Changes in Mesopotamian Wetlands: Investigations Using Diverse Remote Sensing Datasets

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Abstract
Early civilizations have inhabited areas with stable water resources that supported living needs and activities. The Mesopotamian marshes have experienced dramatic changes during the past five decades. The aim of this study is to observe, analyse and report the extent of changes in these marshes from 1972 to 2020. Data from various sources were acquired through Google Earth Engine (GEE) including climate variables, land cover, surface reflectance, and surface water occurrence collections. Additionally, streamflow data was also analysed. Methods were based on diagnostic analysis to monitor and evaluate the causes and results of the total environmental dynamism. Results show a clear wetlands dynamism over time, a decrease of incoming flow to the region due to the damming of upstream tributaries, and a significant loss in marshlands extent, even though no significant long-term change was observed in lumped rainfall from 1982, and even during periods where no meteorological drought had been recorded. Human interventions have disturbed the ecosystems, which is evident when studying water occurrence changes. These show that the diversion of rivers and the building of a new drainage system caused the migration and spatiotemporal changes of marshlands. Nonetheless, restoration plans (after 2003) and strong wet conditions (period 2018–2020) have helped to recover the ecosystems, these have not led the marshlands to regain their former extent. Further studies should pay more attention to the drainage network within the study area as well as the neighbouring regions and their impact on the streamflow that feeds the marshes.

Keywords Ecosystem disturbance · Remote sensing · Wetlands · Marshes · Ancient civilizations

Introduction
Wetland ecosystems have played one of the most important roles in first human civilizations by offering rich‑resourced habitats and providing stable/durable sustenance‑life during all seasons of the year (Bobbink et al. 2006). These ecosystems help to develop, grow, and sustain the natural environment (Mitsch and Gosselink 1993) and also reduce flood impacts, enhance water quality by absorbing pollutants, and serve as important faunal habitats among many other environmental, recreational, and economic advantages (Al‑Nasrawi 2018). In fact, wetlands were not only a recreation place for early communities, but they were also the essential key for early civilization development (Maltby and Acreman 2011). People of early civilizations were distributed around the riverine systems and specifically concentrated around the lower reaches where wetlands occurred (Maisels 2001). These prosperous environments resulted from the stability of water and food resources, combined with diverse habitats that could not be found anywhere else (Trigger 1993). Such circumstances facilitated the transformation and development of their communities, technology and ultimately their entire cultures. Innovations and advancements included the implementation of irrigation systems and the invention of the wheel, plow and cuneiform writing (Faiella 2006).
One of the major human settlements sustained by wetland ecosystems gave birth to the Mesopotamian civilization, one of the most ancient civilizations that took place on the banks of the Tigris and Euphrates Rivers ~6000 years ago (Potts 1997). Sumerians were the first to inhabit the wetlands and marshlands that are locally known as Mesopotamian marshes (Banister 1980). These are limited to south-eastern Iraq and extend across the western Iranian border nowadays (Al-Zaidy et al. 2019; Sissakian et al. 2020).

Sumerians were succeeded by Babylonians and in contemporary history gave origin to the Marsh Arabs, which are believed to be their descendants and dwell in the Mesopotamian marshes nowadays. Marsh Arab population flourished to reach between 300,000 and 500,000 inhabitants (Richardson and Hussain 2006). Their main food sources and economy rely on agriculture (rice, wheat, barley crops), fishing, and water buffalo farming, and therefore, strongly depend on the marsh ecosystems.

However, the Mesopotamian wetlands have experienced a rapid environmental alteration (dam construction upstream and the intensive drainage around these marshlands) over the past ~50 years (especially in the late 1980 s and early 1990 s). The speed and scale of these changes were massive and led to turn around 97% of the Central marshland, around 94% of the Hammar marshland, and more than 70% of the Hawizeh marshes to bare soil, in some cases covered with salt crusts (Partow 2001).

Some projects involved in the degradation of the marshes have been carried out upstream the marshes for development purposes (dams for hydropower generation and irrigation in the Tigris and Euphrates rivers; Adamo et al. 2020), while others, such as the drainage network built around the marshes, have been implemented as a retaliation measure against the uprising of Marsh Arab communities that inhabited the region and other Shia in 1991 (Ruiz 2010). Draining the marshes affected the local economy, mostly consisting of agriculture and fishing, and forced remaining communities to abandon the region until the end of Saddam’s regime (Richardson and Hussain 2006). These measures led to a reduction of fully functional marshes to about 10% of their original extent in 2004 and caused a strong reduction of the Marsh Arab population dwelling in the marshes to less than 10,000 (Richardson et al. 2005).

Several studies have investigated the changes in Mesopotamian marshes. While some of these have implied the use of limited scenes to study the changes in vegetation and surface water (Jabbar et al. 2010; Saleh 2012; Zhang and Abed 2013; Hashim et al. 2019), others have used periodic images to monitor vegetation but limited temporal extents not covering the periods of large changes (Al-Handal and Hu 2015; Albarakat and Lakshmi 2019). Similarly, some studies have monitored the vegetation dynamics through multidecadal analysis but have been constrained to the low spatial resolution of some of the data sources utilised and their relation to climate variables has been investigated through limited point observations (Albarakat et al. 2018). Therefore, a necessity to investigate the different changes at relatively high spatiotemporal resolutions and evaluating their dependence to spatial land cover changes and regional hydroclimatic dynamics is still relevant. In this regard, historical hydroclimatic changes and their affectation to the Mesopotamian marshes has been recently studied (Al-Quraishi and Kaplan 2021), shedding light on the reduction of marshland extent primarily caused by upstream hydraulic modifications. However, there is still a gap to better understand and quantify regional and intraregional spatial changes and their relationship with hydroclimatic and land cover changes due to additional hydraulic works carried out to drain the region and subsequent restoration plans implemented to recover the marshes (Jabbar et al. 2010).

**Current stressors and efforts in the region**

In the Mesopotamian region, droughts are the most devastating manifestation of climate hazards, with low precipitation and elevated temperatures having the strongest impact on the ecosystems (Senapati et al. 2019). Drought events have been a notable issue throughout the 6000 years of history in the region (Wilkinson et al. 1994; Wilkinson 1997; Kerr 1998; Kuzucuoğlu 2007), however, the rapidity or cyclical nature of the drought patterns changed after the industrial revolution compared to historical patterns (Cookson et al. 2019). Additionally, human interventions such as water discharge regulation through damming, and the implementation of artificial drainage systems increase the pressures on the wetland ecosystems by disrupting the natural water inflow patterns and draining the region (Sama et al. 2012; Al-Nasrawi et al. 2016).

The water issue has always been a bottleneck that restricted the development of the Middle East (Allan 2012). The Mesopotamian marshlands in southern Iraq and across the Iranian border (Fig. 1) are the largest wetlands and among the most populous in the region, but they have been increasingly faced with water shortage and crises throughout the last decade (Madani 2014).

UNESCO has recently listed all the southern Iraq wetlands (Supplementary 1) as a “World Heritage Site” known as Mesopotamia or the Garden of Eden (UNESCO-World Heritage Centre 2016). On the Iranian side, of the 44 internationally known wetlands (Najaf and Vatanfada 2011), 24 of them are on the Ramsar Convention wetland list (Davidson et al., 2019; Hamman et al. 2019). Many of these wetlands in both countries and all over the Middle East are seriously damaged due to the negative effects of climate change and various human activities (Naddafi et al.
Some of them have even been destroyed, while historically they served as popular safe habitats for many endangered species, including migratory birds (Abed 2007).

Conservation activities that have been carried out so far have not prevented the alteration and destruction of wetlands or reduced the damage due to their dispersal and inefficiencies (Hamzeh et al. 2017). Iraq and Iran made tackling their water crises national priorities but the lack of technical know-how and funding, and the difficulty of cooperating regionally and internationally, has prevented significant improvements. Therefore, it is crucial to put serious efforts into detecting the emerging water problems before they become too costly to resolve. However, the current level of knowledge about precise monitoring and assessment of water variation and land cover changes is still low and not good enough at regional and basin scales. In this regard, the use of Satellite Earth Observation (SEO) applications within innovative geo-spatial analysis is a key tool and unique information source to support the environmental community in various application domains, including wetlands conservation and management. More specifically, existing and future SEO technology can play a key role in obtaining suitable information to support the mapping and inventory of wetlands as a basis for management-oriented assessment and monitoring.

While wetlands have traditionally been studied through field surveys (Olsen et al. 2019), new alternatives have emerged. One of these is remote sensing, which has become increasingly popular due to their potential for monitoring different processes taking place on the Earth surface at different spatio-temporal scales (Chen and Wang 2018; Asokan and Anitha 2019; Karthikeyan et al. 2020). Additionally, different remote sensing applications have been developed to quantify changes in ecosystems (Bergen and Dronova 2007; Pastick et al. 2019). Among these, the detection and monitoring of wetlands has also been achieved through the combined evaluation of water and vegetation which can be detected through different wavelength spectrum ranges (Ozesmi and Bauer 2002; Kaplan and Avdan 2018). However, remote sensing can also allow climate monitoring through microwave or infrared sensors (Funk et al. 2015; Huffman et al. 2019).
Given the socio-environmental importance of Mesopotamian marshes to preserve ecosystem functioning and to support historic human settlements in the region, this study aims to investigate temporal changes in the ancient Mesopotamian wetlands in southern Iraq since the 1970s, using satellite imagery. Additionally, it seeks to understand the causes behind these changes and their quantification, which include land cover and hydroclimatic evaluation. The results of this study will help to describe environmental changes in the area and will ultimately provide a basis to address them.

Materials and Methods

In this study, satellite and water discharge data was used to detect spatio-temporal changes in the land cover dynamics of the Mesopotamian region during the last few decades. The Google Earth Engine (GEE) platform (Supplementary 2) was used to access and pre-process the satellite data. GEE provides the tools to access, process and analyse multi-peta-byte collections of satellite imagery on the cloud, reducing the computational requirements for users and the time required for data acquisition and processing (Gorelick et al. 2017). This facilitates large scale studies and the monitoring of land surface processes occurring at different time scales.

Study Area

The ancient Mesopotamian marshlands (Garden of Eden) lie within the alluvial plain of the Tigris and Euphrates Rivers, which are in southern Iraq and the bordering portion of western Iran (Kohl and Lyonnet 1826; Frankfort 1950; Abdulnaby et al. 2020). These two riverine sources supply the freshwater marshlands that have an average depth of 2 m. These marshlands cover an area of about 20,000 km² with central coordinates of (31°02’04.3″N 47°04’59.2″E) (Abusch 2020). The marshes represent a natural phenomenon of low-lying land, which is characterised by seasonal fluctuations in water levels. Therefore, delineation of the study area boundaries is difficult. In this study, the boundary of the study area was defined “naturally” by using the 5 m elevation-contour line, which has included 15,403.65 km² of the marshlands. This contour includes most of the permanent Mesopotamian marshes. The two ancient rivers (Tigris and Euphrates) divide these marshlands into three main permanent sections, surrounded by seasonal subsections and lakes. Hawizeh marshes are located to the east of the Tigris, and the Hammar marshes occur just south of Euphrates, whereas the Central section (including Al-Chibayish marshes) lies between the Tigris and Euphrates Rivers. Together they make up a large and well-known marsh ecosystem (Pollock and Susan 1999). The study area is covered with fine to very-fine clay and silty soils (Jotheri et al. 2018). The climate of the Tigris/Euphrates marshes is semi-arid with a warm-dry summer (average ~ 43°C with about 0 mm year⁻¹ of rain) and a cold-wet winter (average ~ 11°C, plus ~ 100–170 mm year⁻¹ of rain) (Shubbar et al. 2017). These unique marshlands have been severely drained and subject to eutrophication since the 1990s because of human activities including the war during the 1980s. However, after 2003, restoration plans were formally legislated, and laws were enacted to control water management (Al-Ansari et al. 2012).

Datasets Used

Diverse sources of data were used in this study. Monthly rainfall, lumped at the study area scale, was obtained from the Famine Early Warning Systems Network (FEWS NET) Land Data Assimilation System (FLDAS; McNally et al. 2017), which aggregates and processes rainfall data from the Climate Hazards Group InfraRed Precipitation with Station (CHIRPS) data. This was used to evaluate water input changes in the study area that might explain the wetland dynamic in time. Monthly accumulated daily rainfall data from CHIRPS, forcing FLDAS incoming shortwave radiation and monthly ERA-5 temperatures, surface pressure and wind components were also utilised to estimate the reference evapotranspiration and standardised drought indices, which can shed light on the reasons behind wetlands extent change. Since no river gauge station could be found in the study region, mean annual streamflow obtained from upstream locations was used (Sahle 2010). These locations correspond to the Euphrates river downstream of Hindiya barrage (E03) from 1930 to 2000 and the Tigris river downstream of Kut barrage (T20) from 1930 to 2005.

The Land Cover Type 1 from the Annual International Geosphere-Biosphere Programme (IGBP) contained in the MODIS Land Cover dataset (MCD12Q1 version 6) was used to evaluate regional changes in land cover from 2001 to 2019. This product is derived from a supervised classification on MODIS surface reflectance data and post-processing using ancillary data to improve its quality (Sulla-Menashe and Friedl 2018). Since this collection is based on MODIS data, its time period of application is limited to the operation period of the sensors.

The Joint Research Centre (JRC) Global Surface Water Mapping Layers, v1.0 dataset, was also used to provide spatio-temporal information about surface water in the study area. This allowed splitting the water extent into classes depending on their occurrence. This analysis was carried out because marshes are ecosystems where vegetation and their extent depend on the water occurrence, which can be permanent, seasonal, or sporadic. The JRC Water Mapping dataset provides maps for water detection using Landsat 5, 7 and 8 which were collected from 1984 to 2015 (Pekel et al. 2016) and has been considered as one...
of the state-of-the-art products for water detection. This is based on a classification performed on each pixel of the Landsat collection using an expert system. The expert system is a non-parametric classifier that accounts for the uncertainty in the data, can use multiple data sources, and expertise can contribute to the classification process (Pekel et al. 2016). Furthermore, a water transition map was derived from the JRC dataset to characterize types of changes caused by drought events and human intervention. Lastly, surface water trends were derived from the Monthly Water History v1.2 dataset from the JRC.

Additionally, tier 1 images from the Landsat 1, 5, 7 and 8 collections were cleaned from clouds and cloud shadows using the CFMask algorithm (Foga et al. 2017) and used for the generation of normalised difference vegetation index (NDVI) rasters from 1972 to 2020. The temporal distribution of Landsat images is shown in Fig. 2. NDVI is associated with vegetation growth, and their rasters were used in time series analysis to evaluate vegetational changes in marshes. Landsat 1 tier 2 (Landsat 1 tier 1 data are not available for the study area) also was used to extract the NDVI in the 1972 period. During the 1970’s, there were several scenes available for the study area. However, all these scenes are heavily contaminated with clouds or shifted from their actual locations. Therefore, there were only three images that entirely cover the study area, which were cloud-free and accurately positioned and were acquired between 01-08-1972 and 02-08-1972.

**Drought evaluation**

Drought is considered one of the most expensive and deadliest natural hazards leading to rainfall deficits (Zargar et al. 2011; Ault 2020). Since droughts can be drivers for regional water deficits, these can be responsible for wetlands changes (Stirling et al. 2020). Therefore, the Standardised Precipitation Index (SPI) and the Standardised Precipitation Evapotranspiration Index (SPEI) were used as a proxy for drought evaluation. Both were calculated using the methodology described in Guenang and Kanga (2014) through a standardisation of the data using the gamma distribution:

\[
g(x) = \frac{1}{\beta \Gamma(\alpha)} x^{\alpha - 1} e^{-x/\beta} \quad (1)
\]

in which the shape and scale parameters are calculated as:

\[
\alpha = \frac{1}{4A} \left( 1 + \sqrt{1 + \frac{4A}{3}} \right) \quad (2)
\]

\[
\beta = \frac{x}{\alpha} \quad (3)
\]

being \( A \):

\[
A = \ln \left( \frac{\bar{x}}{n} \right) - \frac{1}{n} \sum \ln(x) \quad (4)
\]
where \( x \) corresponds to rainfall or rainfall minus evapotranspiration observations for SPI and SPEI, respectively. Subsequently, we used the incomplete gamma function (Eq. 5) by which the cumulative probability \( G(x) \) of an observed quantity of rainfall/rainfall-evapotranspiration can be estimated:

\[
G(x) = \frac{1}{\Gamma(a)} \int_0^x t^{a-1} e^{-t} dt
\]  

(5)

where \( t \) is \( x/\beta \). Since the gamma distribution is undefined for \( x = 0 \), positive cumulative probabilities assume the following formulae:

\[
H(x) = q + (1 - q)G(x)
\]  

(6)

being \( q \) equal to \( P(x = 0) \). Finally, the cumulative probabilities are transformed to the standard normal of \( \mu = 0 \) and \( \sigma = 1 \) using Eq. 7:

\[
Pr(X \leq x) = F(x) = \frac{1}{2} \left[ 1 + \text{erf} \left( \frac{x - \mu}{\sigma \sqrt{2}} \right) \right]
\]  

(7)

For SPEI, where the analysis is calculated on the time series of the difference between rainfall and evapotranspiration, an offset was added to the time series in order to preclude negative values, which makes the standardisation undefined.

Evapotranspiration was calculated combining the FLDAS and ERA-5 datasets through the FAO Penman-Monteith equation (Allen et al. 1998):

\[
ET_r = \frac{0.408\Delta(R_n - G) + \gamma C_d C_n (e_i - e_a)}{\Delta + \gamma (1 + C_d \mu_2)}
\]  

(8)

where \( \Delta \) is the saturation vapor pressure-temperature slope, \( R_n \) the net radiation, \( G \) the soil heat flux, \( \gamma \) the psychrometric constant, \( T \) the mean daily temperature, \( \mu_2 \) the mean wind speed at 2 m, \( e_i \) the saturation vapor pressure, \( e_a \) the actual pressure vapor, and \( C_d \) and \( C_n \) are constants equivalent to 0.34 and 900, respectively, being \( ET_r \) the reference evapotranspiration.

The SPI and SPEI calculated were averaged to the study region and evaluated using 1- (monthly), 3- (quarterly), 6- (semi-annually) and 12-month (annually) scales, trying to represent drought in the meteorological (1-month), agricultural (3- and 6-months), and hydrological (12-months) dimensions (Tirivarombo et al. 2018). This allowed us to obtain drought/wet severity classes based on Table 1.

### Surface Water Analysis

The Monthly Water History dataset from the JRC was processed to obtain maps of annual water occurrence. These were estimated at each pixel by summing the times that water occurred in the year and dividing it by the times that the same pixel occurred in the year, leading to values between 0 and 1 corresponding to the occurrence probability. Empty records in the annual occurrence maps were linearly interpolated, especially for 1988, 1989, 1990, 1996 and 1997, assuming a smooth transition between years. The interpolated maps were mosaicked with the annual occurrence maps to fill empty pixels.

Additionally, surface water occurrence rasters were split into three different pixel classes: (i) episodic water pixels, which presented a water occurrence lower than 0.25, meaning that water occurred in one fourth of the images of the year; (ii) seasonal water pixels, which present a water occurrence between 0.25 and 0.75, meaning that water is present seasonally in more than one quarter and less than three quarters of the annual images; and (iii) permanent water pixels with an occurrence of water greater than 0.75, implying that surface water is present in more than three quarters of the images (Fuentes et al. 2020). Pixels from these three classes were summed to calculate the extent of their occurrence. Average water occurrence extent was also calculated as a multiplication of surface occurrence by the pixel area and lumped at the scale of the study region (mean area flooded in the study region).

### Trend Analysis

The land cover rasters and the estimated water occurrence collection were analysed in order to detect temporal changes. Therefore, annual maps were processed through trend analysis tests. In the first case, a trend analysis of the extent for different land cover classes was carried out addressing the overall temporal variability aggregated at the regional scale. In the second case, the analysis consisted of a pixel-based evaluation of water occurrence trends addressing the within-region variability of surface water in time and space. In both cases, non-parametric tests were executed in Python and included the Mann Kendall test and the Sen’s slope, which estimates the median slope of all pairs of points.

| Table 1: Drought/wet severity classes determined by SPI and SPEI |
|-------------|-------|-------|
| Severity classes | SPI | SPEI |
| extreme drought | ≤ -2 | ≤ -2 |
| severe drought | -1.9 - -1.5 | -1.9 - -1.5 |
| moderate drought | -1.4 - -1 | -1.4 - -1 |
| mild drought | -0.9 -0 | -0.9-0 |
| mild wet | 0-0.9 | 0-0.9 |
| moderately wet | 1-1.4 | 1-1.4 |
| severely wet | 1.5-1.9 | 1.5-1.9 |
| extremely wet | ≥ 2 | ≥ 2 |

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Both methods are robust and allow relaxation of some of the requirements for linear trend analysis, such as the normality and homogeneity of residuals. The significance of the tests was also reported to detect significant trends \((p\text{-value} < 0.05)\). Additionally, for those datasets with less than 30 records, such as the land cover rasters obtained from the MCD12Q1 dataset, a pre-whitening of the Mann Kendall test was also applied (Bayazit and Önöz 2007). In the case of river discharge, trends were analysed from 1970 through Ordinary Least Squares (OLS).

**Vegetation Evaluation**

The Normalized Difference Vegetation Index (NDVI) was used as a proxy for vegetation growth (Wan et al. 2019). Therefore, NDVI maps were generated from the Landsat merged collection to monitor vegetation. Surface reflectance bands from Landsat namely the near-infrared and red bands were used to calculate the NDVI. Then NDVI was estimated as the ratio between the difference in reflectance between the near infrared and red wavelengths and the sum of the reflectance at these two wavelengths, to capture the amount of change in the vegetation cover due to drainage/reflooding periods. Since the wetlands have been reported to have experienced dramatic changes in the medium-term (Richardson and Hussain 2006), we selected an interval of five years to detect changes in vegetation, taking the advantage of Landsat satellites that have relatively high spatial resolution (30 m) and high revisit time. Eight images of NDVI were calculated where each NDVI image represents a mean of five years (except the NDVI in 1972 which was calculated from images in August only) starting from 1972 to 2020 to allow monitoring of medium-term vegetation change detection. These images were used to quantify the changes that occurred since 1972. Since NDVI responds in a particular way to different land cover classes (DeFries and Townshend 1994), a range of values between \(> 0\) and 0.2 was assumed to categorise bare soil, and a range between \(> 0.2\) and 1 was used for vegetation.

**Results**

While a strong seasonal pattern in rainfall can be observed at the monthly scale (Fig. 3a), with a lumped intra-annual rainfall pattern ranging from around 0 \(\text{m}^3\) to about \(4 \times 10^8 \text{m}^3\), no trend could be observed in the time series for the study period, which can be additionally inferred from the annual rainfall barplot (Fig. 3b). However, some extreme rainy months can also be observed, in which monthly lumped rainfall has reached almost \(8 \times 10^8 \text{m}^3\).

Series of SPI and SPEI are presented in Fig. 4. While SPI shows a clear dominance of dry scenarios in the 1983–1995 and 2008–2018 period, these are reduced in duration in the SPEI series. On the contrary, SPI shows that the 1995–2008 and 2018–2020 periods are dominated by wet scenarios in

![Fig. 3 Lumped rainfall (monthly aggregated) in the study region on (a) a monthly basis and (b) annually aggregated](image-url)
terms of rainfall, which increase in duration in the agricul-
tural and hydrologic dimensions (over 3-months scale), but
these are not clearly evident in the SPEI series. Differences
between both indices arise as a result of evapotranspiration,
which may be dominant in the time series. That is the rea-
son why in several cases, the duration of such dry - wet
conditions reduces when evapotranspiration is considered.
Highlights in the last period, 2018–2020, the severity of the
wet conditions, which can be observed in both indices, being
also evident from the lumped rainfall.

Mean annual streamflow for the river gauges analysed
are in Fig. 5. Discharge in both rivers has in general
decreased from 1930. However, from 1970, only the
Tigris river downstream the Kut barrage (T20) presents

Fig. 4 Average regional SPI (left) and SPEI (right) series using 1-, 3-, 6- and 12-months

strong evidence (OLS p-value < 0.05) of a decreasing
trend. The magnitude of the trend is quite large,
implies an annual decreasing discharge rate of 15.22
m$^3$s$^{-1}$. This means that the incoming flow to the study
region has strongly decreased from 1970. However, the
trend analysis only covers the period until 2005, and
might help to explain any change in the marshes until
that year. This progressive reduction in discharge in both
rivers has been, among others, a response to damming,
which has implied the building of 32 dams in the Tigris
and Euphrates rivers and their tributaries in Turkey, and
14 dams or barrages in both rivers in Iraq until 2017
(Adamo et al. 2020).

Fig. 5 River discharge from
gauge stations located upstream
the study region in the
Euphrates river downstream the
Hindiya barrage (E03) and in
the Tigris river downstream the
Kut barrage (T20). A significant
trend from 1970 is also pre-
sented in the plot
Land Cover Changes

Land cover trends in terms of surface extent are presented in Table 2. Overall, strong evidence of land cover changes has been registered, especially in the extent of wetland, urban and water areas. In these cases, a trend is evident using both the Mann Kendall and the pre-whitened Mann Kendall methodologies. These trends imply significant changes in the extent of some of the land cover classes, including the wetlands, since 2001.

The results based on the most conservative approach (pre-whitening Mann Kendall) are showing that, while urban areas have grown at a small trend of 0.15 km² y⁻¹ (15 hectares per year), wetlands and water extents present much larger change rates (20.94 km² y⁻¹ and 8.79 km² y⁻¹, respectively). However, these trends do not contribute to understanding where such changes are occurring and are most probably a response to the restoration plans started in 2003.

Surface water dynamics

The different lumped water fraction extents and the mean surface water extent derived from Landsat images between 1986 and 2019 are presented in Fig. 6.

A seasonal oscillation of surface water extent is present in the area, which is evident in cycles of wet and dry years, which may be a response to interannual climate variability. The different surface water fractions split quite evenly. It highlights the year 2019 for presenting the largest extent of surface water, and the years 1998–2003 for being the driest in the study period. Even though no trend was found in the surface water fractions when lumped at the regional scale, large and significant local changes are evident in Fig. 7.

Similar to the result found using MODIS data, the annual surface water occurrence time series indicates strong evidence of 1,138 km² experiencing significant changes in surface water occurrence, corresponding to 7.39% of the study area extent, presenting a mean rate of change of 0.72% per year. From this, 704 km² experience positive surface water occurrence trends at a mean rate of 1.94% per year. On the other hand, the remaining 433 km² experience negative surface water occurrence trends at a rate of, on average, 1.25% per year. While the upstream areas present mostly negative trends, the opposite behaviour can be observed in the downstream areas of the marshes, which may be affected by artificial hydraulic modifications. The marshland area faced a high level of anthropogenic modifications during the 1980s war and the 1990s’ economic sanctions (Ochsenschlager 2004; Lawler 2005; Ahram 2015). For many reasons, the “Ba’athist Regime” has built an extensive drainage network on and around the marshlands south of Iraq, particularly within the study-site (Fig. 8). This led to severe changes in the natural surface water pattern that needs to be further studied to reach to further conclusions about the reasons behind these modifications and the effect on the ecosystems at various localities.

The most obvious change in water bodies (Fig. 9) was due to disappearance/reappearance of water (as a natural seasonal process) and due to drainage and reflooding periods (including human intervention). However, a significant loss (533 km²; Supplementary 4) of permanent water extent can be observed, mostly in the eastern part of the marshlands. This significant loss was caused by building dams upstream and drainage canals, especially around the Hammar and Central marshes (Al-Chibayish), which led to diversion of water from the marshes (Lawler, 2005). Large areas (~ 52% of the study area, equivalent to 8,222 km²) did not experience any change since the 1980’s. Whereas, new seasonal areas (1,253 km²) have appeared in the marshes, and this was expected because of the government legislated restoration plans to restore Mesopotamia’s marshlands (after 2003; effectively after 2005).

Response of Vegetation Cover to Drainage and Reflooding Periods

Significant changes in the vegetation have taken place during the study period. Figure 10 shows that the vegetation cover was not affected by any stressors especially in the Central marsh for the periods (1985–1990 and 1990–1995). However, the subsequent two periods (1995–2000 and 2000–2005) showed that there was a massive reduction in the vegetation cover which subsequently affected the ecosystem functioning in this marsh (Hashim et al., 2019). After 2003 the restoration plan began, which started effectively after 2005, as a scheduled reflooding program (Richardson and Hussain 2006). The reflooding periods helped this marsh to recover slightly during the periods (2005–2010 and 2010–2015) (Albarakat et al. 2018). This helped most of the plant species to re-appear in this marsh (Hamdan et al. 2010). The last period (2015–2020) did not show any decline in the NDVI, meaning that vegetation was not affected in this period.

The vegetation covers experienced successive loss of extent, especially until 2005 (Supplementary 3), which resulted from the vanishing marshlands extent. Changes from vegetation to bare soil were associated with each NDVI threshold, where we used a threshold of 0.2 to capture the transition from non-vegetation to vegetation. According to the statistics, bare soil and vegetation covers for the period 1985–1990 were about 8699.52 and 4487.95 km², corresponding to 56.5% and 29.1% of the study region extent, respectively (Supplementary 5). For the period 2000–2005, the vegetation cover had lost its extent vastly to be about 1450.95 km² (9.4% of the study region), whereas bare soil
extent increased to about 13,376.48 km² (86.8% of the study region). After the start of the reflooding (2003) programmes, the area of the vegetation cover had noticeably increased due to releasing water into these marshes to cover about 3,921.24 km² (25.4% of the region). This has been accompanied with an increase of the marshland’s extent (~1,928.75 km²) for the period 2015–2020, whereas bare soil extent reduced to about 9,536.3 km² (61.9% of the region) and this was expected after the recovery of the marshlands.

Discussion

Monitoring and investigating the spatio-temporal changes within the ancient Mesopotamian marshlands were the main aims of this study. The best extent/availability of satellite data was employed from 1972 to 2020. Several studies have successfully been carried out combining different remote sensing datasets to characterise ecosystem changes, including wetlands (Ozesmi and Bauer 2002; Bergen and Dronova 2007; Kaplan and Aydan 2018; Pastick et al. 2019). In the present study, the different remote sensing datasets used present different spatial resolutions and temporal coverage lengths, which might cause different issues. For instance, MODIS covers the time series between 2001 and 2019, which led to a positive trend in surface water and wetland extent. On the other hand, Landsat time series covered more consistently from 1986, leading to longer time series that showed a decrease in wetland extent during some periods until 2005. Thus, as stated by Serinaldi et al. (2018), trends may differ depending on the length of the time series evaluated.

In terms of spatial resolution, different sources were evaluated independently of each other. Therefore, resolution differences should not affect the results. On the other hand, temporal resolution of the data, despite being limited, allowed the characterisation and continuous monitoring of marshes through a linear interpolation of recording gaps. This was due to the relatively slow recession of vegetation and surface water. Faster processes may require a higher temporal resolution, which might be achieved by using MODIS or Sentinel satellites (Drusch et al. 2012; Fuentes et al. 2019). However, these present shorter temporal coverage, and therefore a balance of pros and cons should be done.

Table 2

| Land cover | Mann Kendall Z score | Sen’s slope (km² y⁻¹) | p-value | Pre-whitening Mann Kendall Z score | Pre-whitening Sen’s slope (km² y⁻¹) | Pre-whitening p-value |
|------------|----------------------|------------------------|---------|-----------------------------------|------------------------------------|------------------------|
| Grasslands | 2.66                 | 24.10                  | 7.84e-03 | -                                 | -                                  | -                      |
| Wetlands   | 3.56                 | 26.29                  | 3.59e-04 | 2.88                              | 20.94                              | 3.99e-03               |
| Urban      | 5.79                 | 0.54                   | 6.92e-09 | 4.36                              | 0.15                               | 1.30e-05               |
| Barren     | 3.01                 | -55.06                 | 2.62e-03 | -                                 | -                                  | -                      |
| Water      | 3.36                 | 13.38                  | 7.83e-04 | 2.35                              | 8.79                               | 1.88e-02               |

*Non-significant trends obtained (p-value > 0.05)

Fig. 6

Annual time series of surface water fractions and average surface water extents in the study region.
based on the goals of projects to make the correct decision on the sources of remote sensing data to use.

The importance of the studied wetlands lies in their essential ecosystem functionality and because they host communities of dwellers in these semi-dry regions (Al-Hilli et al. 2009). Marshlands have overall been negatively impacted in the past on several temporal scales during this case study period, presenting small lapses of recession and regrowth. On the spatial scales, significant regional and within-region changes have occurred affecting the marshland’s extent from 1972 to 2020. These changes could have happened due to several reasons: locally (including the bordering wars, and all its associated activities, as well as the governmental water control schemes); regionally at the catchment scale (including the upstream damming and construction on the main runoff streams); and globally (e.g., due to climate change and precipitation reductions). However, the analysis of the rainfall time series does not show evidence of long-term rainfall declines at the regional scale and even in wet years, the marshlands have shown to decline. Similarly, Al-Quraishi and Kaplan (2021) did not find significant trends in annual rainfalls in Iraq and Turkey, but decreasing trends were reported in Syria. On the other hand, Euphrates discharge upstream the study region has strongly decreased in the last decades as a result of damming, while Tigris presents a large fluctuation, has also been affected by dam construction, and also shows negative trends but using a larger period of analysis (1930–2020; Al-Quraishi and Kaplan 2021). Therefore, the incoming flow to feed the marshes has strongly decreased. This, together with the drainage carried out in the marshes, has led to drying the marshes, especially in the period 1996–2004, with a strong reduction in open water and vegetation extents. These changes have also led to salt accumulation and strong increases in salinities, which are also hindering the recovery of the marshes (Richardson et al. 2005; Sama et al. 2012). Understanding environmental changes in the region should ultimately provide a basis to make important decisions to address the health of the marsh system.

Even though lumped local rainfall shows some fluctuations over time, it does not show a long-term drying trend. Likewise, periodic cycles of interannual variability might be observed, leading to wet and dry conditions, but these do not strongly relate to the marshland dynamic. This might imply two things: (1) local rainfall has not played a major role in the flooding/drainage pattern, except in some large rainfall events such as those observed in 2019; rather human intervention such as upstream damming, over-exploitation or poorly planned water management through drainage in the catchment may have led to most changes in the marsh area, or (2) rainfall trend patterns might be spatially variable, which might be obscured when aggregating rainfall at the catchment scale. This study has confirmed the literature and

**Fig. 7** Surface water occurrence trends using the Man Kendall and the Sen’s slope tests in the study region
shown that drainage and reflooding periods in the Mesopotamian wetlands led to disappearance and reappearance of water in the three major marshes – the Hammar, Hawizeh, and the Central marshes (including Al-Chebaish marshes). These periods of drainage and reflooding have caused a reduction in the wetland’s extent by about −15%.

Although Table 2 of the land cover change analysis has shown an increasing trend in most of the surface cover classes, and particularly within the wetlands and water bodies, it was actually a reflection of the MODIS data length that started in a very dry period in 2000. In fact, this period post-2001 has turned into a slightly positive water occurrence (and its vegetation consequences), particularly after the restoration efforts in 2003. This was confirmed in Fig. 6, which shows a clear decline in water occurrence from 1980 s to 2001 and a slight increase after 2000 (the MODIS launch date).

The most obvious change in water bodies (Figs. 7 and 9) was due to a decrease in the incoming flow to the region caused by the progressive building of dams upstream the marshes and the disappearance/reappearance of water due to the initial drainage and later reflooding periods, which was labelled as “lost seasonal” surface. A significant loss (533 km²) in permanent water extent can be observed mostly in the eastern part of the marshlands. The eastern marshes are on/near the political border between Iraq and Iran, and the decline in water extent is partly related to the long border war that was concentrated within this area, as well as...
previous government policies of water diversion. Strong wet conditions have affected the region from mid 2018 to 2020 which has translated into a high surface water extent and a regain in marshland areas.

Furthermore, permanent water bodies delineated in the transition map (Fig. 9) can also be contrasted with the trends presented in Fig. 7. However, it is clear from both sources of information that water infrastructure built downstream from the study region had a slight effect on the surrounding marshland areas but does not translate into seasonal water changes. That should indicate that the key controlling factors may come from the upstream area (the catchment).

The fluctuation from wet to dry periods is reflected by the significant NDVI dynamism, which indicates a significant change in the vegetation cover. Figure 10 shows that there was a slight decrease in the NDVI values between 1972 and 1995 indicating that the vegetation cover was not much affected by any stressors especially in the Central marsh for the periods 1985–1990 and 1990–1995. However, the decline in the NDVI trend showed sharp reductions in the vegetation cover in the periods 1995–2000 and 2000–2005, which caused the destruction of the ecosystem in these marshes. Since 2005, the NDVI trend has gained some positivity of vegetation occurrence as a reflection of the reflooding schemes, which have helped these marshes to recover, even during moderate to severe drought conditions. Moreover, within the past five years (2015–2020), the NDVI trends showed a significant increase, which indicates the success of the restoration efforts and the occurrence of strong wet conditions. This has helped most of the plant species to re-appear in these marshes (Hamdan et al. 2010).

However, the marshlands did not regain their original extent seen in the 1970’s and 1980’s despite the releasing of water into them. This might indicate the need to understand that releasing water after long and successive drought periods might not be enough to recover all the vanished plant species, nor the ecosystem’s biodiversity that was developed over thousands of years. Additionally, climate drivers were studied in the region, but the catchment delineation goes beyond the borders defined in this study. Therefore, climate in neighbouring regions is required to further understand the changes taking place. However, this will be a matter for future studies.
Conclusions

Landsat data allowed to monitor the water occurrence and surface cover classes (and the dynamism of the NDVI in particular) in an area size of ~15,000 km², showing an adequate resolution for this study; both spatially (covering the whole study site) and temporally (1972–2020). The available MODIS dataset allowed obtaining clear land cover change trends. However, its limited time coverage precludes it from making conclusions before 2001. A significant overall decrease in the wetlands, including surface water and the vegetation cover extents, with a major decline at around 2000 was observed. Since then, trends have increased until 2020, helped in the last period for strong wet conditions. In other words, 2000 could be considered as a turning year in the environmental situation of the study site, which is supported by the positive trends in water and marshland extents found with MODIS.

On the longer-term, significant wetlands reductions since 1972 can be explained by several environmental and
anthropogenic factors including: intensive local activities (e.g., the Iraqi government planned “drainage” of the marshes plus the eight years of war); damming of upstream rivers, catchment modifications (especially water diversion); and climate change. Some of these factors play a major role in the decline of marshland area within the study site, including the drainage networks constructed by the government. Others, such as rainfall, do not seem to have significantly changed over time. Thus, local human interventions seem to be the major causes affecting the marshland ecosystems. Modern restoration has started a “slow” recovery of the marshes’ water extent but not to their original state.

Thus, it is essential for the future research to provide a greater understanding of the main influencing factors from outside the Mesopotamian marshes with additional investigations to follow up some of the key-points raised by this study, including: the need for a detailed study of the catchment area modifications and water discharge fluctuations; local soil/water quality sampling and analysis; and ground truthsing the whole remotely sensed data and its analysis.

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