Nature Reserve (2020–2024) (Consorzio di Gestione di Torre Guaceto, 2018). The figure highlights the dense network of drainage channels crossing the area, dating back to the land reclamation works carried out during the Fascist period (1920s–1930s). These channels were constructed to drain groundwater directly into the sea through outlets located at the points with the lowest elevation in the area, identified in Figure 29 as A and B. In the area B water freely discharges into the

sea, flooding events are rare and limited to small volumes and short durations following intense rainfall. Conversely, in the area A, a permanent pond has formed, most likely as a result of the obstruction of the drainage outlet. This pond represents the core of the wetland system and constitutes the focus of the third WetMAT module.

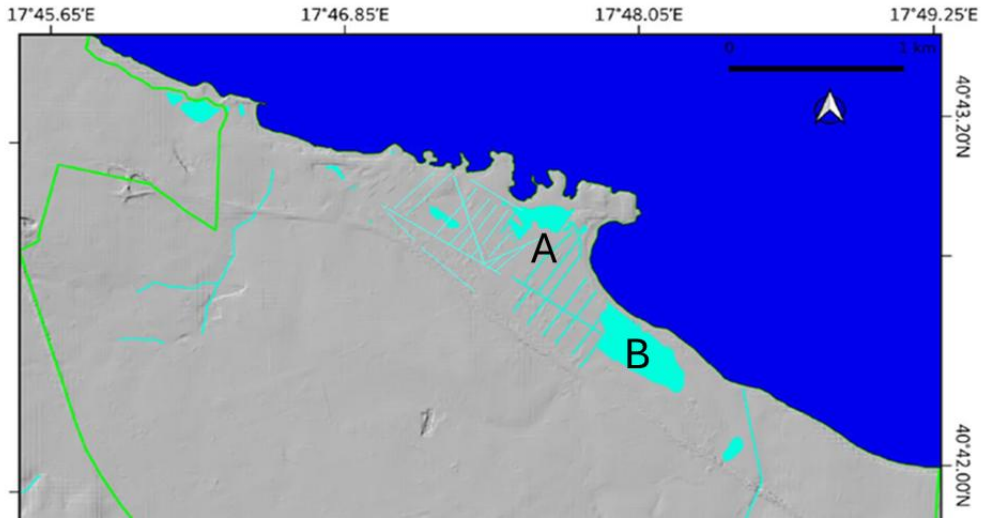


Fig. 29 –Digital Terrain Model of the study area. Source: Regional Wildfire Prevention Plan, Consorzio di Gestione di Torre Guaceto, (2018). In the figure, the green line delineates the boundary of the protected area. Blue areas represent the sea, while light blue areas indicate low-lying zones prone to flooding, including drainage channels, watercourses, and ponds.

Based on this overview, Figure 30 presents a conceptual scheme of the system, focusing exclusively on the basin associated with the pond. Evapotranspiration and precipitation act directly both on the channels network and on the pond itself. The aquifer drains into the channel system, which in turn flows into the pond. The pond also has a direct discharge into the sea, which will be detailed in the following section.

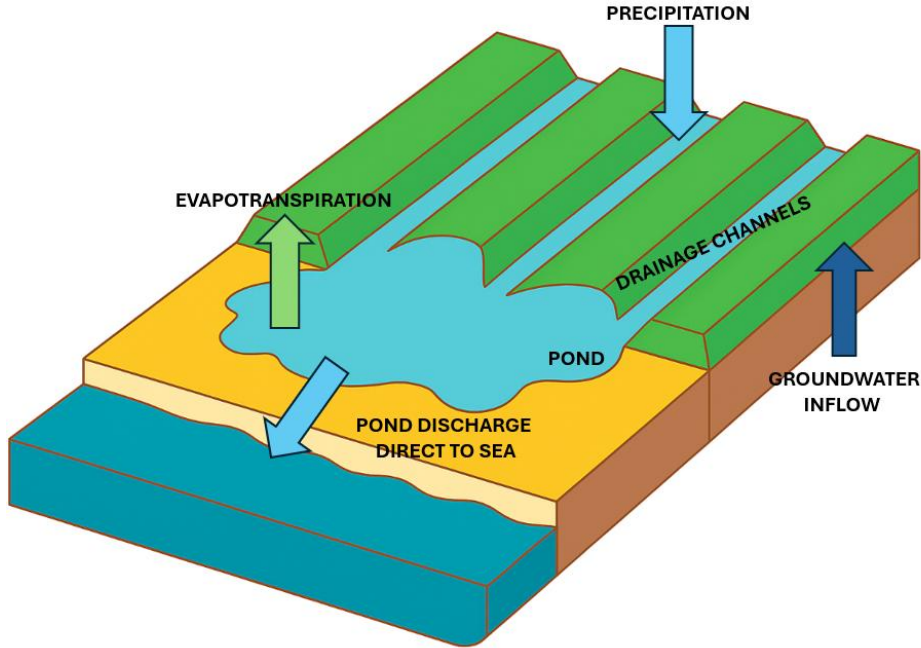


Fig. 30 – WetMAT model scheme for Torre Guaceto case study, with its main components

The *Soil Water Balance* in WetMAT analyses the water balance in the soil vadose zone and it is substantially based on the calculation of the soil water content SWC [mm] according to Equation 10:

$$SWC = SWC_{(t-1)} + P_{(t-1)} - ETa_{(t-1)} + G_c \quad (10)$$

where $SWC_{(t-1)}$ is the soil water content in the previous temporal step, $P_{(t-1)}$ [mm] is the precipitation in the previous temporal step, $ETa_{(t-1)}$ [mm] is the actual evapotranspiration during the previous temporal step and G_c [mm] represents the groundwater component in the soil water content, which is only activated if the soil moisture is not in its optimal water availability zone, according to the Feddes approach for the crop water stress (Feddes and Zaradny, 1978). The following Equation 11 expresses the calculation of ETa using the Feddes approach:

$$ET_a = ET_0 * (\max(0; \frac{\min(\vartheta - \vartheta_{WP}; \vartheta_{RAW} - \vartheta_{WP})}{\vartheta_{RAW} - \vartheta_{WP}}) \quad (11)$$

where ET_0 is the daily reference evapotranspiration, calculated from daily temperature data using the Hargreaves- Samani approach (Hargreaves and Samani, 1985), ϑ_{WP} is the soil water content at wilting point capacity, ϑ represent the actual soil capacity and ϑ_{RAW} is the soil water content's threshold for its optimal water availability zone. The actual soil capacity ϑ is calculated as shown in Equation 12:

$$\vartheta = \min(\vartheta_{FC}; SWC) \quad (12)$$

where ϑ_{FC} is the soil water content at field capacity and SWC is the soil water content as calculated in Equation 10. The parameter ϑ_{RAW} , is calculated as follows:

$$\vartheta_{RAW} = \vartheta_{FC} - p (\vartheta_{FC} - \vartheta_{WP}) \quad (13)$$

where p represents the soil water depletion fraction under no stress conditions (Allen et al., 1998).

As previously mentioned, the G_c component is only activated in the soil water balance if rainfall recharge is not sufficient to keep moisture, according to Equation 14:

$$G_c = \max(0; \min(G_d; ET_0 - ET_a)) \quad (14)$$

where G_d [mm] is the daily groundwater discharge in the system, a calibrated quantity that directly influences the Drainable Water Flux DWF [mm], the *Channel Water Balance* input, described as follows in Equation 15:

$$DWF = (SWC - \vartheta) + (G_d - G_c) \quad (15)$$

This component reaches its maximum values during the winter season, as the recharge generated by precipitation alone is able to fully feed the soil balance, and often even creates a surplus, which is added to G_d .

In the Channel Water Balance analysis, the DWF (which is derived from the Soil Water Balance) is conveyed through the drainage channels towards its final destination, either the sea or the groundwater-fed pond. From this point onward, the model works as a semi-distributed system, and the estimation of drained flow relies on the classical formulations of drainage theory. In this case, these formulations are applied in an inverse way, as the objective of WetMAT is to estimate the drained flow at each time step (i.e., on a daily basis), rather than to determine the hydraulic characteristics of the system, which are instead derived from field measurements and analysis of the existing channels. In order to consider a time-varying DWF input as can be seen from Equation 15, De Zeeuw and Hellinga's approach (Štibinger, 2009) was used in the modelling, which considers a non-uniform distribution of the groundwater input to the drainage system, assumed to be constant only in the single time interval, which in this case is one day.

The following Equation 16 expresses De Zeeuw Hellinga's classical formula for unsteady-state:

$$q_t = q_{t-1}e^{-\alpha\Delta t} + DWF (1 - e^{-\alpha\Delta t}) \quad (16)$$

where q [$m\ day^{-1}$] is the drained discharge from groundwater to channels and α [day^{-1}] it is a reaction factor that considers hydraulic and physical characteristics of the system and will be discussed afterwards in Equation 18.

Equation 16 is used by WetMAT in its version:

$$h_t = h_{t-1}e^{-\alpha\Delta t} + \frac{DWF}{0.8\mu\alpha} (1 - e^{-\alpha\Delta t}) \quad (17)$$

in which h_t [m] is the height of the water table at the maximum distance from channels at time t and μ [—] is the soil drainable pore space. Equation 17 is valuable in

our case as it adds to De Zeeuw Hellinga's approach the simplification introduced by Hooghoudt's formula for the steady state analysis (Ritzema, 1994) whereby through Equation 17 it is possible to obtain the estimation of q [$m\ day^{-1}$] for each time step as follows in Equation 18:

$$q = \frac{8\ K\ D}{L^2} h = \frac{8\ \mu\ \alpha}{\pi^2} h \approx 0,8\ \mu\ \alpha\ h \quad (18)$$

where D [m] is the water depth in the drainage channel during the specific temporal step, K [$m\ day^{-1}$] is the soil hydraulic conductivity and L [m] is the distance between two drainage channels.

It is relevant to specify for the sake of completeness that both De Zeeuw Hellinga's approach for unsteady state in Equation 16, and the simplifications introduced by Hooghoudt's approach in Equation 17 through Equation 18, descend directly from the Dupuit- Forchheimer (Ritzema, 1994) assumptions.

Figure 31 shows the main components of *Channel Water Balance* just described.

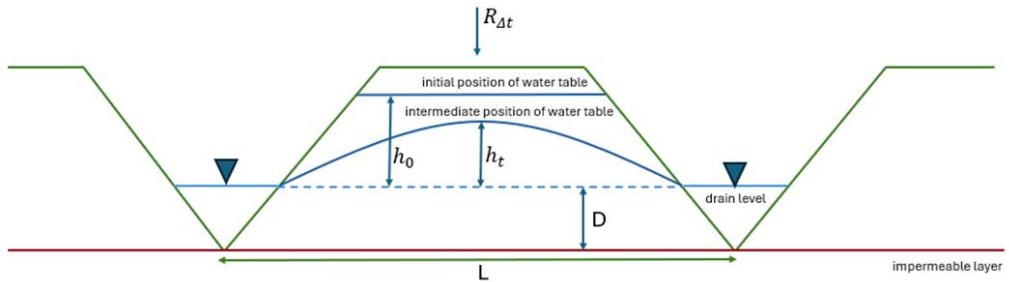


Fig. 31 – Schematic representation of the drainage system assumed to model groundwater drainage in the WetMAT model

The determination of the updated water depth in the channels at the end of each time step, useful as input for the next time step, is computed using a generalised reservoir method (Beven, 2001) as follows in Equation 19:

$$\frac{Q}{Q_{max}} = \left(\frac{V}{V_{max}} \right)^\alpha = \left(\frac{D^2 \cdot L_c}{D_{max}^2 \cdot L_c} \right)^\alpha \quad (19)$$

where Q [$m^3 d^{-1}$] is the drained discharge, Q_{max} [$m^3 day^{-1}$] is the outflow rate under full channel conditions, L_c [m] is the length of the entire drainage network and α [-] is a shape parameter, generally set at 1,5 in the case of open nets. Equation 19 therefore becomes in this case:

$$D = D_{max} \left[\left(\frac{Q}{Q_{max}} \right)^{\frac{1}{1.5}} \right]^{\frac{1}{2}} \quad (20)$$

with D_{max} [m] is measured from the case study and Q_{max} [$m^3 day^{-1}$] is dimensioned using the Chezy approach considering the characteristics of a triangular channel with 45° bank slopes (Chow, 1959) as follows in Equation 21.

$$Q_{max} = A_{max} \cdot C \cdot \sqrt{RJ} = D_{max}^2 \cdot \frac{1}{n} \cdot R^{\frac{2}{3}} \cdot J^{\frac{1}{2}} \quad (21)$$

where A_{max} [m^2] is the wetted area obtained from D_{max} (i.e. D_{max}^2), R [m] represents the maximum hydraulic radius and J [-] is the channel slope. C [$m^{\frac{1}{2}} s^{-1}$], the Chezy coefficient, is expressed as:

$$C = \frac{1}{n} R^{\frac{1}{6}} \quad (22)$$

where n [$s \cdot m^{-\frac{1}{3}}$] is the Manning's coefficient that varies with the roughness of the channel.

The third and final balance proposed by WetMAT is in this case the Pond Water Balance implemented starting from the output flows of the Channel Water Balance. The hydrological balance of the pond is expressed using Equation 23

$$V_t = V_{t-1} + Q_{in(t)} + P - E - Q_{out(t-1)} \quad (23)$$

where $Q_{in(t)}$ [m^3] is determined daily from Equation 18, in relation to the basin that drains its water into the pond, P [m^3] is the direct precipitation on the pond, E is the direct evaporation from the pond and $Q_{out(t-1)}$ [m^3] represents the daily outflow from the pond directly to the sea. This term has been modelled according to Darcy's approach (Darcy 1856), adapting it to the current case study as expressed in Equation 24.

$$Q_{out} = \frac{K \cdot B \cdot l^2}{L_s} \quad (24)$$

In Equation 24 K [$m \text{ day}^{-1}$] is the soil hydraulic conductivity, B [m] is the pond section involved in the process, L_s [m] represents the distance between the pond and the sea outlet, l [m a.s.l] represents the pond level. Equation 24 was used by varying the value of K as a function of l : analysing observed data, it was found that at 0.56 [m asl], the permeability of the pond system changes, decreasing dramatically, encountering a soil layer with a different composition and therefore the aforementioned value of l was used as a threshold for the change of permeability.

The dependence of Q_{out} on the pond level could thus be observed, which is also one of the output variables later considered for calibration. For l determination, given the widespread difficulty in making level and volume measurements in natural reservoirs such as small lakes or wetland ponds and thus in determining their Stage-Storage Curve (Messenger et al., 2016; Hollister et al., 2011), the approach of Hayashi and van der Kamp (2000) was used. This approach is useful for determining the pond area and volume through a few area and level measurements, and in this study, using an inverse approach starting from original volume calculation, the value of l for each time step could be obtained according to Equation 25:

$$l = \left[\frac{V \left(1 + \frac{2}{p}\right)}{s} \right]^{\left(\frac{1}{1 + \frac{2}{p}}\right)} \quad (25)$$

where $V [m^3]$ is the pond volume calculated as in Equation 23, p and s are two parameters relating respectively to the depression profile of the reservoir and the size of the wetland (Minke et al., 2010). The daily area of the wetland has been also calculated according to Equation 26, to obtain an accurate daily assessment of direct precipitation and direct evaporation in Equation 23.

$$A = s \cdot l^{\frac{2}{p}} \quad (26)$$

The equations presented so far describe wetland processes focusing on water quantity. However, the specific issues observed at Torre Guaceto (e.g., aquifer salinization and, consequently, pond salinization) require a modelling approach capable of describing also water quality dynamics. Variations in salinity represent an important indicator for ecological assessment especially in coastal wetlands (Amores et al., 2013) and are equally relevant for wetland management and restoration purposes. For this reason, while maintaining its characteristic simplicity in both computational structure and data requirements, the WetMAT model has been extended to simulate salinity dynamics within the pond.

The salt balance is also implemented in WetMAT on a daily scale and is expressed according to Equation 27:

$$S_t = \frac{S_{t-1} V_{t-1} + Q_{in} S_{Qin} - Q_{out}(t-1) S_{t-1}}{V} \quad (27)$$

where $S [g/L]$ is the pond salinity, $V [m^3]$ is the pond volume, $Q_{in} [m^3]$ is the daily discharge from channels to pond, $Q_{out} [m^3]$ is the daily discharge from pond to sea, as calculated in Equation 24 and $S_{Qin} [g/L]$ is the salinity of the inflow into the pond, calculated in Equation 28.

$$S_{Qin} = \frac{G_d \cdot S_{GW} + (SWC - \vartheta) \cdot S_P}{(SWC - \vartheta) + G_d - G_c} \quad (28)$$

$S_{GW} [g\ l^{-1}]$ is the initial groundwater salinity, taken with a constant calibrated value for model simplicity given the uncertainty in monitoring the dynamics that generate salinity variations in groundwater (Blanco-Coronas et al., 2022; El Azhari et al., 2025); the term “ $SWC - \vartheta$ ” [mm] represents the precipitation recharge rate that does not contribute to the *Soil Water Balance* and is also saline ($S_P [g\ l^{-1}]$ represent its salinity) as it flows in soils salinized by groundwater irrigation (Middleton and Boudell, 2023). $S_{Qin} [g\ l^{-1}]$ as can be deduced from Equation 26, varies its value daily, particularly with a decrease during wet and winter periods and an increase during the summer season characterised by strong evapotranspiration, when part of the flow from groundwater feeds the *Soil Water Balance* and salt accumulate in the remaining part of the flow directed to the pond.

The parameters adopted for the implementation of WetMAT in the Torre Guaceto case study are summarised in Table 5: “S” denotes parameters that are directly involved in *Soil Water Balance*, “C” parameters related to *Channel Water Balance* and “P” parameters involved in *Pond Water Balance*. In “Typology” section there are indications about guidance on how to choose values for each individual parameter. *C* parameters underwent a calibration process, *M* parameters were measured, and *D* parameters were defined on the basis of literature.

Table 5 – WetMAT parameters in Torre Guaceto case study

PARAMETER	DESCRIPTION	UNIT OF MEASURE	BALANCE	TPOLOGY
K	Soil conductivity	$m\ day^{-1}$	S-C	C
μ	Soil porosity	-	S	C
L	Channel distance	m	C	M
n	Manning's coefficient	-	C	D
S	Channel slope	-	C	M
D_{max}	Max channel depth	m	C	M
ϑ_{WP}	Wilting point's soil content	$mm\ m^{-1}$	S	C
ϑ_{FC}	Field capacity's soil content	$mm\ m^{-1}$	S	C
ρ	Soil water depletion fraction	-	S	D
A	Catchment area	m^2	C	M
ρ	Depression profile parameter	-	P	M
s	Pond dimension parameter	-	P	M
L_s	Pond distance from sea outlet	m	P	M
-	K threshold	m	P	M
B	Pond section	m	P	M
S_{GW}	Groundwater salinity	$g\ L^{-1}$	P	C
S_P	Precipitation salinity	$g\ L^{-1}$	P	C

For n coefficient, Manning's roughness coefficient, a value of 0.06 was chosen, which is generally used for modelling strongly vegetated channels (The Engineering ToolBox, 2004), while for the soil water depletion fraction for no stress, a value of 0.64 has been adopted, typical of maize cultivation under the specific conditions of evapotranspiration. This choice was made not to represent local cropping patterns, but to provide a well-established proxy for herbaceous vegetation growing in a coastal wetland under Mediterranean climatic conditions (Allen et al., 1998).

With regard to the soil water content thresholds or hydraulic conductivity values, although the values were used for calibration, literature was taken into account to ensure that they fall within a suitable range for the specific soil type. It is worth noting that the implementation of each module, whether related to the soil, channels, or pond, requires a comparable number of parameters. No disproportion is observed among the different components of the model, which suggests that no single module has been developed with significantly greater detail than the others. Compared to the previous application of WetMAT in Doñana case study, more parameters are used, which are necessary to model small-scale spatial dynamics in greater detail. Nevertheless, the model preserves its parsimonious structure and continues to rely on a limited set of input data operate.

3.3 *WetMAT calibration and validation*

In hydrological modelling of small basins, it is crucial to have as much observed data as possible. Due to their small size, the processes involved are highly detailed and require sufficient calibration to ensure an accurate representation (Latron and Lana-Renault, 2018).

Given the increasing scientific interest for the study area, the site has been recently gauged with multiple monitoring instruments to measure key environmental variables such as air and water temperature, pressure, and salinity. Specifically, Figure 32 illustrates the layout of the groundwater monitoring network in Torre Guaceto, consisting of several observation wells and stations, including a monitoring station located within the pond itself. Data from this station, marked as “POND” were used as a reference for the model calibration and validation processes.

Figure 32 highlights not only all the monitoring stations but also those equipped with continuous data acquisition systems, including the pond station. These stations define an ideal hydrogeological transect, represented by a dotted line, along which salinity variations and aquifer characteristics are being investigated within broader environmental research. The gradient intercepted by the transect spans from areas influenced by agricultural activities in the inland sector to increasingly natural conditions

approaching the coastline. The “BARO” station refers to barometric pressure measurements, which differ from the others as they are not conducted on water surfaces.

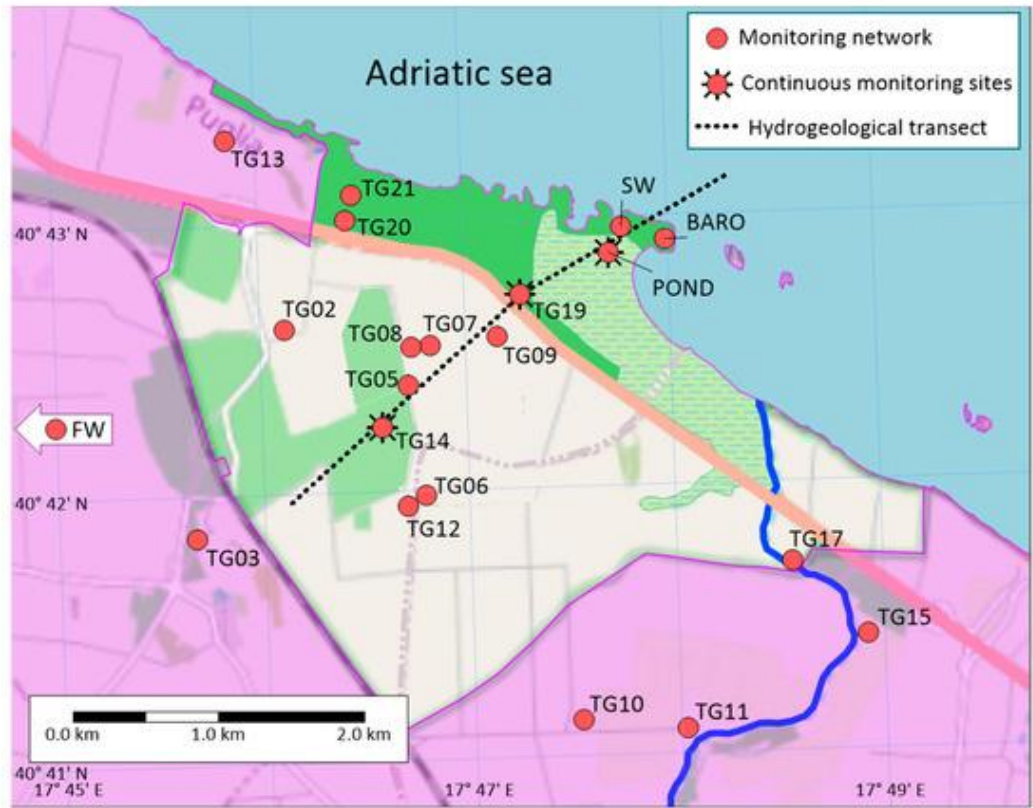


Fig. 32 –Map of the Torre Guaceto groundwater monitoring network

As the monitoring network has been installed only in November 2022, observed data are still limited for proper calibration and validation. The pond level [m a.s.l.] was therefore used to calibrate WetMAT using the entire available data set, up to July 2024. Subsequently, measured salinity data [g/L], obtained from electrical conductivity [mS/cm at 25°] from the pond gauge within the same time window, were used for model validation: the change of variable used for the calibration and validation processes is due to the lack of long historical series. A constant and calibrated daily

component G_d is here assumed. Figure 33 shows the results of the WetMAT calibration and validation process.



Fig. 33 – Panel (a) shows the WetMAT model calibration according to the “pond level” variable. Panel (b) shows the WetMAT validation process results according to the “pond salinity” variable

As Figure 33 shows, both the calibration and validation process led to satisfactory results. Using the Kling Gupta Efficiency (KGE) metric (Gupta et al., 2009), a result of 0.87 was obtained for calibration and 0.802 for validation, which suggest a significant model reliability (Khatami et al., 2020).

Concerning the pond level, both the model and the observation show an unstable behaviour during the summer period when the pond level is lower. This behaviour justifies changing the K parameter below a given threshold. The observed behaviour of the pond level between January and February, with an abrupt rise followed by a sharp fall that the model cannot reproduce perfectly, is due to the effect of a meteorological tide, an exceptional event that also affects the pond due to its connection with the groundwater and that does not occur every year. In any case, such an event produces a local disturbance only, without a significant impact on the rest of the analysis.

As far as salinity is concerned, no unstable zones are observed even in summer periods. WetMAT tends to reproduce quite well observed data, although salinity peaks are higher in the model compared to observed data.

3.4 *WetMAT outputs from Torre Guaceto case study*

As discussed in Section 3.3, two of WetMAT's most relevant outputs are pond water level and salinity. In addition to their relevance for calibration and validation, given the availability of historical datasets for comparison, these outputs provide both a robust representation of water quantity (through water level) and quality (through salinity) that can be used also to support ecological analyses e.g. in the scenario analysis of Section 3.5. At the same time, it should be noted that the WetMAT model is also capable of simulating additional outputs, which are described below, and which may be useful for other analyses not discussed in this work or for the implementation of indicators, so as to enable comparison between different case studies.

As far as the soil balance is concerned, its actual capacity (θ) can be estimated, with a daily temporal step, as shown in Figure 34.

Wet and dry periods, such as the summer dry season, can be identified as well as winter periods with differences in soil moisture.

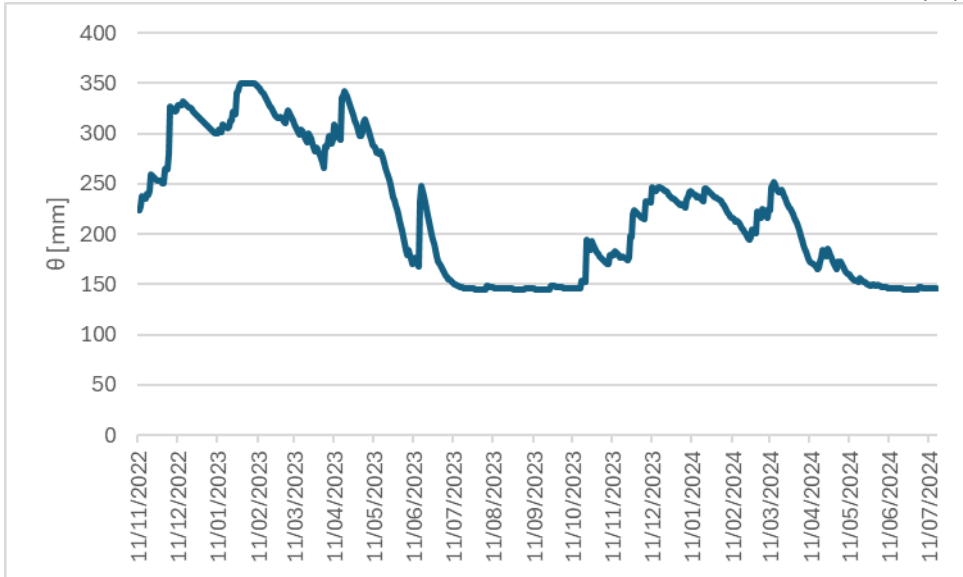


Fig. 34 – Daily soil moisture plot, main output of the Soil Water Balance

A representative output of the Channel Water Balance is the flow rate Q_{in} [$m^3 day^{-1}$], which represents the daily volume leaving the drainage network and reaching its final destination, either the sea or the pond, as shown in Figure 35.

The plot illustrates that during periods of soil saturation, the surplus water exceeding soil storage capacity adds to the groundwater contribution drained into the channels. Conversely, during dry periods, when evapotranspiration exerts a strong influence on soil moisture, the final destinations receive reduced inflows, as a greater portion of water is consumed within the Soil Water Balance.

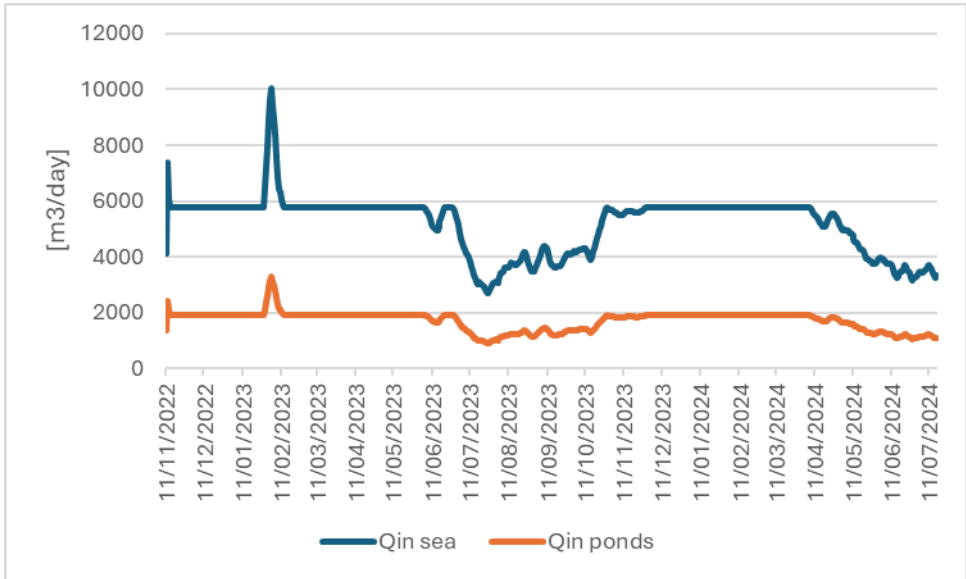


Fig. 35 – Daily plot of Q_{in} , flow rate drained from the canals to the pond or to the sea and representative output of the Channel Water Balance

With regard to the Pond Water Balance, the most frequently used outputs, pond level and salinity, have already been discussed earlier in this work. Additional outputs, namely the pond area and volume, are presented in Figure 36. The area is a variable that is difficult to measure directly, as the wetland is largely covered by reeds and other emergent vegetation: for this reason, although the model provides an estimate of the flooded area, establishing a correlation with observed data proves challenging. The presence of dense vegetation results in remote sensing consistently detecting the same unvegetated water surface, regardless of seasonal changes. However, field observations confirm that during wetter periods, flooding extends beyond these areas into vegetated zones, which are not adequately captured by standard surface detection methods. The volume plot reflects the hydrological balance described by Equation 23, showing a strong correspondence between pond volume and surface area.

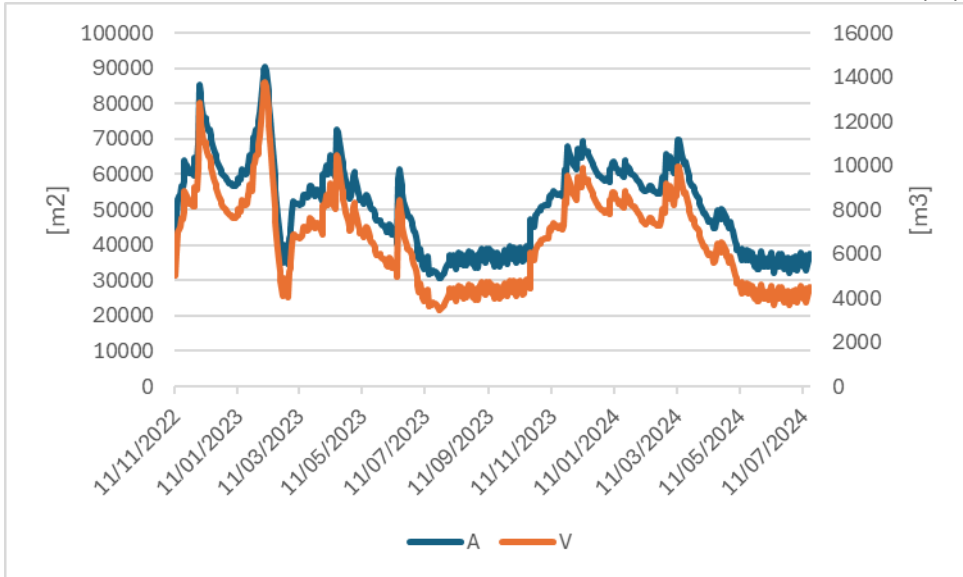


Fig. 36 – Daily plot of pond area and volume representative outputs of the Pond Water Balance

3.5 Groundwater management scenarios for salinity control

In the Torre Guaceto case study, the WetMAT model was applied to develop sustainable groundwater management scenarios aimed at reducing salinity levels, particularly during the summer months. To this end, different management strategies were simulated by modifying groundwater abstraction for agricultural use, as described in the following section.

Figure 37 shows three different scenarios, under a constant G_d , in which from October to April of each hydrological year a daily flow rate of treated wastewater is discharged in the system. This period is crucial because outside the irrigation season of the crops contiguous to the study area, can be therefore useful to use the water for environmental purposes.

All proposed scenarios are compared with the current scenario, validated in Section 3.2. The first scenario analysed involves the discharge of a daily portion of treated wastewater equal to 496 m^3 directly into the pond. This quantity corresponds to the assumption of $2000 \text{ m}^3 \text{ day}^{-1}$ of wastewater discharging into the study area and di-

vided proportionally to pond area. As shown in Figure 37 this solution can contribute to reduce salinity during the period when the spills occur compared to the baseline scenario, but this does not lead to a lowering of salinity during the summer months. The second scenario is focused on the same amount of daily discharge of treated wastewater that is, however, discharged into the canal system. In this way, the effects of a possible temporal lag on salinity during the summer season need to be studied. The ‘channel discharge’ scenario, however, does not produce any significant improvement compared to the previous scenario. In the third scenario, three quarters of the total daily flow available from the wastewater treatment plant (1500 m^3) are diverted into the pond. During the periods when the flow is diverted (Oct-Apr), there is a significant decrease in salinity compared to the “baseline” scenario. However, in the other period (May-September), salinity conditions can be even worsened. This phenomenon may be attributed to the temporary dilution of the system induced by freshwater inputs, which is insufficient to establish a new hydrological equilibrium. During summer months, intensified evaporation significantly reduces the overall pond volume, thereby concentrating residual salts. As a result, salinity levels may exceed those of the baseline scenario, which does not experience the same sequence of dilution followed by evaporation. This dynamic is consistent with patterns observed in closed systems where salinity is strongly influenced by evapotranspiration-driven mass balance processes (Nourein Mohammed and Tarboton, 2012).

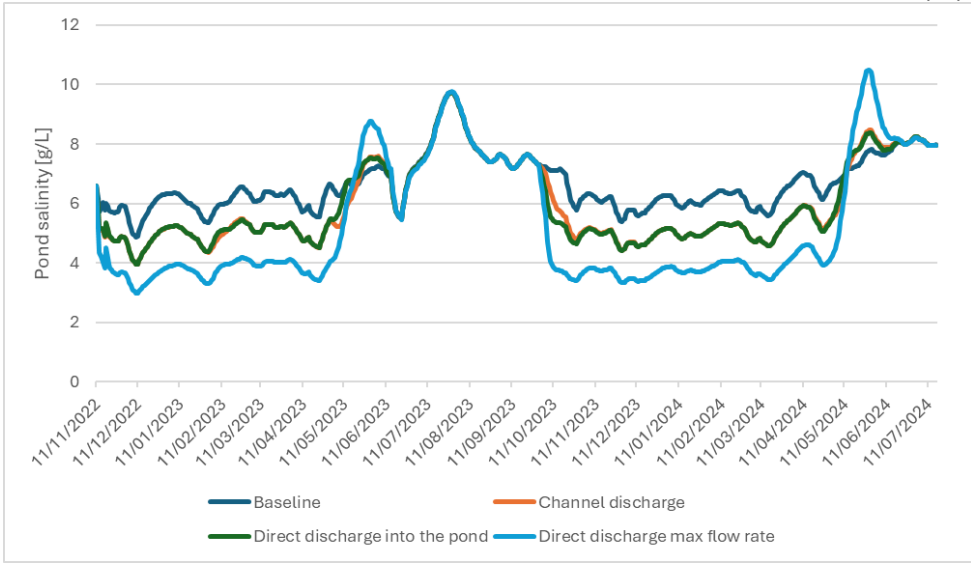


Fig. 37 –Comparison of scenarios for lowering salinity in the pond under constant daily G_d conditions

Scenarios in Figure 37 show that the reduction in salinity following the use of treated wastewater in the system does not occur with significant time lags; it is immediate, even though drainage channels. However, this does not solve the problem of high salinity in summer months, which would instead require a direct discharge of treated water. This solution is not feasible in practice, since it would be impossible to store such volumes of water. A new approach is therefore being proposed, with the aim of reusing wastewater not directly in the wetland system but for supporting the irrigation of adjacent fields (Obijanya et al., 2025). Besides the beneficial effects on agriculture (Trotta et al., 2024), it could also support a better groundwater management (Benaafi et al., 2024) which can, ultimately, help limiting the salinisation of the pond. Data from another hydrological model, developed within the framework of the INTERACTION project and described in one of the project's technical deliverables, are used to run this scenario, as the upstream area of the basin, which include irrigated areas, need to be included. Through a simple assessment of the crops present in the area and their current irrigation status, this hydrological model is able to provide a

G_d

value as input for WetMAT model that varies on a monthly basis, depending on climate conditions and on the amount of withdrawals upstream. The variability of climate is reflected in groundwater recharge data, and modelling water withdrawals in the upstream basin allows scenarios for managing these withdrawals to be formulated. The hydrological model used for this purpose is based on the water balance of a model developed for the hydrogeological balance of agricultural landscapes (Portoghese et al., 2005). Figure 38 shows the G_d values [mm] used in WetMAT: higher values of the variable are observed in the winter months. A decrease characterizes spring as the irrigation season intensifies, with a minimum value in June, as evapotranspiration and irrigation processes occur simultaneously.

Figure 39 shows model calibration and validation processes, based on the variables ‘Pond Level’ and ‘Pond Salinity’. In particular, Fig. 39 a) shows the calibration results with a KGE value of 0.71, which is acceptable (Choi et al., 2021) given the higher degree of uncertainty related to the introduction of G_d and the monthly scale of the hydrological model used.

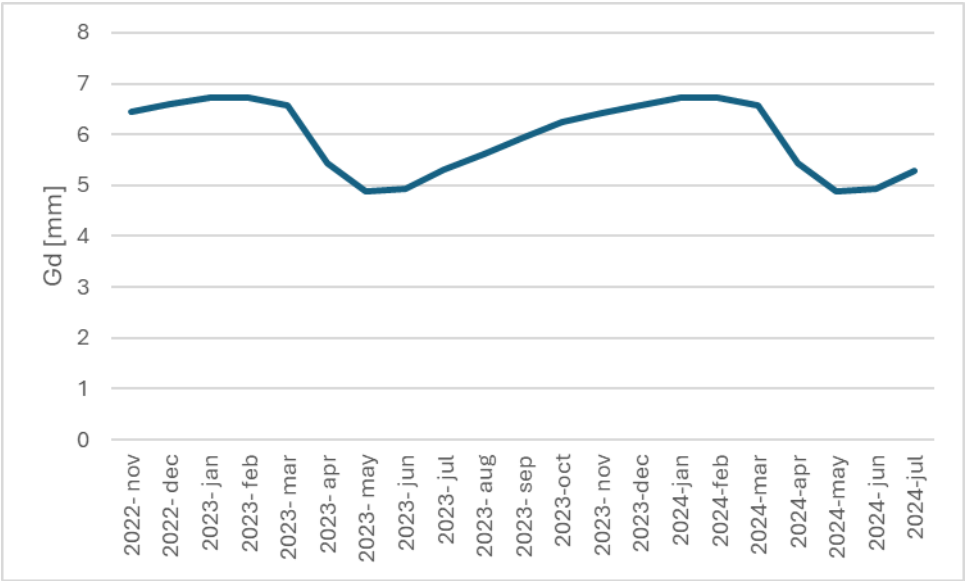


Fig. 38 – Monthly variation in G_d according to irrigation and climatic variations in the wetland upstream basin

Figure 39 b) shows the WetMAT validation process along with the comparison between ‘observed’ and ‘modelled baseline’, which is satisfactory ($KGE = 0.71$). Two scenarios are analysed, in which 20% and 40% of the water used for irrigation and taken from groundwater is replaced with treated wastewater. The daily G_d increases in these scenarios compared to the baseline in response to lower pumping levels. This does not show any significant effect compared to the baseline scenario, apart from the salinity peaks recorded in the summer months, which are reduced up to 18.5%.

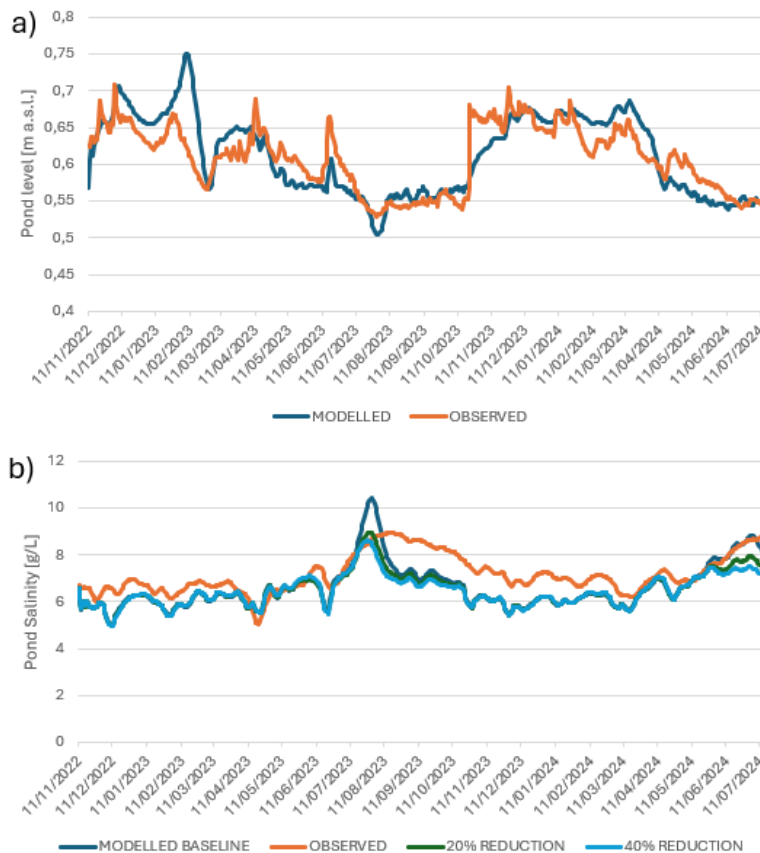


Fig. 39– Panel (a) shows the WetMAT model calibration with daily variable Gd according to the “pond level” variable. Panel (b) shows the WetMAT validation according to the “pond salinity” variable, and management scenarios for salinity decrease

4 DISCUSSION

This chapter provides a critical discussion of the results, examining how far the research has succeeded in addressing the initial questions and objectives. It summarises the main findings, explores their broader significance, and analyses the main methodological challenges and limitations encountered throughout the study. It is worth recalling that the research questions guiding this work focused, first, on the possibility of characterising and replicating the hydrological dynamics and regimes of wetlands, particularly given the complexity typical of Mediterranean climate conditions through a simple hydrological model. The second question concerned the potential for a hydrological model to provide outputs suitable for scenario analysis and to support decision-making aimed at wetland restoration and conservation. The third question addressed the model's scalability and its ability to adapt to diverse hydrological and geographical contexts.

The results presented in Chapters 2 and 3 indicate that the work answers the first research question, because a parsimonious hydrological model named WetMAT was developed which enabled to reproduce the dynamics and regimes of Mediterranean wetlands. For Doñana (Figure 19) and Torre Guaceto (Figure 33) marshland basins, the satisfactory correspondence between simulated and observed variables shows that WetMAT can capture both intra-annual and inter-annual variability in wetland regimes, a critical requirement in Mediterranean-climate systems. This predicting capability is particularly salient for Doñana case study for which inter- and intra-annual fluctuations in flooding conditions crucial for the ecological functioning of the marshland.

As shown by the calibration and validation outputs for the two case studies, and by the WetMAT results detailed in Sections 2.5 and 3.4, the same model structure can be applied to different sites characterized by different dominant processes and conservation targets. Beyond water level, which is assessed in both sites for the Doñana marshland and for the Torre Guaceto pond, we focused on different hydro-ecological response variables. In the Doñana study, the flooded area is prioritised because it var-

ies markedly both within and across years and is therefore synthesized by the annual maximum flooded area and the hydroperiod, with the proposed Inundation Persistence Index. Flooded area, which can be consistently derived from satellite imagery (Díaz-Delgado et al., 2016), is also a widely adopted metric in studies of temporary wetlands (Tsyganskaya et al., 2018), as it enables the derivation of hydroperiod, another key descriptor for ecological and hydrological analyses (Gómez-Rodríguez et al., 2010).

In the case of Torre Guaceto, although WetMAT produces daily estimates of inundated area, this variable is not used as a primary evaluation metric. The restricted spatial extent and dense emergent vegetation hinder a reliable delineation of flooded surfaces from remote sensing (canopy occlusion, mixed pixels), and a small core area remains perennially inundated, which reduces the diagnostic value of “maximum flooded area.” In any case, this does not represent a strong limiting issue, as Torre Guaceto is a permanent wetland area. As discussed in Chapter 3, the key conservation issue is therefore more qualitative than quantitative, pertaining mainly to the dynamics and drivers of salinity rather than to short-term changes in areal extent. Quantitative observations of pond salinity remain essential to constrain the salinity balance, and the corresponding model outputs.

With reference to the second research question, we demonstrate, across both case studies, how a single model structure can underpin decision-support aimed at addressing wetland conservation strategies. For Doñana case study, the implementation of an Inundation Persistence Index (IPI) enables, for each hydrological year, an estimate of the additional water required, beyond direct precipitation, to sustain a condition of average marshland inundation. To these objectives the chart presented in Figure 24 is relevant as it offers a simple yet dynamic means to diagnose the wetland’s water requirements, based on the model capability to reproduce the daily inflows reaching the system and the corresponding flooded extent.

In the Torre Guaceto case study, although relying on different output variables as discussed above, it was possible to address the second research question successfully. Through the application described in Section 3.5, the WetMAT model allowed evaluat-

ing water-resource management scenarios aimed at reducing pond salinity, particularly during summer peaks. Acting on agricultural water management to improve nature conservation suggests two considerations. First, it has been shown that tailored hydrological modelling can be crucial when discussing water-management strategies that exert a relevant though indirect impact on coastal freshwater ecosystems. Second, this model application is a genuine example of how the Water–Ecosystem–Food Nexus is a valuable approach for managing human–environment interactions.

In the case of Torre Guaceto, applications directly useful for current water management were presented, demonstrating the viability of WEFE modelling under future projections to analyse scenarios, thereby supporting the integration of scientific results into decision-making.

Overall, the applications in Chapters 2 and 3 attest to the model's effective practical applicability, while also detailing its implementation (sensitivity analysis, calibration, and validation) and summarising the derived outputs.

With regard to the third research question, the evidence presented supports an affirmative answer. Although WetMAT has not yet been tested across a large number of case studies, it has been applied in two contexts that, beyond sharing a Mediterranean climatic setting and similar conflicts between anthropogenic water use and nature conservation, lie at opposite ends of the spectrum in terms of hydrological characteristics and spatial scale, suggesting that the two case studies together offer an encouraging starting point for assessing the model's applicability to other wetland systems. The contrasts between the two case studies indeed are substantial: the model was performed in Doñana marshland and in a system approximately 2 orders of magnitude smaller. Moreover, while Doñana site was treated as hydrologically isolated, with minimal exchanges with groundwater and channelled rivers, the Torre Guaceto application has explicitly represented groundwater drainage to channels and, more generally, the dynamics of a groundwater-fed wetland. This demonstrates that the proposed model structure can be adapted to systems of differing complexity while retaining credible performance.

Attention is drawn to the process equations required in the two applications. In Torre Guaceto, the extremely small spatial scale necessitated more detailed process modelling. This is informative for replicability under varying conditions: despite differences in equations and outputs, the model retained its core architecture, i.e. three interconnected balance equations. Indeed, the shared WetMAT foundation with increasing complexity depending on model's stated objectives also allowed the hydrological understanding of investigated system (Giordani et al., 2008), culminating in a semi-distributed parameterisation.

One of the challenges of this work is the limited data availability for model implementation, calibration and validation phases. For some parameters, such as those related to vegetation root depth, it was necessary to rely on values reported in the literature. Although the sensitivity analysis indicates that this choice did not significantly influence the results, it nonetheless highlights a common limitation in the modelling of natural environments, where site-specific measurements are often unavailable. For Doñana case study, part of the evidence was based necessarily on grey literature, i.e., non-peer-reviewed or locally produced sources that reported flooding events and offered site-specific expert information. These materials were used to complement and cross-check instrumental records and satellite products, thereby expanding the empirical basis.

As reported in Green et al., 2024 the streamflow gauges of surface watercourses in the area do not have complete time series. Similarly, although gates exist for the surface drainage of the marshland, they are not gauged and therefore it is impossible to derive information related to its discharges. As for the ecotones and the levels of groundwater, there are reports on their dramatic drops and decreases but there reliable information about their possible contributions to the wetland is missing. However, the rationale behind WetMAT is to keep the model as simple as possible, although this may introduce simplifications such as the use of precipitation and temperature as the only input for the water balance model.

Another limitation relates to the calibration and validation phases due to the limited length of the available historical time series. Furthermore, the available data on the

maximum MFA, derived from the literature, lacks indications on when it occurs during the year.

These limitations also affect the Torre Guaceto case study. Given the short historical record, calibration and validation were conducted against two distinct variables. Modelling of surface watercourses was not required; this does not constitute a substantive limitation, as an implementation would largely mirror the approach adopted for groundwater drainage.

A common limitation of both case studies concerns the selection of soil-related model parameters. Whereas geometric characteristics of channels and of the wetland as a whole are relatively straightforward to measure, soil parameters require calibration and, in any case, reflect average values. This constitutes a limitation, particularly in large study areas.

The main challenge in Torre Guaceto modelling, which increases the value of the work as it really adds a step to what has been done on Doñana, concerned the representation of groundwater drainage. In the absence of detailed documentation on the existing drainage system's design, construction criteria were hypothesised to characterise its functioning. Combined with direct measurements of channel number, depth, and length, this information enabled an inverse-modelling approach to infer daily groundwater-derived flows conveyed by the channels.

In addition to quantifying wetland water requirements for decision-making under present climate conditions, a similar workflow can be adopted to address climate-change effects on wetland response by incorporating climatic uncertainty alongside dynamic anthropogenic pressures for long-term scenario analysis.

CONCLUSIONS

This PhD thesis presents the implementation of a parsimonious hydrological model based on a lumped parameterization structure, which allowed reproducing the hydrological regimes of Mediterranean wetlands across a range of spatial scales and evaluating the effectiveness of relevant protection measures for conservation and restoration purposes. The model was applied to coastal wetlands that differ substantially in their characteristics yet share contrasts between human development and nature conservation. The principal insight is that representing flooding or salinity dynamics in wetlands does necessarily require highly complex models which are usually data intensive and case-specific, thereby undermining scalability and broader applicability. By contrast, what is essential is a detailed understanding of the wetland site's constitutive components and dynamics (geological setting, hydrological functioning, and major anthropogenic alterations) because these factors determine historical evolution and current conditions and inform about which model outputs should be prioritised to support decision-making for a specific wetland protection.

Future developments of this work concern several fronts. First, applying the WetMAT model to additional case studies, both within and beyond the Mediterranean region, would provide further confirmation of replicability across climate zones. The two case studies in this work demonstrate that the model is capable of operating across a wide range of spatial scales, from very large to very small systems, while also reproducing markedly different hydrological conditions. Future works will focus on testing the model under intermediate spatial scales and in contexts characterised by hydrological or management challenges different from those addressed in this study. Applications to other generic case studies could also identify and evaluate additional wetland protection measures aiming at quantitative and qualitative improvements. A second line of research involves the systematic integration of WetMAT outputs with other types of modelling, both ecological and decision oriented. In particular, the inclusion of hydrological modelling for estimating environmental water requirements within decision-making processes will be pursued through modelling ap-

proaches such as participatory modelling of socio-ecological systems. As a third line, the use of a simple hydrological model like WetMAT combined with remote sensing monitoring is a promising data-model fusion for effectively mitigating the lack of regular hydrological observations, described above and common in wetland basins, thereby constituting a useful management tool for addressing water management issues regarding wetland protection and conservation.

NOTES

1. The research activities described in the present work relating to the Doñana case study have been performed within the LENSES Project (PRIMA programme – Grant Agreement n. 2041, <https://www.lensesprima.eu/>)
2. The research activities described in the present work relating to the Torre Guaceto study have been performed within the framework of INTERACTION project, funded by the Belmont Forum under the Collaborative Research Action (CRA) call on the theme “Towards Sustainability of Soils and Groundwater for Society.” The project received support through a grant from the Belmont Forum, administered by the Earth System Science and Environmental Technologies Department of the National Research Council of Italy (Belmont Forum–CNR: B55F21001250007)

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CURRICULUM



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Collaboration with the Water Research Institute, National Research Council (IRSA-CNR, Bari, Italy) : 16/10/2023 – 31/10/2025

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- LENSES (Learning and action alliances for Nexus Environments in an uncertain future) – PRIMA (EU)
- NEXUSLABS (Nexus Living Labs for Multi-Level Innovation in Mediterranean Irrigated Agriculture) – PRIMA (EU)
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Politecnico di Bari jointly with the Water Research Institute, National Research Council (IRSA-CNR, Bari, Italy)

01/11/2022 – ongoing

Supervisors: Prof. Vito Iacobellis (Politecnico di Bari), Eng. Ivan Portoghese, PhD (IRSA- CNR), Prof. Alessandro Pagano (Politecnico di Bari)

Thesis title: “Development of a simplified hydrological model to investigate wetland regimes for conservation and restoration purposes”

Politecnico di Bari, Master’s Degree in Civil Engineering (Curriculum Hydraulic Engineering)

09/2019 – 12/2021

110/110 cum laude

Thesis in Catchment Hydrology, entitled “ Study of the hydrological balance extended to the Canale Reale basin”, Prof. Vito Iacobellis, Eng. Ivan Portoghese, Prof. Vincenzo Totaro.

Politecnico di Bari, Bachelor’s Degree in Civil and Environmental Engineering

Thesis in Materials Science and Technology, entitled “ Cementitious composites based on polymer waste materials: the case of PET bottles”, Prof. Andrea Petrella, Prof. Dora Foti.

AWARDS

Best Presentation Award

PhD Days and Marchi Lecture, 11-12/09/2025

'Estimation of wetland water balance using the WetMAT lumped model in different hydrological contexts'

PUBLICATIONS

Paper/Abstract for conference proceedings

- **Panciera, C.**, Iacobellis, V., Pagano, A., Masciale, R., Passarella, G., Portoghese, I. (2025). Un modello semplificato per lo studio del bilancio idrologico della zona umida costiera di Torre Guaceto (BR). Proceedings of "Le Giornate dell'Idrologia 2025", Bari, Italia.
- **Panciera, C.**, Portoghese, I., Pagano, A., Giordano, R. (2025). Estimation of wetland water balance using the WetMAT lumped model in different hydrological contexts. In New challenges in understanding and managing water-related risks in a changing environment, Proceedings of the 13th World Congress of EWRA on Water Resources and Environment, Palermo, Italy. ISBN: 978-618-84419-2-7.
- Gunacti, M.C., Cetinkaya, C.P., Onsluel Gul, Barbaros, F., Gul, A., **Panciera, C.**, et al., (2024). Evaluating the Impact of Nature Based Solutions on Water Resources Management. Proceedings of International Union of Soil Sciences (IUSS) Congress, Florence, Italy.
- **Panciera, C.**, Portoghese, I., Pagano, A., Giordano, R., Yesilyurt, S.N., Gunacti, M.C., Cetinkaya, C.P., Onusluel Gul, G., Najar, M., Lopez, E., Bea, M., (2024). A methodological approach for supporting socio-hydrological modelling of a complex Nexus system. In Water-Across boundaries, Full Paper in Proceedings of 8th IAHR Europe Congress, Lisbon, Portugal. DOI: 10.34638/jv1h-bs60.
- **Panciera, C.**, Pagano, A., Giordano, R., Portoghese, I., Vergine, P., Bea, M., Najar, M., Gunacti, M.C., Onusluel Gul, G. (2023). Unraveling the complexity of water-ecosystems-food nexus through integrated modelling: the Doñana case study in SW Spain. In Managing Water-Energy-Land-Food under Climatic, Environmental and Social Instability. Proceedings of the 12th World Congress of EWRA on Water Resources and Environment, Thessaloniki, Greece. ISBN: 978-618-84419-1-0.
- Portoghese, I., Totaro, V., **Panciera, C.**, Iacobellis, V. (2022). Calibration of a large-scale water balance model in a semi-arid ungauged catchment. In Proceedings of the 39th IAHR World Con-

gress, Granada, Spain. DOI: doi://10.3850/IAHR-39WC2521716X20221812

Paper for journal

- **Panciera, C.**, Pagano, A., Iacobellis, V., Bea Martínez, M., and Portoghese, I.: A simplified model to investigate the hydrological regimes of temporary wetlands: the case study of Doñana marshland (Spain), EGU sphere [preprint], <https://doi.org/10.5194/egusphere-2025-2931>, 2025.

Other research outputs

- Santoro, S., **Panciera, C.**, Portoghese, I., Giordano, R. (2024) Catalogue of the mitigation solutions available in the 4 different topographical areas (Deliverable D.1.4.1, LocAll4Flood Project).
- Pagano, A., Giordano, R., Portoghese, I., Berardi, M., Vergine, P., Santoro, S., Selicato, L., **Panciera, C.**, Yaseen, M. (2024) Evidence from scenario analysis and policy recommendations (Deliverable D4.4, WP 4, LENSES Project).
- Gunacti, M.C., Cetinkaya, C.P., Onusluel Gul, G., Gul, A., Barbaros, F., Yesilyurt, S.N., Najar, M., **Panciera, C.**, Pagano, A., Portoghese, I., et al. (2023) Flexible approaches to managing competing demands for water (Deliverable D7.4, WP 7, LENSES Project).

FURTHER INFORMATIONS

- Winner of Call for Applications No. 1 DD No. 24 of 15/05/2023, 'Call for Applications for the assignment of positions related to supplementary teaching activities for the academic years 2022/2023 - 2023/2024' for the course 'Water Resource Assessment', 6 CFU, 40 hours, for the Master's Degree Course in Civil Infrastructure Management Engineering.
- Member of the Organising Committee for the Conference "Le Giornate dell' Idrologia - Addressing Climate and Territorial Challenges: Scenarios, Risks and Adaptation Strategies" Polytechnic University of Bari, 8-10 September 2025.