Groundwater dependent ecosystems in coastal Mediterranean regions: Characterization, challenges and management for their protection

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Abstract

Coastal lagoons deliver a wide range of valuable ecosystem goods and services. These ecosystems, that are often maintained by direct or indirect groundwater supplies, are collectively known as groundwater dependent ecosystems (GDEs). The importance of groundwater supplies is greatly exacerbated in coastal Mediterranean regions where the lack of surface water and the over-development of anthropogenic activities critically threaten the sustainability of coastal GDEs and associated ecosystem services.

Yet, coastal GDEs do not benefit from a legal or managerial recognition to take into account their specificity. Particular attention should be paid to the characterization of environmental and ecological water requirements. The hydrogeological knowledge about the management and behavior of coastal aquifers and GDEs must be strengthened. These investigations must be supplemented by a stronger assessment of potential contaminations to develop local land-uses and human activities according to the groundwater vulnerability. The quantitative management of water resources must also be better supervised and/or more constrained in order to ensure the water needs necessary to maintain coastal GDEs.

The transdisciplinary approach between hydrogeology, hydrology, social sciences and law is essential to fully understand the socio-economic and environmental complexity of coastal GDEs. Priority must now be given to the development of an appropriate definition of coastal GDEs, based on a consensus between scientists and lawyers. It is a necessary first step to develop and implement specific protective legislation and to define an appropriate management scale. The investment and collaboration of local water users, stakeholders and decision-makers need to be strengthened through actions to favor exchanges and discussions. All water resources in the coastal areas should be managed collectively and strategically, in order to maximize use efficiency, reduce water use conflicts and avoid over-exploitation. It is important to continue to raise public awareness of coastal aquifers at the regional level and to integrate their specificities into coastal zone management strategies and plans. In the global context of unprecedented anthropogenic pressures, hydro-food crises and climate change, environmental protection and preservation of coastal GDEs represents a major challenge for the sustainable socio-economic and environmental development of Mediterranean coastal zones.

Key words: coastal aquifer, coastal hydrosystems, Mediterranean climate, groundwater management, anthropogenic impact

Summary

Introduction

Coastal lagoons cover about 13% of the coastlines from arid to humid environments (Kjerfve, 1994). Being transitional areas from land to sea, the water balance of coastal lagoons is resulting from both terrestrial (fresh groundwater and surface water) and marine water influences. This dual influence allows the development of specific ecosystems that provide a wide range of ecosystem goods and services (Newton et al., 2014; 2018). Over the past few decades, several studies have highlighted the importance of groundwater in maintaining the physico-chemical conditions of these sensitive ecosystems. Coastal lagoons and surrounding wetlands may then constitute "groundwater-dependent ecosystems" (GDEs) (Krogulec, 2016; Menció et al., 2017) and are referred in the document as "coastal GDEs".

The importance of groundwater is further exacerbated in regions suffering from water stress, when surface water is chronically unavailable. Groundwater inputs support or compensate for surface water inputs and play a vital role in maintaining coastal GDEs. This problem is encountered in a majority of coastal regions with an arid or semi-arid Mediterranean climate (Fig. 1) (Köppen, 1936) such as the Mediterranean basin (European Union –EU- and non-EU countries) but also on the southwestern coasts of Australia, Chile and the State of California (United States) and on the southern coast of South Africa. In these regions, referred to throughout this document as "Mediterranean regions", the lack of surface water is combined with a high anthropogenic pressure (UNEP/MAP, 2012). Population growth proceed together with the development and expansion of human activities, such as urbanization, agriculture, tourism and industrial activities (Lotze et al., 2006). Increasing human water needs often lead to overexploitation of aquifers and/or degradation of groundwater quality, which present a risk both to the well-being of human activities and to the freshwater needs of coastal GDEs.

These degradations are expected to be worsen under the effects of climate change. Climatic disturbance in terms of increasing temperatures (Bille et al., 2009; Hallegatte et al., 2009), global hydrological cycle (IPCC, 2014) and sea level rise (FitzGerald et al., 2008; Carrasco et al., 2016; Benjamin et al., 2017) should greatly affect the groundwater and coastal GDEs. This is true not only for the Mediterranean basin, considered as a Hot Spot of climate change, but also for all the Mediterranean regions.

Since the 1990s and the Rio de Janeiro Earth Summit, the conservation, the maintenance of potentialities and the improvement of the ecological status of the coastal water bodies constitute a major concern. Nowadays, a first statement can be made on the progress and limitations of groundwater management strategies and consideration given to coastal GDEs in coastal Mediterranean regions. To this aim, this review proposes to:

- Expose the specificities of coastal GDEs and the key role of groundwater in their sustainable development

- Highlight the vulnerability of coastal GDEs to the socio-economic development and climate conditions of Mediterranean regions

- Revise the consideration given to GDEs and particularly to coastal GDEs in the management policies of Mediterranean regions and discuss their implication for the sustainability of coastal GDEs.

1. Specificities and importance of coastal GDEs

1.1. The wide diversity and essential functions of coastal GDEs

GDEs are defined as *"ecosystems that require access to groundwater on a permanent or intermittent basis to meet all or some of their water requirement so as to maintain their communities of plants and animals, ecological processes and ecosystem services"* (Richardson et al., 2011). This definition clearly expresses the crucial role of groundwater in the functioning of GDEs. However, the multitude of processes and services grouped under the terms "ecological processes" and "ecosystem services" does not necessarily make it possible to understand all the specificities and complexity inherent to certain types of GDEs, such as coastal GDEs. The Table 1 summarizes the morphologic and hydrological characteristics, the hydrological knowledge and the protection and conservation status of 14 of the most studied lagoons present in Mediterranean regions subject to Mediterranean climate (Fig. 1) (Newton et al., 2018; Pérez-Ruzafa and Marcos, 2008).

The coastal GDEs are distinguished by their diversity, making each of them a special case. This diversity is expressed on several levels. From a morphological point of view, water bodies of coastal GDEs are separated from the sea/ocean by a barrier, connected at least intermittently to the ocean by one or more restricted inlets (Kjerfve, 1994). According to the most widely used classification, these coastal lagoons can be classified into three categories including (i) choked, (ii) restricted and (iii) leaky lagoons Kjerfve (1994). These categories reflect the importance of interactions between coastal lagoons and seawater. Choked lagoon are connected to the sea by a single or few narrow and shallow entrances, resulting in delayed and dampened tidal oscillation or low water exchange with the open sea. Leaky lagoons are connected by many entrances to the adjacent sea and are therefore characterized by almost unimpaired water exchange. The stretch of coastal lagoon can greatly vary, from <0.01 km² to more than 10 000 km², as is the size of the hydrological watersheds, without an obvious proportionality relationship between the two (Table 1). If the mean depth can also vary, coastal lagoons still remain shallow water environments, generally characterized by shallow mean depth (< 2m) (Table 1).

Although rainfall, pounding of surface flows or flooding are an important source of water for most of coastal GDEs, groundwater plays also a role in many coastal wetlands (Le Maitre et al., 1999). Coastal GDEs can be completely dependent on groundwater discharge, whilst others may have limited dependence, such as only under dry conditions (Howe et al. 2007). Thus, depending on the hydrologic balance, water bodies of coastal GDEs could vary from coastal fresh-water lake to a hypersaline lagoon.

The fauna and flora that make up coastal GDEs are also very diverse. The type of vegetation and wildlife is mainly defined by the salinity of the water and the moisture level of the environment (permanent, semi-permanent or ephemeral wetlands) but also the location and climate. Several thousand plant species grow in coastal wetlands such as reeds, grasses and shrubs (Frieswyk and Zedler, 2007; Lemein et al., 2017; Ramírez G. and Álvarez F., 2017). Hundreds of animal species can also be listed, including fish, reptiles, mammals, frogs and birds. The degree of dependence of wildlife on coastal GDEs ranges from those who need wetlands for part of their life cycle to those who are totally dependent on them.

The environmental importance of coastal GDEs is greatly recognized for most of them, as evidenced by the establishment of various protection or conservation status (Table1). Because of their relatively low flushing rates, the important availability of nutrients allows high rates of primary production (phytoplankton and aquatic plants) thereby supporting high rate of secondary production (fisheries nurseries) compared to other aquatic ecosystems (Nixon 1995). Coastal GDEs contribute to the overall productivity of coastal waters by supporting a variety of habitats, including salt marshes, seagrasses or mangroves. These habitats host specific and sensitive ecosystems and provide a rich support for biodiversity, including vital habitats for many fish, shellfish and bivalves (Basset et al., 2013). They constitute also refuge from predation, nursery and feeding habitats for estuarine, marines and terrestrial species (Heck and Thoman, 1984; Harris et al., 2004). Many coastal GDEs support a variety of migratory water bird and shore bird species. Some birds depend on coastal GDEs almost totally for breeding, nesting, feeding, or shelter during their annual cycles. The main migratory birds utilizing the coastal GDEs are ducks, shorebirds, gulls, terns and flamingos.

1.2. Ecosystem services and coastal GDEs

Coastal GDEs harbor a large part of the human population that depends directly on these ecosystems (Willaert, 2014) and provide not only livelihoods but also numerous benefits to human health and welfare (Newton et al., 2014, 2018). Coastal GDEs have therefore a socio-economic interest which makes them complex social-ecological systems (Newton et al., 2014; Wit et al., 2017). Since the 1970s, and more particularly in the 2000s, the concept of "ecosystem services" has attempted to express the complex relationship between human communities, their environment and the non-human living beings to which they are linked (Sartre et al., 2014). The "ecosystem services" can be defined as the full range of benefits that humans derive from the functioning of ecosystems. Ecosystem services include 4 major types of services (Blanchart et al., 2017):

- Provisioning services: correspond to direct products provided or produced by ecosystems such as water, food, construction materials,
- Regulating services: include benefits from regulation of ecosystem processes such as carbon storage, climate regulation, flood and erosion protection,
- Cultural services: include nonmaterial benefits from ecosystems such as recreation, aesthetic or educational benefits,

- Supporting services: are related to necessary factors for producing ecosystem services (photosynthesis, nutrient cycle, refuge areas…).

Ecosystem services are linked to the ecological structure and functions of the environment. In coastal GDEs, many ecosystem services are derived or supported by the presence of groundwater inflow because of its role in regulating the hydrology of wetlands and lagoons (UNEP-MAP, UNESCO-IHP, 2015). One of the main ecosystem services provided by coastal GDEs is related to provisioning services (livestock, fishing, aquaculture) (UNEP-MAP, UNESCO-IHP, 2015). Coastal GDEs are highly productive and food provisioning can often be key for regional economy (Newton et al., 2014). For example, the Ria Formosa in Portugal provided up to 90% of the national production of clams (Newton et al., 2003). Coastal GDEs also have a very important place in the hydrological cycle. They contribute to water flow regulation and control and therefore help to flood protection. They also participate to water retention, quality (salinity regulation) and purification. Finally, cultural services, *e.g.* cultural heritage, tourism or aesthetics are also very profitable for several coastal GDEs. In some specific case, such as the Venice lagoon (Italy), cultural services can exceed 5.10^{8} euros/year (Newton et al., 2018).

The various protection and/or conservation status applied to coastal GDEs (Table 1) does not necessarily involve a high level of knowledge of the hydrosystems' behavior. For a large majority, the role and the dependence on groundwater is largely under studied, even if it is suspected (Table 1). Very few coastal lagoons have a sufficient level of knowledge to understand their level of dependence to groundwater (Table 1) and then developed sustainable methods/policies to ensure their conservation. Moreover, even in the case of good knowledge of hydrological functioning and establishment of a conservation/protection status, it does not seem to guarantee the good state of these environments (Leruste et al., 2019; Leterme et al., 2015). The lack of hydrological knowledge then appears to be as much a problem as the lack of specific protection status adapted to the particular cases of the GDES.

1.3. Understanding the dependence on groundwater supplies

Under natural conditions, without pumping, fresh groundwater flows from recharge to discharge areas (Fig. 2). Local groundwater flow is mostly near the surface and over short distances, i.e. from a higher elevation recharge area to an adjacent discharge area. In this case, the discharge of the aquifer (Fig. 2) occurs as diffuse outflow, as for coastal GDEs. Coastal GDEs are thus relying on the surface expression of groundwater (Richardson et al., 2011). On a larger scale, over long distances, groundwater flow is preferentially at greater depths and fresh groundwater meets salt marine water at depth in the transition zone. The discharge of groundwater is composed by two processes: i) the discharge of fresh groundwater (fresh submarine groundwater discharge, FSGD) toward the sea and the discharge of saline groundwater (recirculated submarine groundwater discharge, RSGD) (Fig. 2). Groundwater supplies to coastal GDEs can originate from one or several aquifer formations of variable nature and extension (Table 1). This dependence on groundwater can be variable, ranging from partial and infrequent dependence (seasonal or episodic) to total, continual dependence (Hatton and Evans, 1998).

Groundwater and surface water are the most often characterized by strong interactions (Fig. 2). These interactions result in groundwater discharge to the river (groundwater discharge, Fig. 2) or, conversely, in aquifer recharge through river and lake water infiltration (Fig. 2). Rivers and streams that flow all year (perennially flowing) are often groundwater dependent because a significant proportion of their daily flow is supported by the groundwater flow discharging into the river course (Acuña et al., 2005; Bonada and Resh, 2013; Datry et al., 2014). Groundwater is particularly important in arid and semi-arid regions and in case of extended dry periods, during which evaporation markedly exceeds precipitation and surface water is scarce or even disappeared (Eamus et al., 2006). Both groundwater and surface water flow toward the lagoon, which constitute the last collector of the watershed (Fig. 2). The discharge of groundwater toward coastal GDEs can be either directly into the wetland or indirectly via the river (Fig. 2).

For a long time, groundwater studies in coastal areas focused mainly on seawater intrusion impacting coastal aquifers. The groundwater has only recently been recognized as important contributors to hydrological and biogeochemical budgets of coastal environments such as coastal GDEs (Table 1) (Johannes, 1980; Burnett et al., 2001, 2006; Slomp and Van Cappellen, 2004; Moore, 2006, 2010; Rodellas et al., 2015; Luo and Jiao, 2016; Malta et al., 2017; Correa et al., 2019; David et al., 2019). The presence of groundwater drives the evolution, persistence and resilience of coastal GDEs and their ecosystems on at least two aspects including i) physical characteristics, such as the quantity, location, timing, frequency and duration of groundwater supply (Jolly et al., 2008; Rodríguez-Rodríguez et al., 2008; Bertrand et al., 2012, 2014) and ii) chemical characteristics (Burnett et al., 2006; Moore, 2010), such as water quality (Ganguli et al., 2012), salinity (Menció et al., 2017), nutrient concentrations (Szymczycha et al., 2012; Ji et al., 2013; Rodellas et al., 2015; Hugman et al., 2017) and temperature (Brown et al., 2007; Richardson et al., 2011). Although recognized as essential, the characterization of coastal hydrosystems' behavior still remains under studied in many cases (Table 1) due to the important monitoring and financial resources required to improve their understanding.

1.4. Groundwater dependence monitoring

The "Groundwater dependence" clearly expresses that the prolonged absence of groundwater as well as its quality degradation have a negative impact on the growth, health, composition, structure and function of the ecosystem. Potential threats to groundwater inflow toward the coastal GDEs can be assessed through the study of the groundwater flow paths, the spatial and temporal variability of groundwater discharge and surface/ground water interactions (Kløve et al., 2011). Yet, the groundwater dependence of coastal GDEs remains still difficult to characterize. This difficulty is exacerbated by the thinness of the unsaturated zone, *i.e.* the thickness of the soil between the soil surface and the top of the saturated zone, which allows important mixing between surface and ground waters. Differentiating and quantifying the contribution of these end-members is highly complex. A wide range of methodologies have been developed to improve the understanding of coastal GDEs (Sophocleous, 2002; Kalbus et al., 2006; Howe et al., 2007). First of all, the monitoring of groundwater levels and the establishment of piezometric map are often the first steps to highlight the groundwater dependence of coastal GDEs (Sena and Teresa Condesso de Melo, 2012). Then, in the particular case of coastal GDEs, the two main approaches commonly used to assess surface/ground water interaction are i) temperature, geochemical and isotopic tracers (Dimova et al., 2017; Duque et al., 2016; Mudge et al., 2008; Sadat-Noori et al., 2016; Sánchez-Martos et al., 2014; Santos et al., 2008; Schubert et al., 2011) and ii) numerical modeling (De Pascalis et al., 2009; Martínez-Alvarez et al., 2011; Sena and Condesso de Melo, 2012; Read et al., 2014; Menció et al., 2017). Less common approaches, such as geophysical method can also be carried out to obtain information on the spatial scales and dynamics of the fresh water–seawater interface, the rates of coastal groundwater exchange and the total fresh water discharge (Dimova et al., 2012).

2. Dominant human and climatic stressors on groundwater and consequences for coastal GDEs in Mediterranean regions

Although essential, coastal GDEs are one of the most threatened ecosystems in the world. Human activities are exerting increasing pressure on these sensitive systems or on the resources on which they depend, such as groundwater. Water withdrawal, drying, pollution, habitat destruction or overexploitation constitute the main causes of their degradation (Millennium Ecosystem Assessment, 2005). More than 50% of wetlands have disappeared during the 20th century in some regions of Australia and Europe (Millennium Ecosystem Assessment, 2005). Only in the Mediterranean basin, national or sub-national datasets suggest a probable loss of 50% of its wetlands (Perennou et al., 2012). In the specific case of coastal wetlands, global losses are estimated at between 64 % to 71% during the 20th century (Gardner et al., 2015).

The characteristic overdevelopment of coastal Mediterranean regions has already led, for several decades, to a significant pressure on groundwater resources. The growing drinking, industrial or agricultural water requirements tend to the overexploitation of the coastal aquifers. Coastal aquifers are threatened by both horizontal exchanges with seawater and vertical infiltrations of pollutants. The development of human activities often constitutes an important source of pollutants and groundwater can constitute an important vector of pollution towards the coastal GDEs (Moore, 2006).

2.1. The harmful human overdevelopment of coastal Mediterranean regions

The strong and increasing urbanization as well as fast growing demography represent the two main pressures. For example, in Australia, more than 85% of the population is living within 50km of the sea. The population density of Australian's coastal areas increased by 14% between 2001 and 2009, from 3.75 hab/km² to 4.27

hab/km² (Fig. 3a). A very important difference is observed for the urban, coastal population. The population density measured in coastal capital cities is 94 times higher than the average population density of coastal areas (Fig. 3a).

In the Mediterranean basin, the coastal population grew from 95 million in 1979 to 143 million in 2000 and could reach 174 million by 2025 (UNEP/MAP, 2012) (Table 1). In the Mediterranean basin' population, France is the 3rd most populated country (after Turkey and Egypt) (UNEP/MAP, 2012) and allows for a good observation of the attractiveness of the Mediterranean coastline (Insee/SOeS, 2009). Indeed, among the 3 French coasts (Mediterranean, Atlantic and Channel coasts), the Mediterranean coast is clearly distinguished by a rapid population growth (Fig. 4) (Insee/SOeS, 2009). Between 1960 and 2010, the French Mediterranean coast recorded the highest population increase with 56%, although it is the least extensive coastline (Fig. 4). The highest growth of population rate is recorded in the Mediterranean island of Corsica, with an annual increase of 1.3% between 2006 and 2010. The coastal municipalities accounting for 80% of the Corsican population and 30% of the urbanization is concentrated within 1 km of the shoreline (SDAGE, 2015).

In USA, California tops the coastal populations chart. Currently, of the total population of 39.6 million in California, 69% is living in coastal areas (U.S. Census Bureau, 2019) and 95% is living in urban areas. Coastal population density is 3 times higher than the state' population density (Fig. 3b). In less than 60 years, coastal population density went up by a factor of 2.5, from 135.6 hab/km² in 1960 to 278.4 hab/km² in 2017 (U.S. Census Bureau, 2019) (Fig. 3b). In the major coastal cities, such as San Francisco and Los Angeles, population density exceeds several thousand inhabitants per km². In 2018, population density was 7003 hab/km² and 3230 hab/km² respectively.

This demographic growth is accompanied by a very fast development of urban infrastructure. In the Mediterranean basin, the urbanization increased from 54% in 1970 to 66% in 2010 (Table 1) and the urban coastal population could increase by 33 million between 2000 and 2025 (UNEP/MAP, 2012). The South and the East Mediterranean countries (Non-EU countries) are urbanizing more rapidly than the rest of the world. These that were essentially rural countries, with average urbanization of 41% in 1970, will become urban countries, with 66% urbanization by 2025 (UNEP/MAP, 2012). This tendency is also observed in Australia. Peri-urban and rural cadastral parcels are progressively replaced by urban areas leading to an increased artificialization of coastal areas (Clark and Johnston, 2017).

2.2. Perturbations induced by groundwater degradation

2.2.1. Reduction of groundwater inputs and coastal GDEs dewatering

The modification of fresh groundwater flowing to the lagoons disrupts the fragile balance of the coastal GDEs' ecosystems. As surface water is limited and increasingly affected by pollution and eutrophication, the exploitation of groundwater from coastal aquifers as a source of freshwater has become more intense (Bocanegra et al., 2013; Liu et al., 2017). The number of groundwater abstraction infrastructures have drastically increased. This process is the one most frequently exacerbated by unsuitable water resource management plans and/or poor control of water extraction facilities. Unregulated but also illegal pumping draws a high and unreasoned amount of water which is uncountable in the water management policies and leads to groundwater depletion and reduce river, spring and wetland flows. The progressive lowering of the groundwater level reduces or removes the connections between the aquifer and the coastal GDEs. As a result, aquatic vegetation in these transitional wetlands is gradually being replaced by terrestrial vegetation. This process leads to the drying, reduction and disappearance of coastal GDEs. In the worst case, changes in the structure and the functioning of the ecosystem (Balasuriya, 2018; Pérez-Ruzafa et al., 2019) results in a partial or total loss of ecosystem services provided by coastal GDEs.

Anthropogenic activities require a growing demand for space for agricultural production, housing or industrial land use. The land gain can be achieved by the conversion of natural lands or by partially or totally draining wetlands (El-Asmar et al., 2013). The construction of artificial drainage network in order to control the humidity is an old and relatively common practice (Gerakis and Kalburtji, 1998; Avramidis et al., 2014). These practices are highly constraining for the hydrosystems. They drastically alter the natural flow of surface groundwater and greatly affect the coastal GDEs, which are relying on the surface expression of groundwater.

Changes in land use can have a significant impact on aquifer recharge processes and thus on fresh groundwater supplies to coastal

GDEs. . Infiltration is increasing with the proportion of bare soil and evapotranspiration’s patterns are conditioned by the type and the stages of crops development. Soil compaction by urbanization or intensive agriculture may reduce the infiltration and enhance the surface runoff (van den Akker and Soane, 2005; Gregory et al., 2006; Nawaz et al., 2013). In addition, the urban pavement of the shore (El-Asmar et al., 2013) makes the soil impermeable and drastically reduces infiltration and recharge into the aquifer. 40% of the 46,000 km of Mediterranean coast were already artificialized in 2000 and it is expected to exceed 50% by 2025 (AViTeM, 2018).

If groundwater extraction is clearly the main threat in coastal Mediterranean regions, it is important to underline that increasing groundwater flow is also problematic. Some activities, such as irrigation, terracing, land-clearing or managed artificial recharge of aquifers, can appreciably increase the permeability of upper soils and then lead to the increase of the aquifer recharge (Baudron et al., 2014). In urban areas, tap water leaks can also constitute a significant source of groundwater recharge (Minnig et al., 2018; Vystavna et al., 2019). The flow of fresh water to the coastal GDEs can therefore be significantly increased. The physical and chemical disturbances can disturb and modify bio-community structure of the coastal GDEs.

2.2.2. The role of groundwater as a vector of pollution

Coastal GDEs often represent the last collector of water and their quality degradation results, and reflects human activities over the watershed. Anthropogenic activities such as the demographic, economic, industrial and commercial development often introduce new potential contamination sources (Appelo and Postma, 2005) which infiltrate towards the aquifer.

In the coastal Mediterranean regions, the main problem is related to the sewage inputs. The fast growing of urbanization is not always accompanied by the development of sewage infrastructures that results in less efficient treatment of urban wastewater and sewer leaks (Michael et al., 2013). In the Mediterranean basin, almost 40% of coastal settlements with more than 2000 inhabitants do not have any wastewater treatment plant (UNEP/MAP, 2012). This problem is especially exacerbated on the southern Mediterranean basin due to the rapid growth of many coastal cities and towns. In addition, coastal Mediterranean regions are privileged tourism destinations (UNEP/MAP, 2012). The touristic flow picks lead to higher rates of sewage inputs in urban sewerage networks that are often aged and failing. Wastewater and associated pollutants from domestic and industrial sources consequently infiltrate towards the aquifer or through the interaction between groundwater and river water (McCance et al., 2018; Erostate et al., 2019; Koelmans et al., 2019; Vystavna et al., 2019). Nitrogen pollutants, phosphorus, but also organic compounds and heavy metals are the most frequent contaminant affecting the groundwater resources (Wakida and Lerner, 2005; Petrie et al., 2015; Xu et al., 2019). The second main source of groundwater quality degradation is the agricultural activity. The excess of nutrients from fertilizers (nitrogen and phosphorus), pesticides, emerging compounds and, less frequently, pathogenic microorganisms related to agricultural activities contribute to the degradation of both ground and surface water quality (Symonds et al., 2018; Xin et al., 2019).

Once infiltrated, the pollutants follow the groundwater flow and can migrate to coastal GDEs (Rapaglia, 2005; Knee and Paytan, 2011; Jimenez- Martinez et al., 2016; David et al., 2019). According to the temporal dynamic of the aquifer, groundwater can represent a direct short and/or long term vector of pollution for coastal GDEs. Groundwater with short residence times (a few years) into the aquifer will rapidly flow towards the lagoons, carrying pollutants along its way. In case of groundwater with long residence time (several decades) and if no remediation process occurs, pollutants can be accumulated into the aquifer for several decades. The currently observed groundwater contamination can therefore be the result of the legacy of pollution related to human activities previously developed over the watershed (Erostate et al., 2018). This groundwater archiving capacity allows the storage of pollutants that will reach the coastal GDEs in the future.

Once the pollutants are in the coastal GDEs, prolonged groundwater residence times favor the accumulation of pollutants in water but also in aquatic organisms. The progressive accumulation of pollutants, especially heavy metals, along the food chain can pose serious human health issues and greatly impact economical profit by deteriorating ecosystems services such as aquaculture and fisheries. The most frequent impact of exceed in nutrients, sediments and organic maters is the eutrophication which can lead to important degradation or loss of seagrass beds, community structure and biodiversity (National Research Council, 2000; Pasqualini et al., 2017). More than 400 coastal areas have been identified worldwide as experiencing some form of eutrophication (Selman et al., 2008).

2.2.3. Impacts of climate change on aquifer recharge and implications for coastal GDEs

Important changes regarding the aquifer recharge in terms of timing, duration and magnitude (McCallum et al., 2010; Hiscock et al., 2012; Taylor et al., 2013) as well as the storage and the quality of groundwater are expected in a context of climate variability. These modifications will be more pronounced in arid regions and especially in the Mediterranean basin, considered as a Hot Spot of climate change (IPCC, 2014). By the middle to the end of the century, the southern European regions as well as Australia are expected to suffer from increasing arid conditions with longer and more frequent droughts (Stigter et al., 2014) due to the increase in the temperature (Ducci and Tranfaglia, 2008; McCallum et al., 2010), in evapotranspiration (Hiscock et al., 2012), modification of seasonal patterns of precipitation (Polemio and Casarano, 2008; Stigter et al., 2009; Barron et al., 2011) and of average effective infiltration (Ducci and Tranfaglia, 2008). An amplification in the frequency and intensity of drought is also expected in the southern Mediterranean basin, such as in Morocco (Stigter et al., 2014).

The results of predictive models to assess the impact of the climate change on aquifer recharge are often highly variable. The main tendency highlights a decrease in the groundwater recharge in Mediterranean regions, leading to a significant loss of groundwater resources (IPCC, 2007; Barron et al., 2011). In the Mediterranean basin, the decrease of the recharge can reach 30% to up to 80%

(Ducci and Tranfaglia, 2008; Döll, 2009; Moseki, 2017). Modification in coastal aquifer recharge as well as the expected sea level rise (Hertig and Jacobeit, 2008; Somot et al., 2008; Mastrandrea and Luers, 2012) can lead to the inland migration of the mixing zone between fresh and saline water .

Climate change will exacerbate existing pressures rather than bring a new set of threats. With the water requirements that are projected to increase under a drier climate, severe water shortages can occur. The outflow into the coastal GDEs can be strongly reduced by the end of the century which could accelerate their drying up. Groundwater degradation by salinization could also greatly affect the physico-chemical conditions and thus the ecosystem balance of the GDEs lagoons. In response to these treats, a decrease in groundwater abstraction and an appropriate management appear as the principal way to ensure the preservation and sustainability of coastal GDEs (Candela et al., 2009; Stigter et al., 2014).

There may be exceptions to this general trend at the local level. In some cases, the modification of rainfall patterns and/or land uses modification can favor the recharge of the aquifer and improve the groundwater quality (Cartwright and Simmonds, 2008; Crosbie et al., 2010; Santoni et al., 2018). For example, in the Murray-Darling Basin in Australia, the clearing of the native vegetation is likely to favor the infiltration and increase the recharge of 5% for future climate around 2030 (Crosbie et al., 2010). If land-clearing could favor the recharge, the strong alteration of the hydrological cycle by vegetation cutting also has strong negative aspects which should be underlined. Among others things, land-clearing can increase runoff and streamflow, favor soil erosion, massive drainage of natural nutrients and salinization of soils and waters (Koivusalo et al., 2006; Cowie et al., 2007; Peña-Arancibia et al., 2012; Kaushal et al., 2018; Cheng and Yu, 2019). The consequences of these practices are often irreversible. Yet, for watersheds severely degraded by salinization, this increase in recharge could help the dilution and potentially improve quality of groundwater (Cartwright and Simmonds, 2008).

The existence of local specificities shows the importance of establishing adaptive case-by-case water management strategies. Water resource management requires the definition of appropriate management scale which makes it possible to manage the hydrosystem as a whole, taking into account the complexity of interactions between water bodies but also between humans and their environment.

3. Management strategies and current considerations for coastal GDEs

3.1. From international environmental awareness to Integrated Water Resource Management

The definition and establishment of water resources management strategies and policies result from an awareness of environmental issues initiated in the 1970s, with in particular the Stockholm Earth Summit in 1972 (Fig. 5a). This ecological awakening then continued in the 1980s with a collective awareness of the existence of pollution and harmful disruption on a global scale. It is in this context that the

Bruntland Report define for the first time in 1987 the concept of "sustainable development": *"The sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs"*. This report requires the management of water resources as a common heritage and lays the foundations for integrated natural resource management. Only 5 years later, the Rio Earth Summit marked a turning point in the sustainable management of water resources with the "rediscovery" of the concept of Integrated Water Resource Management (IWRM) (Petit, 2006) and Integrated Coastal Zone Management (ICZM) (Deboudt, 2005).

These two concepts, which appeared in the 1970s (Deboudt, 2005; Petit, 2006), were then highlighted in the 1990s through the media coverage of the Rio Earth Summit and became a key concept in the 2000s thanks to the launch of the concept of sustainable development on the international political scene. In 2000, the Global Water Partnership, an international network created to advance governance and management of water resources, published its first's report on IWRM and clearly define the concept as a "process which promotes the coordinated development and management of water, land and related resources, in order to maximize the resultant economic and social welfare in an equitable manner without compromising the sustainability of vital ecosystems" (GWP, 2000). The IWRM was and remains widely promoted by many international organizations or donor agencies (Rahaman and Varis, 2005; Biswas, 2008), as a strategic approach to water management (Meublat and Le Lourd, 2001). The Johannesburg Earth Summit in 2002 even recommended its implementation in all countries by 2005. This summit also insists on the establishment of ICZM. Sharing the same precepts as IWRM, ICZM is nevertheless committed to taking into account the specific risks associated with water on the coast (Morel et al., 2004). ICZM is developing rapidly, particularly in Europe, thanks to its institutionalization and recommendation of the Council and the European Parliament in 2002 (Ghézali, 2009). Although coastal GDEs are in theory elements in their own right in integrated management strategies, they are still too often forgotten and do not benefit from legal or managerial recognition to take their specificity into account (Cizel, 2017).

3.2. Integrated groundwater management without specific regards for coastal GDEs

Since the 2000s, we have seen an acceleration of sustainable resource management measures at the global, regional and national levels (Fig. 5a). GDEs have been partially propagated in water management policies developed over the past two decades, that recognize a link between groundwater and surface water. Some countries or group of countries particularly vulnerable to shortage of water and repeated severe droughts *e.g.* Australia, countries of the EU, the United-States (California) and South Africa, have yet incorporated specific reference to general GDEs into the legislation. Even if the protection of GDEs is included under water management policies, the implementation of an appropriate management policy is often lacking (Rohde et al., 2017).

Countries of the EU and Australia are the first to have included GDEs in their legislative framework (Rohde et al., 2017). The French model of water management by Water Agencies (created by the law of 1964) and the Australian model, derived from the experience of the Murray Darling Basin (Murray Darling Basin Authority created in 1987) are often considered as a reference model in terms of river basin management (GWP/RIOB, 2009; Brun and Lasserre, 2018). Legislative framework and groundwater managerial strategies set up by the EU and Australia however have shortcomings that undermine their effectiveness in protecting the resource (Fig. 6).

Australia provides the most comprehensive groundwater governance (Ross, 2016). As early as 1994, the agreement of the Council of Australian Governments (COAG) (Fig.5c) required the development of a comprehensive system of water allocations and rights to ensure better, more sustainable water management. The water reform program initiated by the COAG agreements was then updated in 2004 by developing a new National Water Initiative (NWI) (Fig. 5c). The NWI - currently signed by all states and territories - has been recognized as the national blueprint for water sector reform to improve the state of industry and provide long-term environmental benefits (Willett, 2009). The annually adjustable water entitlements and related water market provide a great flexibility and a better adaptability to the state of the resource (Ross, 2016). However, monitoring of groundwater quality is limited (except for drinking water) and is often carried out on a short-term basis without consistent national program (Geoscience Australia, 2010). In Europe, on the other hand, both the quantitative and qualitative aspects benefit theoretically from an equivalent level of attention. The legislative framework implemented by the Water Framework Directive of 2000 (WFD) (Fig. 5b) provides thus the most comprehensive groundwater protection (European Commission, 2008; Ross, 2016). Member states are required to preserved the groundwater quantity and quality based on threshold values established to prevent any significant diminution of the ecological or chemical quality of surface water nor in any significant damage to terrestrial ecosystems which depend directly on the groundwater body (European Directive 2000/60/CE). The degree of freedom given to the member states to define groundwater and GDEs management plans and the wide disparity between them can yet reduce the enforcement of EU recommendations (Liefferink et al., 2011). While some countries are considered as models for their efficiency in water management, such as France, Spain or Germany (Rahaman and Varis, 2005) (Fig. 5b, c), others are experiencing significant delays in the transposition of the EU recommendations (Ghiotti, 2011). In EU frameworks, an important point of divergence is the concept of “water bodies” that supports the WFD. This concept requires precise identification, delimitation and definition. However, the scientific knowledge is often incomplete or inaccurate and fails to provide the appropriate level of precision (Bartout, 2015). The lack of knowledge represents a significant bias for the definition of priority actions and the implementation of effective public policies to achieve the good qualitative and quantitative status set by the European recommendations (Maillet, 2015).

These two management models, one based on strong qualitative regulation of the resource (Australia) and the other on the monitoring of threshold values (EU), lead to significant disparities in GDEs management. In Australia, management decisions are based on an ongoing monitoring and research which help to establish an adaptive GDEs management (Richardson et al., 2011; Rohde et al., 2017). The great adaptability of annual water allocation allows a better consideration to the vulnerability of GDEs, particularly in a case of severe drought. However, the poor water quality monitoring exposes lagoons to high risks of undetected contamination. Efforts made for the qualitative management of the water resource clearly need to be completed and reinforced by an improvement of groundwater quality management to ensure the preservation of GDEs (Ross, 2016). In EU, monitoring threshold values allows a better understanding and thus, a better prevention of qualitative and quantitative degradation risks for GDEs. The groundwater allocation is often included in river basin plans of member states but the adaptability of water withdrawals, particularly in the event of drought, can lack reactivity and damage the GDEs (Sommer et al., 2013; Stein et al., 2016). To really benefit from the European directives, particular attention must be paid to their concrete application in all member countries. In addition, the concept of “water bodies” must be better defined in order to enable the implementation of truly effective public policies.

In the particular case of coastal lagoons, considered by the WFD as “transitional water bodies”, the lack of knowledge and data in the early 2000s has triggered the development of monitoring networks implementation. Indeed, the monitoring programs developed for freshwater ecosystems are not relevant for coastal GDEs. These transition environments are subject to many influences that induce a large variation in physical parameters, including salinity. The consideration of biological indicators and the evaluation of shifts in the species presence on coastal ecosystems has emerged as a valid strategy to characterize ecological status (Delpech et al., 2010; Pérez-Domínguez et al., 2012). This approach, followed in the same way by several EU countries, has led to the creation of indicators validated by the EU to improve the assessment of the status of transitional water bodies in the North-East Atlantic (Le Pape et al., 2015). For the Mediterranean region, this work has yet to be completed. Currently, only Greece, Italy and France have developed classification tools, but further developments are still needed to properly assess the ecological status of coastal lagoons (Le Pape et al., 2015).

Even if the groundwater resource management plans help to manage GDEs, specifics on GDEs management are often lacking (Rohde et al., 2017). Coastal GDEs form part of a continuum between continental and marine ecosystems and share common characteristics, species and ecological functions (Pérez-Ruzafa et al., 2010). Inland and coastal waters must be managed as a whole and coordination at river basin and coastal sea levels is required (Pérez-Ruzafa and Marcos, 2008). The IWRM is generally focused on the inland watersheds but likely neglects coastal specificity. Conversely, ICZM focuses exceptionally on coastal areas. However, the coastal area rarely extends to the entire watershed, which influences the quality and quantity of water resources that reach the coast. The link between IWRM and ICZM appears essential to respect the physical, ecological and social continuum of watersheds and their coastal zones.

3.3. Limitations of the project-based approach

The IWRM does not automatically lead to the sustainability of resource uses, although it is a prerequisite (Aubin, 2007). The project-based approach, often applied in environmental protection, makes it difficult to develop a coherent policy. Encouraged by cooperation projects, several countries have tried to initiate the IWRM (Garnaud and Rochette, 2012). This is particularly the case in non-EU countries, such as Morocco and Algeria (Vecchio and Barone, 2018). The coastal GDEs of Nador (Morocco) (Fig. 5b) constitutes a representative example (Garnaud and Rochette, 2012).

Since the 1970s, coastal development has been announced as a priority by the Moroccan government, but there is no national public policy for coastal areas. The growing development exerts a strong pressure on the coastal GDEs, classified as RAMSAR site (Nakhli, 2010). The Nador lagoon is thus the subject of a succession of projects (Fig. 5b) whose objective is to establish a sustainable management of this area (Garnaud and Rochette, 2012). To be "sustainable", resource management must yet be both based on previous actions and forward-looking. Most often, projects follow one another, without taking into account previous results. The standardized procedures proposed by donors do not sufficiently take into account the specificities of the territories. The multiplicity of projects is often counterproductive and compromises the effectiveness of this environmental development assistance. The succession of projects without convincing results ends up reducing the mobilization of local actors and users. This generally too short-term approach limits the involvement and appropriation of target actors. This problem of appropriation is in addition to the problem of the limited funding period, which threatens the sustainability of the actions undertaken (Garnaud and Rochette, 2012). By the end, Morocco's commitment to Integrated Coastal Zone Management (advocated by the - too short - Cap Nador project, from 2006 to 2008) finally found little support in these international collaborations (Garnaud and Rochette, 2012).

4. Better global understanding for a better management of GDEs

Due to their complexity, the development of management strategies adapted to coastal GDEs is particularly complex because it requires a strong transdisciplinary approach. Scientists in the technical sciences (at least hydrology, ecology, hydrogeology, oceanography) need to develop collaborative approach between them but also with social and legal scientists. Although difficult and slow to implement, this transdisciplinary approach has two major advantages. Firstly, it allows scientists to question their own discipline, in particular by putting into perspective the relevance of their own concepts and methods. Then, the development and construction of common methods and concepts results from a shared reflection. These new concepts are thus more relevant because they come from a collaboration work and not from the interweaving of specificities borrowed from each discipline.

4.1. Improving the understanding of GDEs

The improvement of GDEs' management inevitably involves an increasing knowledge of their hydrogeological and ecological condition and processes (IAH, 2016). This information is the most often unavailable and gaps at the intersection of groundwater hydrology and ecology do not facilitate the study of GDEs (Tomlinson, 2011). These gaps are even more important in the case of coastal GDEs which require collaboration between terrestrial hydrology and marine sciences - two epistemic communities that are not necessarily, or very rarely, used to working together. In addition, the implementation of the necessary monitoring systems to improve the understanding of GDEs is often financially and technically expensive and/or difficult to implement (Bowmer, 2003; Roll and Halden, 2016). Improving the management of coastal GDEs inevitably requires the management and understanding of hydraulic processes throughout the water cycle (fresh and salt water).

To overcome the lack of knowledge about GDEs, EU countries and Australian Government and the scientific community have been working together to establish practical guides. These "GDE practical guides" can in theory assist state agencies in the identification and management of GDEs for water management plans (Clifton et al., 2007; Richardson et al., 2011; Hinsby et al., 2015). They offer a range of methods for determining ecosystem reliance to groundwater and help water managers conducting the necessary technical investigations and monitoring protocols to define ecological water requirements for GDEs. In practice, these often complex guides seek data keys to understand all types of systems but each GDE is an individual case, having specific characteristics and behavior that prohibit any generalization of diagnoses and solutions. The identification of appropriate study tools requires significant scientific support and the evaluation and monitoring of the relevance of the tools used is yet another debate.

Generally, the improving of knowledge depends on the strategic and economic interest of GDEs, assessed by the costs and benefits related to their protection (Millennium Ecosystem Assessment, 2005). The "ecosystem services approach" of the United Nations Millennium Ecosystem Assessment Project thus recommend to complete the technical approach of GDEs by a relevant assessment of the GDEs' valuation and relationship between ecosystems and human well-being. While the evaluation of ecosystem services tends to highlight man's dependence on his environment, this economist approach to nature raises two concerns. Firstly, this new way of thinking about nature conservation places nature at the service of mankind (Dufour et al., 2016). GDEs are then considered as providers of valuable goods and services. The diversity and complexity of the relationship between humans and nature cannot be summarized as a monetary evaluation exercise (Sartre et al., 2014). Moreover, human societies had already understood the importance of coastal GDEs and how to benefit from them well before the concept of "ecosystem services" was adopted. Secondly, the economic assessment of GDEs requires a clear definition of the benefits of these ecosystem services including direct (fish and plant production, water storage and purification…) and indirect values (cultural, aesthetic, social reasons…) to the human population (IAH, 2016). Estimating the economic values of ecosystem services is far from easy. Recreation and tourism are the most easily quantifiable services, firstly because the direct revenue they generate are easily quantifiable but also because they receive special attention due to the attractiveness of coastal GDEs (Rolfe and Dyack, 2011; Clara et al., 2018). On the other hand, essential services such as protection against erosion, climate regulation or pollution control are neglected, largely underestimated and/or under-studied due to the lack of available data (Barbier et al., 2011)

4.2. Determining the appropriate management scale

The watershed is considered as the most environmentally and politically relevant management unit. This watershed-based approach can contribute to reinforce the lack of consideration given to "hidden" groundwater resources, while they are essential to establish an integrated management of GDEs. An appropriate management scale is a necessary first-step for the sustainable management of supporting aquifers and of the coastal GDEs (Bertrand et al., 2014; Vieillard-Coffre, 2001).

Firstly, surface and ground water are not constrained by the same geological boundaries. The hydrogeological and hydrological watershed do not necessarily (or rarely) overlap (Affeltranger and Lasserre, 2003). The extension of an aquifer and the drained groundwater can extend well out of the boundaries defined by the hydrological basin. Human activities developed outside the hydrological basin can impact qualitatively and/or quantitatively the groundwater resources flowing within the basin and/or hydraulically connected. A significant water supply-demand gap can therefore be induced. A broader consideration of a "water-supply area" would allow a better assessment of the water resources actually available. This approach would ensure a better allocation of water between human and ecosystem needs.

Surface and groundwater have very different flow dynamics. Groundwater flow takes on average several years, even centuries, compared to a few days or a few weeks for river water (Fetter, 2018). The capacity of recharge and renew is much longer. Their inertial behavior supports their capacity to accumulate the pollutants and to record the degradation caused by human activities over several decades (section 3.2.2.). The positive or negative effects of the land use planning made over the hydrological basin can take several decades or even centuries before being noticeable on groundwater quality and quantity (Boulton, 2005). The notion of sustainability preached by IWRM can then be strongly questioned if the groundwater dynamics are not enough understood and/or not considered by management strategies.

The existing hydraulic exchanges between the different water bodies and the vertical linkages are not always fully appreciated (Boulton, 2000). Part of the problem relates to the difficulties of assessing groundwater volumes, recharge rates and sources but also to the low recognition of the linkages between groundwater and many surface water ecosystems (Boulton, 2005). The qualitative and quantitative status of a water body has an impact - positive or negative - on all the water bodies connected. It is then important to understand the existing relationships between the aquifer and all the other water bodies, which means neighboring aquifers, fresh surface water and brackish surface water.

More and more water resources managers are becoming familiar with the necessity of considering large spatial areas to establish a relevant water management (Boulton, 2005). Even if their perceptions of hydrologic interactions are often restricted to lateral and longitudinal flows (Pringle, 2003), the importance of vertical connectivity is slowly being appreciated (Boulton, 2000). A greater consideration of the ecological processes that support the proper functioning of the GDEs is being given. The study of the "proper functioning areas" of GDEs would define the extension of the surrounding area that supports the ecological processes that ensure the sustainability and resilience of the wetland (Chambaud and Simonnot, 2018). It would take into account all the factors that contribute to the functioning of the GDE, *i.e.* water qualitative and quantitative supply, but also animal species for which all or part of the life cycle occurs near the GDE and the connectivity of the GDE with other biodiversity reservoirs, animal and plant populations.

4.3. Partnership, appropriation and relevant definition of coastal GDEs

The efforts required to establish effective multi-scale governance are not often sufficient to ensure the sustainable management of groundwater and GDEs (Molle et al., 2007) (Fig. 7). Several shortcomings already mentioned above, partially explain these difficulties (Fig. 7). The development of regional guidelines based on too approximate or minimalist knowledge of GDEs, inevitably leads to inconsistencies in management strategies at the local level. Coastal GDEs often suffer from incomplete, inappropriate or even contradictory definitions. Scientific definitions are sometimes in conflict with legal definitions and make the recognition and conservation of these environments more complex (Cizel and Groupe d'histoire des zones humides 2010; Cizel, 2017). Coastal GDEs are often recognized and grouped into the large family of wetlands. A simplification that does not take into account their specificity, consisting of a wetland, a water body and an aquifer, all hydraulically connected, which must be recognized and managed as an inseparable whole. Improving the definition of coastal GDEs is essential both to better understand and to delimit them, but also to develop and to apply specific and appropriate protective legislative acts.

While the advancement of scientific knowledge and its better consideration at the regional level could be a way to improve the management of GDEs, a large part of the solution also seems to come from the local level. At the local scale, collaboration between water stakeholders for integrated resource management can be complicated (Chanya et al., 2014; Mostert, 2003). The initial appropriation by state entities (Water Agencies or Basin Organizations) of the recommendations formulated by regional and national institutions often appears insufficient for the local implementation of adapted and sustainable management strategies (Fig. 7). A real appropriation of existing regulations on coastal GDEs by all local stakeholders, decision-makers and actors in the territory appears essential for the preparation of relevant planning or development documents and the implementation of appropriate action programs. The elements required to define the challenges and perspectives related to GDEs must not be a local adaptation of regional recommendations but rather a collective elaboration by all the actors concerned. Efforts must be made to develop a framework for effective public participation at six levels: information, education, consultation, involvement, collaboration and capacity building (Das et al., 2019).

Coastal aquifers are particularly vulnerable to water users conflicts (Zepeda Quintana et al., 2018). All water users want to be able to benefit from the quality and quantity of water resources they need. No user can be abandoned in favor of another, nor can the need for environmental waters. Environmental water needs cannot be forgotten and must be taken into account in management strategies. Sustainable water management thus requires water demand management, which must be achieved through agreements and collaboration at an appropriate scale. The establishment of a strong collaborative processes appears as the only way to guarantee the essential groundwater supply to coastal GDEs and their sustainability (Boulton, 2005). The management of coastal GDEs must take into account its hydrological basin as well as its territorial water management unit and all territorial units important for its management, *i.e.* tourist unit, geographical unit, air of influence of neighboring cities or migratory bird management (Mermet and Treyer, 2001)...

Conclusion

Nowadays, coastal Mediterranean regions suffer from an over-development of anthropogenic activities which strongly impact the groundwater resources and depending coastal GDEs. Although some Mediterranean regions have included the protection of GDEs in their water management policies, the implementation of an appropriate intergraded and collaborative management is often lacking and coastal GDEs do not benefit from a particular status due to their complexity.

The preservation of coastal GDEs is subject to the stability over time of fresh water supplies (ground and surface water) in sufficient quantity and quality. However, the determination of the qualitative and quantitative needs of coastal GDEs is difficult to evaluate and each coastal GDE is a unique case. Particular attention should therefore be paid to the characterization of environmental and ecological water requirements. The hydrogeological knowledge about the management and behavior of coastal aquifers and GDEs must be strengthened. Hydrogeology must be considered as an integral component of the coastal GDEs and not a sub-discipline of hydrology, as is too often the case at present. The inventory and characterization of coastal GDEs must be improved through in-depth systemic approaches. To this end, the coupling of hydrogeochemical and geophysical techniques, which are inexpensive, seem to constitute a relevant strategy. These investigations must be supplemented by the identification and evolution of the sources of contamination present in the catchment areas. In order to better understand the role of groundwater as a vector of pollution, particular attention should be paid to the identification of the main groundwater discharge areas and the assessment of contaminant flows and loads. The systematic mapping of groundwater vulnerability in the coastal areas must be promoted, using methods accounting for both the intrinsic and specific vulnerability of groundwater. This kind of data must help to develop land-uses and human activities according to the groundwater vulnerability. Finally, in the case of effective degradation processes, restoration plans should be considered. A reflection must be carried out for the definition of relevant indicators of the ecological coastal GDEs status. For these environments subject to high variabilities, particularly in terms of salinity, there is a necessity of developing sensitive indicators for monitoring ecological status. Biological indicators seem to be helpful but needs to be further and widely developed.

From a qualitative point of view, the estimation of groundwater withdrawals is often very approximate because of the poor knowledge of the extraction points. It seems essential to carry out an exhaustive inventory of wells and boreholes in the coastal GDE watershed. The implementation of retroactive measures for reporting private wells would also allow a better knowledge of the existing structures, which are currently not recorded. Regularly monitored water quotas for private individuals could also be helpful for the qualitative management of the resource.

At present, the lack of an appropriate definition for coastal GDEs is a huge problem. Lack of discussion and consensus between lawyers and scientists does not facilitate the establishment of management strategies. To be efficient, this definition needs to be the result of a joint reflection between several disciplines. As showed in this synthesis, the transdisciplinary approach between hydrogeology, hydrology, social sciences and law is essential to fully understand the socio-economic and environmental complexity of coastal GDEs. The inventory of coastal GDEs characteristics could help to establish a complete and relevant definition of coastal GDEs. In addition to involve several discipline, thoughts about coastal GDEs definition need to be based on the mobilization of scientist, lawyers but also water users and stakeholders. Information, appropriation and collaboration are clearly strategic, interdependent points to be developed. Local water users and managers must feel concerned by the problems related to coastal GDEs to build appropriate and sustainable management plans. Without this process, all possible efforts can be taken, but their chances of achieving successful results will remain low. The creation of permanent mechanisms such as water user groups or groundwater forums could be useful. These moments of exchange and discussion would also allow managers and decision-makers to better understand the role and benefits of coastal GDEs. Indeed, evaluation of the ecosystem services is essential for valuing the coastal GDEs and decision makers at many levels are unaware of the connection between wetland condition and the provision of wetland services and consequent benefits for people.

All water resources in the coastal areas should be managed collectively and strategically, in order to maximize use efficiency, reduce water use conflicts and avoid over-exploitation. In other words, the management strategy must consider the lagoon water body, the surrounding wetland and groundwater as an inseparable set of communicating vessels whose nature of exchanges is subject to temporal and spatial variations. In the global context of unprecedented anthropogenic pressures, hydro-food crises and climate change, the consideration given to coastal GDEs represents a key issue for the socio-economic and environmental sustainable development of many coastal Mediterranean areas. Integrated water management strategies that consider environmental needs on an equal footing with socio-economic constraints within the coastal hydrosystem need to be improved. The ICZM is the management strategy that most considers water resources in the coastal zone and refers to coastal aquifers as such and specifies a monitoring requirement. However, despite the growing consideration for coastal aquifers, there are still gaps. It is important to continue to raise public awareness of coastal aquifers at the regional level and to integrate their specificities into coastal zone management strategies and plans. Collaboration between states or countries, sharing of knowledge and technology facilitated by the creation of exchange material could also contribute to improving the integration of coastal aquifers into local guidelines and policies.

These practical suggestions could help for improving the management of coastal aquifers and coastal GDEs. In this way, groundwater and coastal water GDEs could really benefit from the optimal environmental conditions required to ensure their sustainability.

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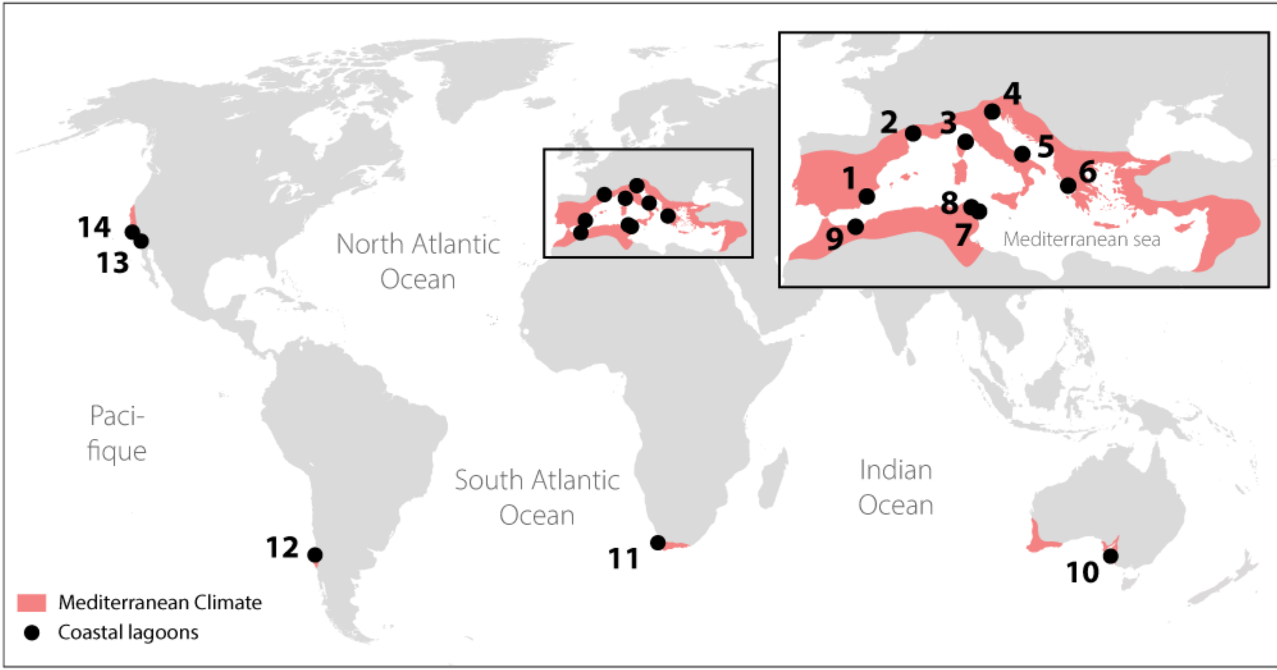


1. Mar Menor
2. Thau
3. Biguglia
4. Venecia
5. Varano
6. Messolonghi
7. Korba
8. Bizerte
9. Nador
10. Coorong
11. Langebann
12. El Yali
13. San Diego
14. Malibu

Fig. 1: Coastal regions under Mediterranean climate and location of the 14 coastal lagoons exposed in Table 1.

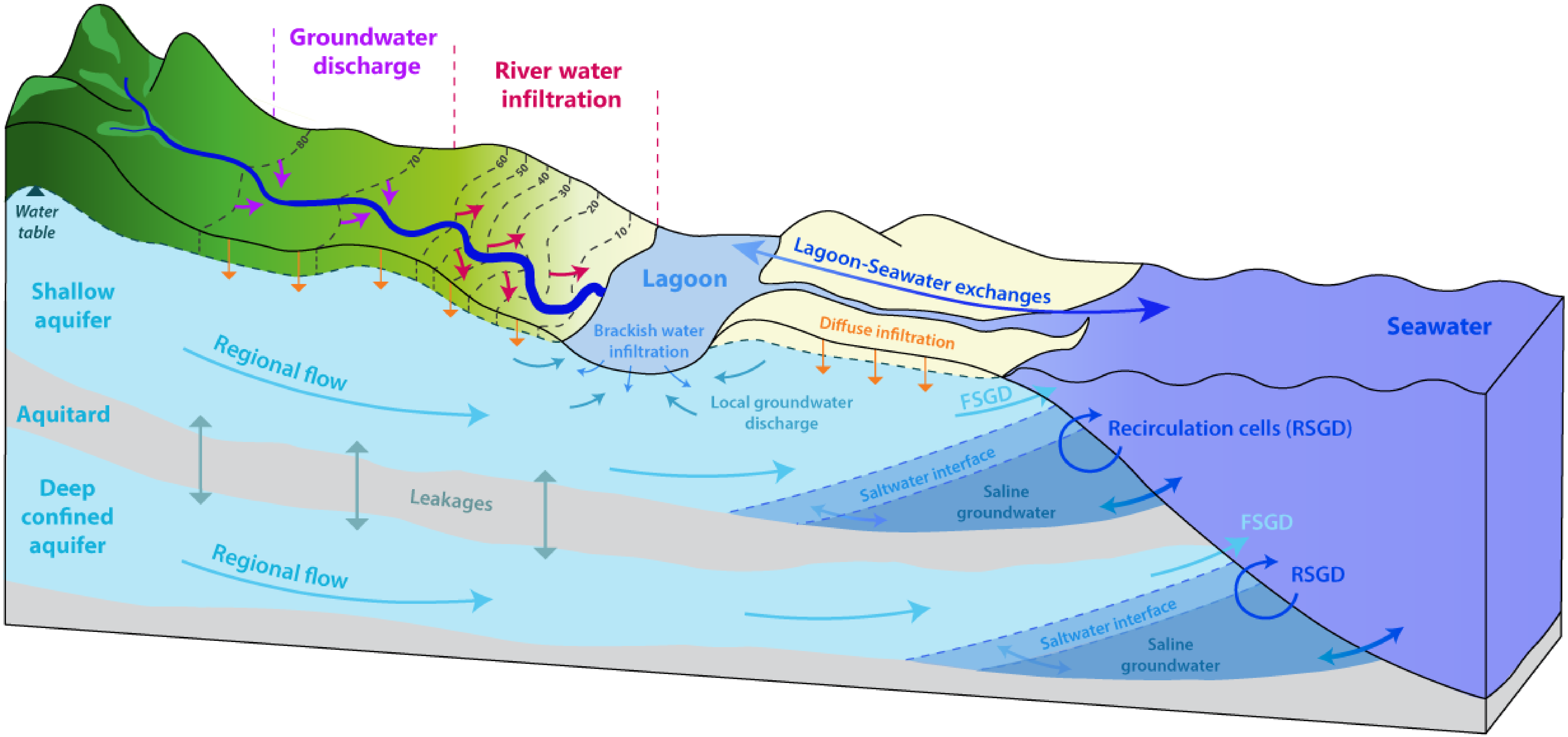


Fig. 2: Conceptual diagram of the hydrogeological behavior of coastal hydrosystems including a coastal GDE. On a large scale, the discharge of groundwater is composed by two processes: i) the discharge of fresh groundwater toward the sea (fresh submarine groundwater discharge, FSGD) and ii) the discharge of saline groundwater, i. e. the discharge of a mixture of fresh water and seawater after recirculation through the transition zone (recirculated submarine groundwater discharge, RSGD).

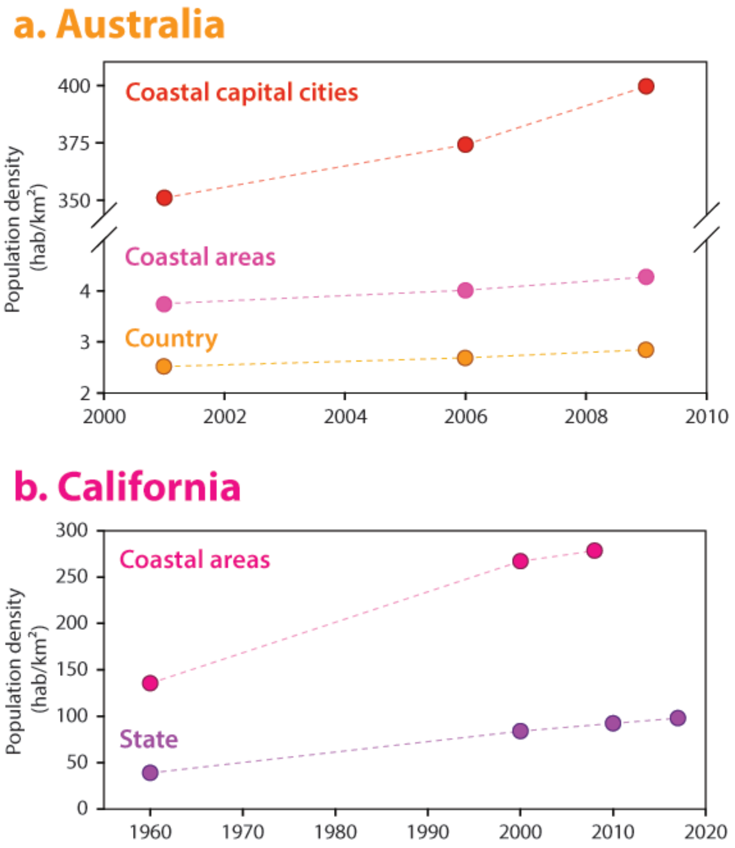


Fig. 3: Population density trends in Australia (a) and California (b).

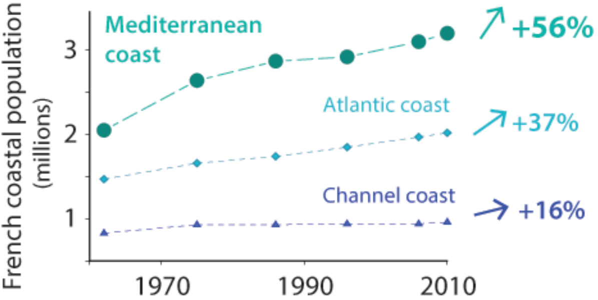


Fig. 4: Demographic trends on the three French coasts.

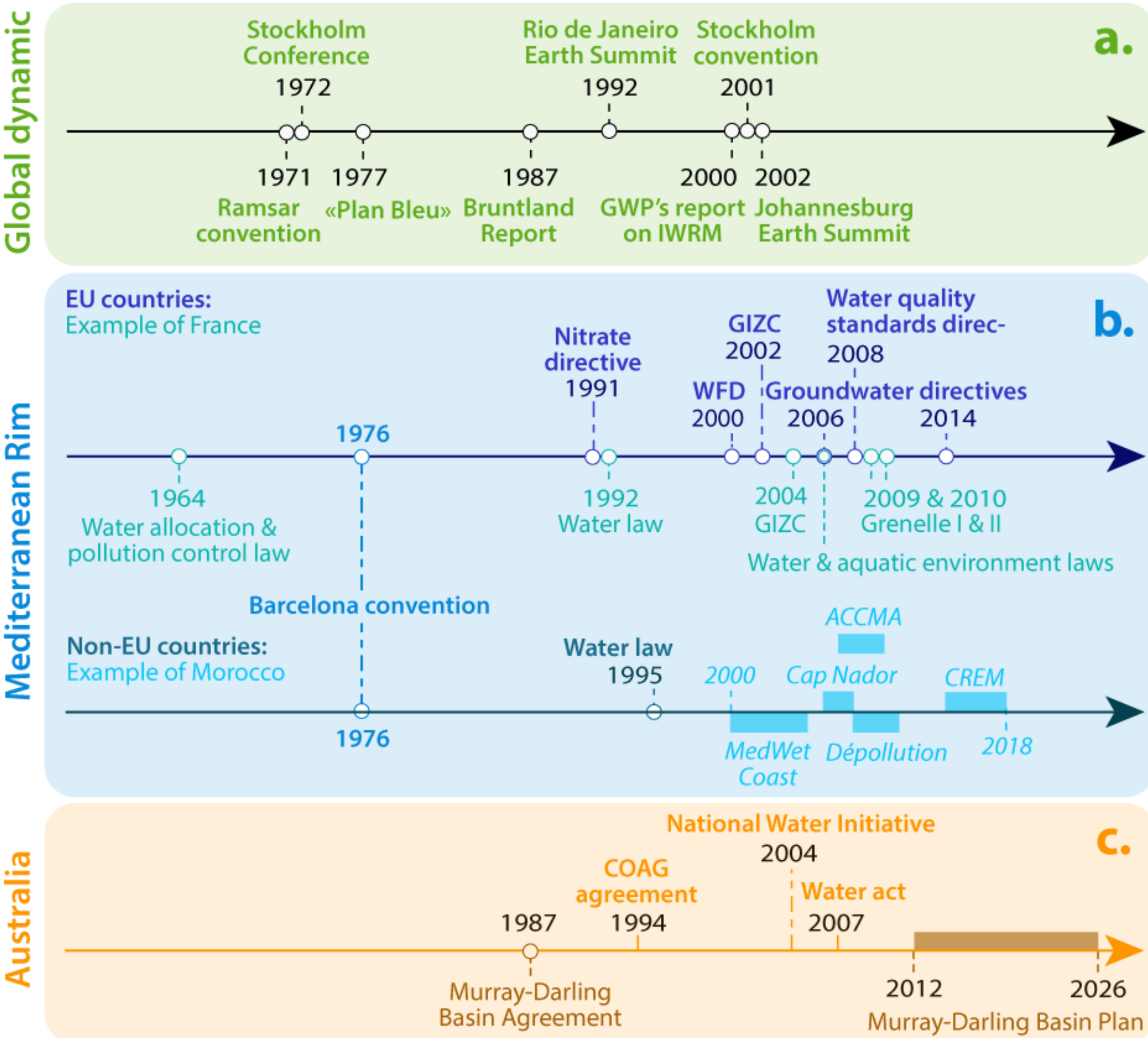


Fig. 5: Main world events that have guided the establishment of sustainable water resources management (a) and their translation into local laws and measures in the case of France (EU country), Morocco (Non-EU country) (b) and Australia (c).

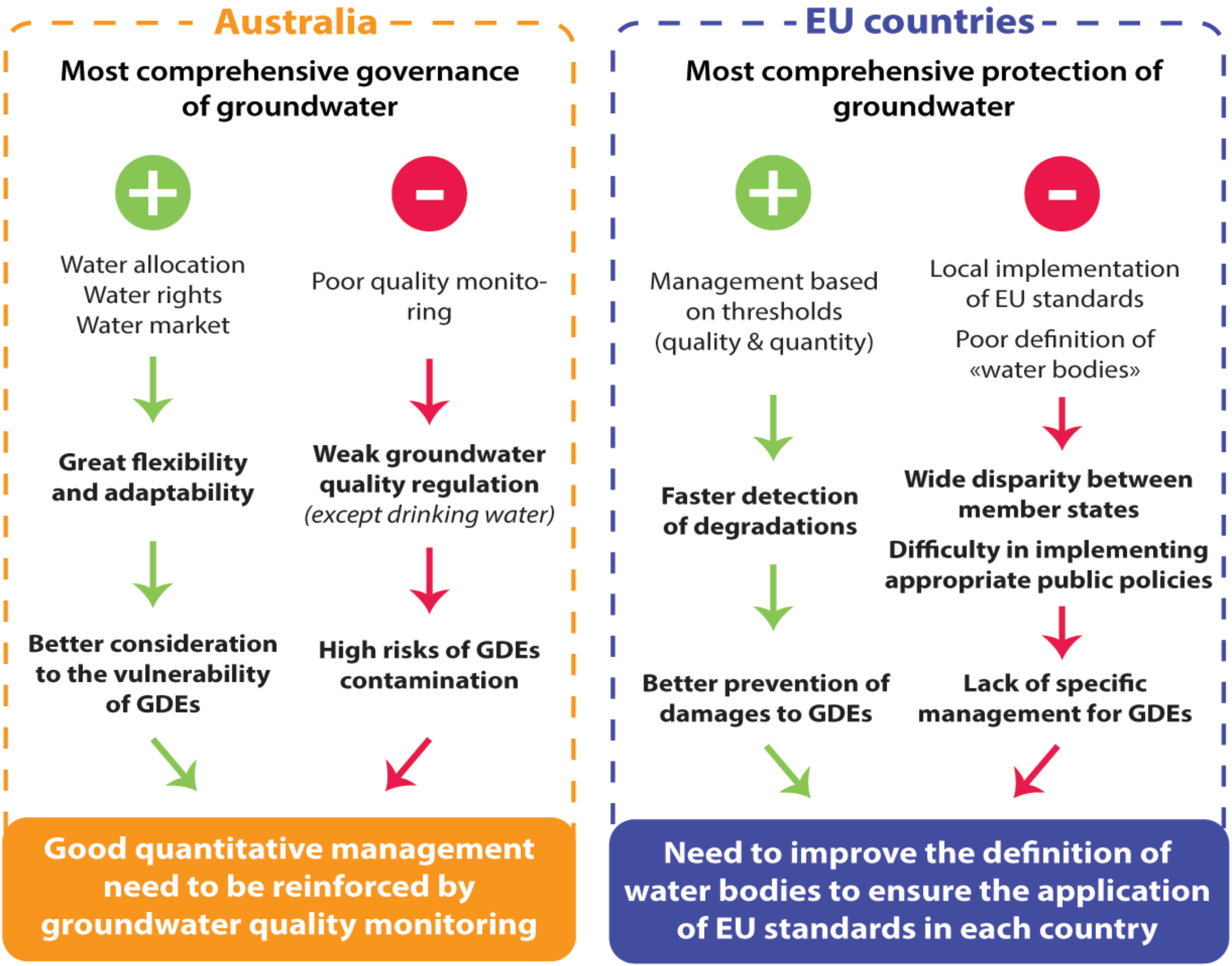


Fig. 6: Comparison of the strengths and weaknesses of management strategies in Australia and Europe.

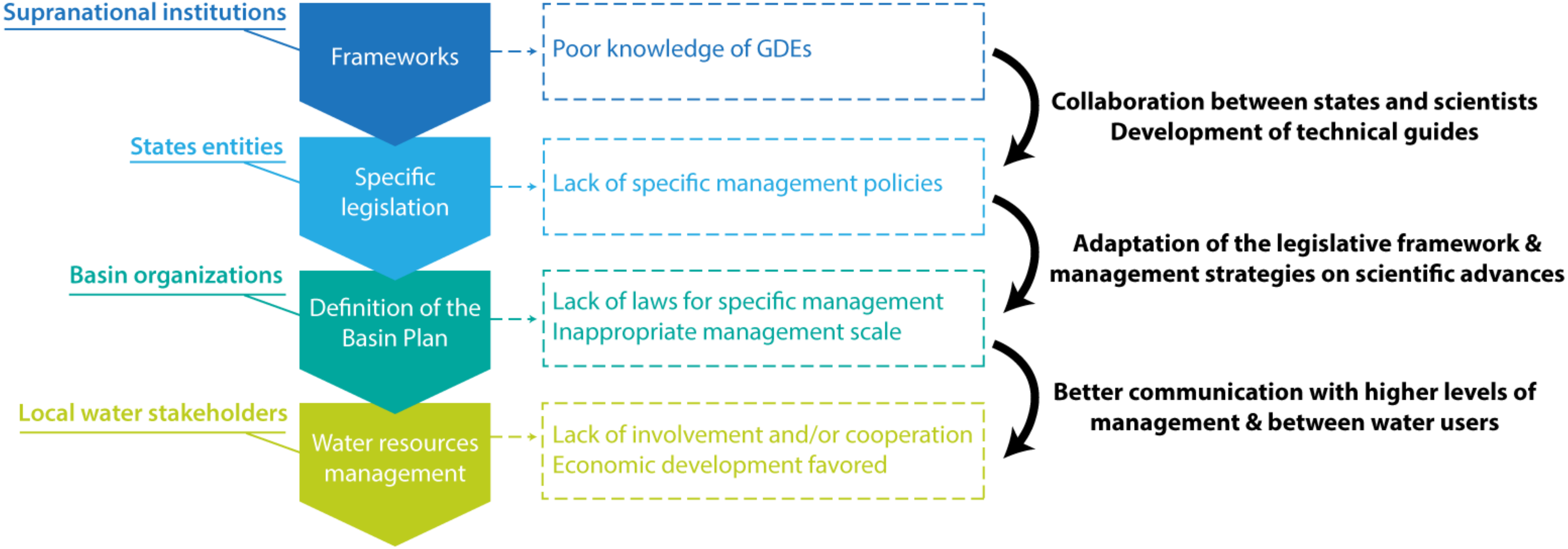
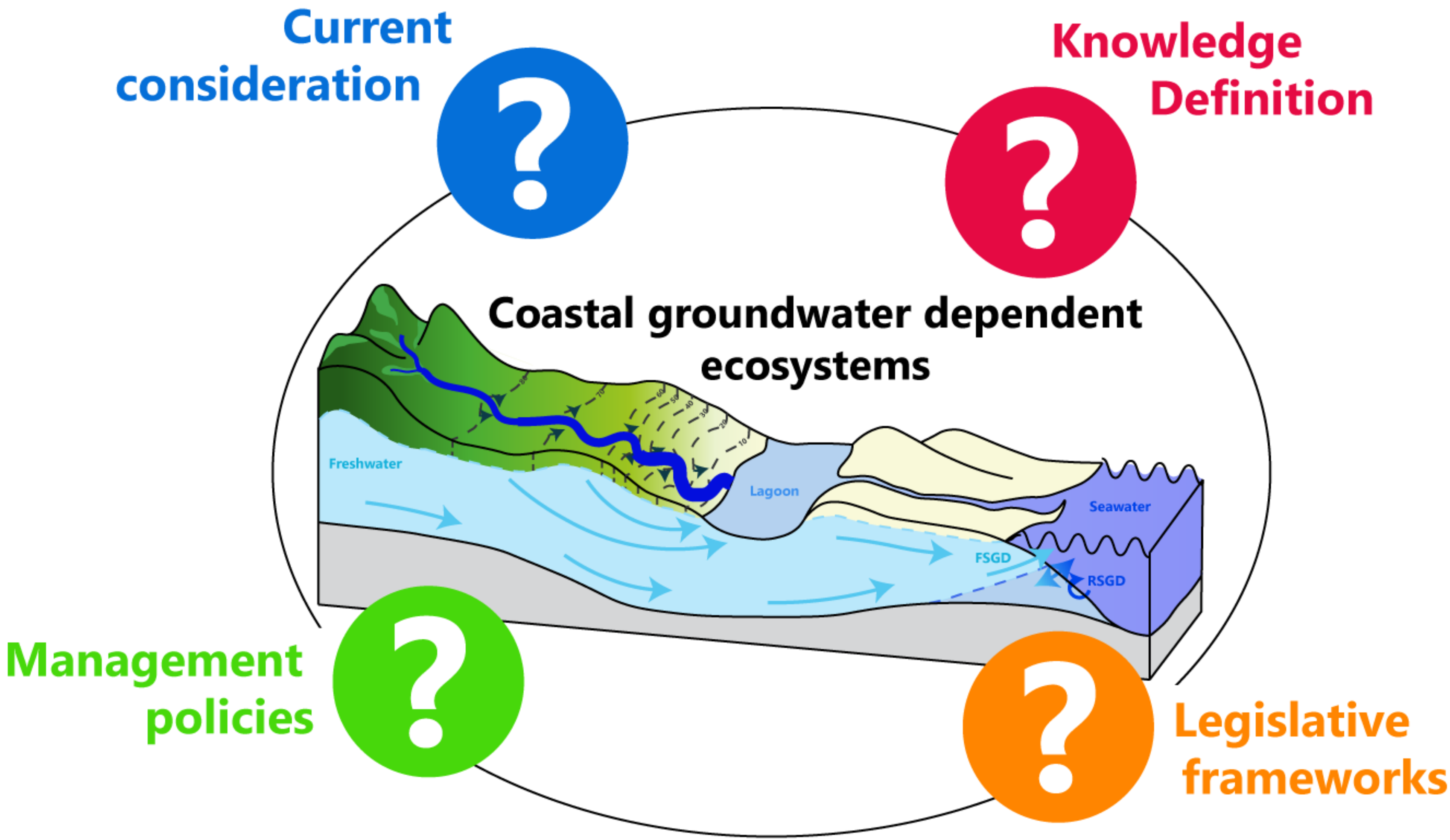


Fig. 7: Conceptual diagram showing the institutions and their roles in water resources management, highlighting major gaps and the points to be improved between two hierarchical levels.

	Lagoons	Countries	Characteristics				Conservation and protection status	Hydrosystem behavior and groundwater dependence		References
			Surface (km²)	Mean depth (m)	Main aquifer formation(s)	Hydrological watershed (km²)		Strongly suspected	Demonstrated	
1	**Mar menor**	Spain, South-East	135	4.5	5 aquifers: Detrital deposits Sandston Limestone Sandy limestone and conglomerate Marble	1200	Ramsar site Special bird habitat Regional park Site of Community importance Specially protected area of Mediterranean importance		Studied and relatively well known	De Pascalis et al., 2012; Baudron et al., 2014; Velasco et al., 2018; Alcolea et al., 2019
2	**Thau**	France, South-Est	75	4	Karstified limestone	280	Special bird habitats Natura 2000 Water Framework Directive site		Studied but lack of data to understand the global behavior	Tournoud et al., 2006; Fleury et al., 2007; Stieglitz et al., 2013; Loiseau et al., 2014; La Jeunesse et al., 2016
3	**Biguglia**	France, Corsica island	14	1.2	Detrital deposits	182	Ramsar site Nature Reserve Special bird habitats Natura 2000 Water Framework Directive site		Studied and relatively well known	Lafabrie et al., 2013; Erostate et al., 2018; Jaunat et al., 2018; Erostate et al., 2019; Leruste et al., 2019
4	**Venice**	Italy, North-East	550	1.5	Detrital deposits	1800	Ramsar site Natura 2000 Special bird habitat		Largely studied and relatively well known	Ravera, 2000; Ferrarin et al., 2008; Rapaglia et al., 2010; Da Lio et al., 2013; Mayer et al., 2014
5	**Varano**	Italy, South-East	65	3.5	2 main aquifers: Detrital deposits	300		Under-documented		Ferrarin et al., 2010; Roselli et al., 2013; Fabbrocini et al., 2017
6	**Messolonghi central lagoon**	Greece, North-West	80	0.8	2 main aquifers: Limestone and breccia Detrital deposits	1979	Ramsar site National Park Important Bird Area Natura 2000	Under-documented		Alexakis, 2011; Karageorgis et al., 2012; Stamatis et al., 2013
7	**Korba**	Tunisia, plain of Cap Bon	3.1	1	Detrital deposits	27	Ramsar site Important Bird Area	Under-documented		Kouzana et al., 2010; Zghibi et al., 2013; Slama and Bouhlila, 2017

8	**Bizerta**	Tunisia	128	8	Detrital deposits	380	Ramsar site UNESCO-MAB Reserve	Under-documented		Bouzourra et al., 2015; PNUE-PAM, UNESCO-PHI, 2017
9	**Nador**	Marocco, Nord-Est	115	5	2 mains aquifers: Detrital deposits	?	Ramsar site Nature Reserve Site of biological and ecological interest	Groundwater contribution known but under-studied		Maanan et al., 2015; Mohamed et al., 2017; Aknaf et al., 2018
10	**Coorong**	Australia, South-East	140	1.8	Limestone Sands	6	Ramsar site National Park		Studied and well known	Haese et al., 2008; Richardson et al., 2011; Leterme et al., 2015
11	**Langebaan**	South Africa	40	3	Detrital deposits and calcrete	?	Ramsar site National Parks	Under-documented		Flemming, 1977
12	**El Yali**	Chile	115	0.5	Detrital deposits	?	Ramsar site National reserve	Groundwater contribution known but under-documented		Dussaillant et al., 2009; Vidal-Abarca et al., 2011
13	**San Diego**	California (U.S.A)	42	5	Detrital deposits	146	National Wildlife Refuge	Under-documented		Delgadillo-Hinojosa et al., 2008
14	**Malibu**	California (U.S.A)	0.05	?	Detrital deposits	280		Groundwater contribution known but under-documented		Dimova et al., 2017; Hoover et al., 2017

Mediterranean basin	1970	2000	2010	2025
Whole population (millions)	276	412	466	529
Coastal population (millions)	95	143	-	174
Urbanisation rate (%)	54	-	66	-


Current consideration
Knowledge Definition
Coastal groundwater dependent ecosystems
Freshwater
Lagoon
Seawater
FSGD
RSGD
Management policies
Legislative frameworks