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Research papers

The impact of reservoir construction and changes in land use and climate on ecosystem services in a large Mediterranean catchment

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ABSTRACT

The Mediterranean region has been identified as one of the most affected global hot-spots for climate change, which is already manifested by faster increasing temperatures than the global mean and significant decreases in annual precipitation. Besides, over the past decades, important land cover changes have occurred, such as reforestation, agricultural intensification, urban expansion and the construction of many reservoirs. Here we study the impacts of these changes in a typical large Mediterranean catchment and focus on relevant ecosystem services, i.e. primary production, water supply, food production, water regulation, flood and erosion control, and cultural ecosystem services. We applied a hydrological model, coupled with a soil erosion and sediment transport model, for the period 1971–2010 to the entire catchment, and for an extended period (1952–2018), with more pronounced land use changes, to one of its subcatchments that plays a crucial role for regional drinking water provision. Climate change in the evaluated period is characterized by a decrease in precipitation and an increase in temperature. Land use changes included an increase in natural vegetation in the headwaters, agricultural intensification and reservoir construction in the central part, and urban expansion in the downstream part of the catchment. The positive impacts on ecosystem services were quantified by a decrease in hillslope erosion, sediment yield, sediment concentration and flood discharge, which are mainly attributed to climate change and reservoir construction. The negative impacts were quantified by an increase in plant water stress and a decrease in reservoir storage, for which the latter is mainly attributed to reforestation in the headwater catchments. While grey infrastructures like reservoir construction showed important positive impacts on some ecosystem services, such as water regulation and flood control, they also potentially induce negative impacts, such as an increase of water demand and ecological integrity. We argue that a shift is needed to green infrastructures and nature-based solutions, such as reforestation and sustainable land management, which may lead to similar benefits on ecosystem services, but without the negative impacts caused by grey infrastructures. This supports the need for integrated land-use management to sustainably protect ecosystem services, considering the expected climate change impacts on Mediterranean environments and societies.

1. Introduction

The Mediterranean region has been identified as one of the most affected global hot-spots for climate change (Giorgi et al., 2006). Recent observations have shown that temperature in the Mediterranean region is increasing at a faster rate than the global mean (Cramer et al., 2018). It is expected that this trend will continue under future climate change, accompanied by a decrease in annual precipitation sum (García-Ruiz et al., 2011), leading to an increase in the frequency, intensity and duration of droughts and a decrease in stream flow (Burke et al., 2006). Impacts of climate change have already been observed in many Mediterranean rivers in the last decades (Gallart and Llorens, 2004; López-Moreno et al., 2006; García-Ruiz et al., 2011; Buendia et al., 2016; Pérez-Cutillas et al., 2018; Masseroni et al., 2020). Land use changes, such as reforestation, deforestation and expansion of (intensive) agriculture, are likely to amplify the impact of climate change on ecosystem services (Sala et al., 2000; García-Ruiz et al., 2011). The large scale construction of reservoirs in the Mediterranean region has also affected river flow and sediment dynamics over the past century (Cooper et al., 2013). Together, these changes impacted the most relevant ecosystem services for Mediterranean environments, including supporting (primary production), provisioning (water supply and food production), regulating (water regulation, flood and erosion control), and cultural ecosystem services. This illustrates the importance to consider

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feedbacks between hydrological processes, land use, society and ecosystems through an interdisciplinary approach, connecting hydrological changes with dynamics of socio-ecosystems (Montanari et al., 2013; Ceola et al., 2016).

A major part of land use in Mediterranean regions is dominated by agricultural areas (Cooper et al., 2013), covering around 40% of the Mediterranean basin (Malek and Verburg, 2017). While total agricultural area remained stable, there has been a shift from rainfed agriculture in headwater catchments to more intensive (irrigated) agricultural practices in the downstream areas (Puigdefábregas and Mendizabal, 1998; García-Ruiz et al., 2010; Nainggolan et al., 2012). Abandoned agricultural land and intensified agriculture are both prone to land degradation, often leading to increases in surface runoff and soil erosion (Lesschen et al., 2007; García-Ruiz et al., 2010; García-Ruiz and Lana-Renault, 2011; Zink et al., 2011; Rogger et al., 2017). Land abandonment took place where irrigation water was not readily available, where mechanization was more difficult to implement or in areas further away from the main distribution infrastructure, such as in mountainous areas (Nainggolan et al., 2012). While soil erosion often increases in the first years after agricultural lands have been abandoned (Poesen and Hooke, 1997; García-Ruiz et al., 2008), under favourable environmental conditions these lands are usually occupied by natural vegetation of shrubland and forest cover after several years (Sirami et al., 2010; Hansen et al., 2013), leading to a decrease in soil erosion (Alatorre et al., 2012) and an increase in soil organic carbon (Novara et al., 2017). The return of natural vegetation, especially forest cover, may have a significant impact on the hydrological cycle and impacts downstream water availability. Most importantly, forest cover increases rainfall interception and transpiration, ultimately leading to a decrease in stream flow (Andréassian, 2004; Brown et al., 2005; Nosetto et al., 2005; López-Moreno et al., 2006; Pérez-Cutillas et al., 2018).

Land use changes often take a relatively long time to have a measurable impact on stream flow and sediment dynamics in rivers, whereas, the construction of dams, as a typical example of grey infrastructures, often has an immediate effect (Wang, 2014). Over the past century, river flow in many Mediterranean catchments has been regulated by the construction of dams. In the Mediterranean basin, most dams were built in Spain and Turkey, many with the purpose to store water for irrigation (Cooper et al., 2013). Dams can have benefits to some hydrological ecosystem services, such as flood control and water regulation (Thoms and Sheldon, 2000; Cudennec et al., 2007; López-Moreno et al., 2011; Cooper et al., 2013). Reservoir construction has mitigated the impact of droughts on stream flow at short time scales, but amplified the impact on longer time scales because of an increase in water demand after reservoir construction (Lorenzo-Lacruz et al., 2013). Alterations of the flow by dam construction also have negative ecological consequences, including a reversal of the seasonal flow pattern, affecting the ecological flow regime of rivers (Belmar et al., 2010). The high degree of fragmentation in Mediterranean rivers due to dam construction affected ecological processes that depend on longitudinal river connectivity, such as transport of organic and inorganic matter, and upstream and downstream movements of aquatic and riparian species (Lehner et al., 2011; Grill et al., 2019). Dams alter the sediment transport dynamics of rivers, most notably by trapping large amounts of sediments that are transported by the river flow. Often, dams trap almost all sediment that is transported towards the reservoir (Vörösmarty et al., 2003; Vericat and Batalla, 2006), potentially leading to channel incision downstream of dams (Vericat and Batalla, 2006) and to morphological changes in the river delta (Syvitski et al., 2009; Ibánez et al., 2019). Because dams trap such large volumes of sediments, reservoirs have a limited lifetime. It has been estimated that the capacity loss of the world's reservoirs is about 1% (WCD et al., 2000; Wisser et al., 2013), which translates to an expected lifetime of 100 years. For Mediterranean areas even higher annual capacity losses (2.5%) are reported leading to even shorter expected reservoir lifetimes (Vanmaercke et al., 2011). Given their often high implementation and maintenance costs and undesired side effects, there is increasing discussion about the effectiveness of grey infrastructures, like dams, to control hydrological and sediment fluxes effectively. As an alternative, there is increasing interest in green infrastructures and nature-based solutions based on vegetation, land use and land management changes, or 'building with nature' to gradually control hydrology and sediment fluxes (Cohen-Shacham et al., 2019).

The Mediterranean climate is characterized by high inter-annual variability in precipitation, summer drought and intense rainfall (Cortesi et al., 2012; Merheb et al., 2016). The combination of these climate characteristics and the typical steep slopes have resulted in a high frequency of flash floods and high erosion rates in the Mediterranean region (Llasat et al., 2010; Tarolli et al., 2012). Over the last few decades, discharge observations have shown a decreasing trend in stream flow in many Mediterranean catchments (Lorenzo-Lacruz et al., 2012; Vicente-Serrano et al., 2019). Part of this decline is explained by climate change through a decrease in precipitation and an increase in temperature, from which the latter causes increased evapotranspiration (López-Moreno et al., 2011; Pérez-Martín et al., 2014; Vicente-Serrano et al., 2014; Lutz et al., 2016; Gudmundsson et al., 2017). In mountainous catchments there is a tendency of an earlier peak in spring flow, caused by an increase in temperature affecting snow melt (García-Ruiz et al., 2011; Morán-Tejeda et al., 2014). While climate change may have had an important impact on stream flow, anthropogenic factors, such as water abstractions for irrigation purposes and reforestation, have played an important role as well (López-Moreno et al., 2011; Milano et al., 2012; Vicente-Serrano et al., 2019), illustrating the feedback between societal changes and hydrological processes (Ceola et al., 2016). Some studies projected that stream flow will decrease even more due to decreasing annual precipitation sum and increasing evapotranspiration under climate change (García-Ruiz et al., 2011; Cramer et al., 2018; Brogli et al., 2019), while other studies highlight the expected increase in extreme precipitation (Tramblay et al., 2018), which could lead to increased runoff and discharge to reservoirs under climate change (Eekhout et al., 2018; Blöschl et al., 2019).

In the second half of the 20th century, many land use and climate changes have occurred in the Mediterranean region, significantly altering the hydrological cycle. The combination of these changes have led to complex interactions among several ecosystem services (Doblas-Miranda et al., 2017). For instance, reforestation may lead to a decrease in hillslope erosion (e.g. interrill and rill erosion), but often leads to a decrease in downstream water availability, reflecting benefits and trade-offs for different ecosystem services (Boix-Fayos et al., 2007, 2015; Ren et al., 2014). Better understanding of these interactions, both from grey and green infrastructures, including feedbacks between hydrology and social systems, and how they lead to benefits and trade-offs for ecosystem services is important to design climate change adaptation strategies through land use and management practices (e.g. Montanari et al., 2013; Cohen-Shacham et al., 2019).

The objective of this study is to assess how historical land use and climate change and the construction of a network of large reservoirs in the Segura River catchment in the last 70 years have interacted and affected a selection of most relevant ecosystem services. By putting emphasis on ecosystem services we aim to contribute to better understanding and prediction of interactions between hydrological and societal changes, which are essential to determine water security and to set priorities for environmental management (e.g. Montanari et al., 2013; Eekhout et al., 2018; Eekhout and de Vente, 2019). The Segura catchment is a typical Mediterranean catchment where many of the before mentioned changes have occurred. For this assessment we applied a hydrological model, coupled with a soil erosion and sediment transport model, to study the impact of climate change, land use change and reservoir construction on the most relevant ecosystem services. We applied the model to the Segura River catchment for the period 1971–2010 and for an extended period (1952–2018) to the Taibilla subcatchment, which plays an important role in the provision of

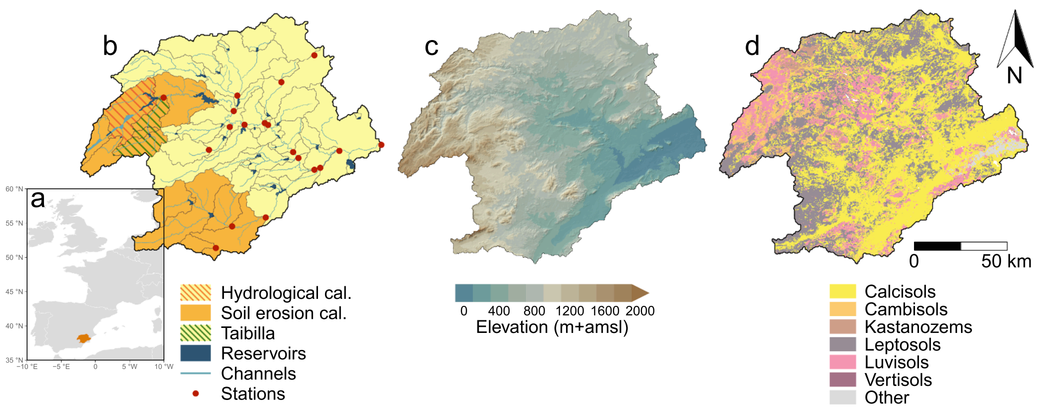

Fig. 1. Location and characteristics of the Segura River catchment: (a) location of the catchment within Europe, (b) the hydrological calibration area (red hashes), soil erosion calibration area (orange), Taibilla subcatchment (green hashes), the channels (light blue), the reservoirs (dark blue), and the stations (red dots), (c) Digital Elevation Model (Ministerio de Fomento de España, 2015), and (d) soil class map (Hengl et al., 2017). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

drinking water in the region and is characterised by pronounced land use changes in that period. With this study we provide a practical example of a methodology to support adaptation of water resources management to the current and future challenges posed by environmental and societal change (Ceola et al., 2016).

2. Material & methods

2.1. Study area

This study is performed in the Segura River catchment (15,955 km^2) located in the southeast of Spain (Fig. 1a). The elevation ranges between sea level and 2336 masl. Catchment-averaged annual rainfall amounts to 377 mm (for the period 1981–2010; Serrano-Notivoli et al., 2017) and mean annual temperature ranges between 10.7 and 18.6 °C (1981–2010; Herrera et al., 2016) from the headwaters to the downstream area. The climate is Mediterranean (Csa and Csb according to the Köppen-Geiger climate classification) in the headwaters (21%) and semi-arid (BSk and BSh) in the rest of the catchment (79%). The main soil classes are Calcisols (41%), Leptosols (35%), Luvisols (4%) and Kastanozems (4%) (Hengl et al., 2017).

It has been estimated that in the Segura River catchment water demands exceed supply by 224% (Sabater and Tockner, 2009). To meet the demand, since the 1980s water from the Tagus River is transported to the Segura River catchment through an inter-basin water transfer over a length of 292 km. Besides, 32 reservoirs have been constructed in the catchment, with a total capacity of 1230 Hm^3 (Fig. 1b and Table S1). Fourteen reservoirs are allocated exclusively for irrigation purposes, the other reservoirs have mixed functions for drinking water supply, electricity generation and flood prevention.

In the Segura catchment we assessed the impacts of land use and climate change and the construction of reservoirs for the period 1971–2010. In addition, we performed a long-term land use change impact assessment in the Taibilla subcatchment (595 km^2), located in the headwaters of the Segura River catchment for the period 1952–2018. The reason for the focus on the Taibilla subcatchment is that it is located in a part of the Segura River catchment with more pronounced land use changes over the past decades. Besides, the Taibilla reservoir (constructed in 1979, with a capacity of 9 Hm^3) plays an essential role in provision of drinking water to approximately 2.5 million people in the provinces Murcia, Albacete, and Alicante (Grindlay et al., 2011; Melgarejo-Moreno et al., 2019). Since 1955, water from the Taibilla subcatchment is diverted through a water diversion work and transported to downstream areas for domestic use. Previous research showed that the drinking water supply may be under threat due to a decline of runoff caused by reforestation in this subcatchment (Quiñonero-Rubio et al., 2016; Pérez-Cutillas et al., 2018). The elevation in the Taibilla subcatchment ranges between 524 and 2030 masl. Catchment-averaged annual rainfall amounts 514 mm (1981–2010) and mean annual temperature is 12.4 °C (1981–2010). The climate is mainly classified as Mediterranean (Csa; 88%), but the downstream part of the subcatchment is classified as semi-arid (BSk; 12%). The main soil classes are Leptosols (53%), Cambisols (17%), Calcisols (12%) and Luvisols (10%) (Hengl et al., 2017).

2.2. Model

To assess the impacts of reservoir construction and changes in land use and climate, we applied the SPHY model (v3.0; Terink et al., 2015; Eekhout et al., 2018; FutureWater, 2019), a spatially distributed hydrological model, fully coupled with a soil erosion and a sediment transport model. The model is applied on a cell-by-cell basis and with a daily time step. We employed a model grid resolution of 200 m. The hydrological model simulates most relevant hydrological processes, such as interception, evapotranspiration, surface runoff, and lateral and vertical soil moisture flow. Water is routed through the catchment using a single-flow algorithm, accounting for reservoir operations through the reservoir module. The reservoir module determines reservoir outflow based on reservoir capacity and actual storage (Hanasaki et al., 2006). We replaced the fixed exponent (1.5) from the model proposed by Hanasaki et al. (2006) by a model parameter (b), to account for the variation in reservoir regulation among the reservoirs in the study area. The soil erosion model, based on Morgan and Duzant (2008), simulates most relevant hillslope erosion processes, such as soil detachment by raindrop impact and runoff. The sediment is transported through the catchment accounting for the transport capacity of the flow and reservoir sedimentation through a trapping efficiency equation. The model incorporates a vegetation model based on the spatial and temporal variation of the Normalized Differenced Vegetation Index (NDVI), which determines actual evapotranspiration, interception, canopy storage, throughfall and canopy cover.

The hydrological model was calibrated in a headwater subcatchment (696 km^2; Fig. 1b) for the period 2001–2010 and validated for the period 1987–2000 using daily observed discharge data. We applied the SPOTPY calibration package (Houska et al., 2015) to calibrate the hydrological model, using the Simulated Annealing algorithm to optimize the Kling-Gupta model efficiency [KGE; (Gupta et al., 2009)], which has an optimum at 1. The calibration resulted in a KGE of 0.62 and the validation in a KGE of 0.58 for daily discharge (Fig. S2). The reservoir module was calibrated separately and includes two parameters, which were optimized for KGE for the 18 reservoirs for which storage and outflow observations were available (Table S3). The median values of the optimized parameters were assigned to the remainder of the reservoirs. The soil erosion model was calibrated in two large subcatchments (Fig. 1b) for the period 2001–2010 and validated for the period 1981–2000, using literature data for hillslope erosion per land use class (Maetens et al., 2012) (Table S5). The sediment transport model was calibrated with reservoir sedimentation data from 6 reservoirs (Avendaño-Salas et al., 1997) and optimized for KGE, yielding an optimized value for the calibration of 0.89 and for the validation of 0.57

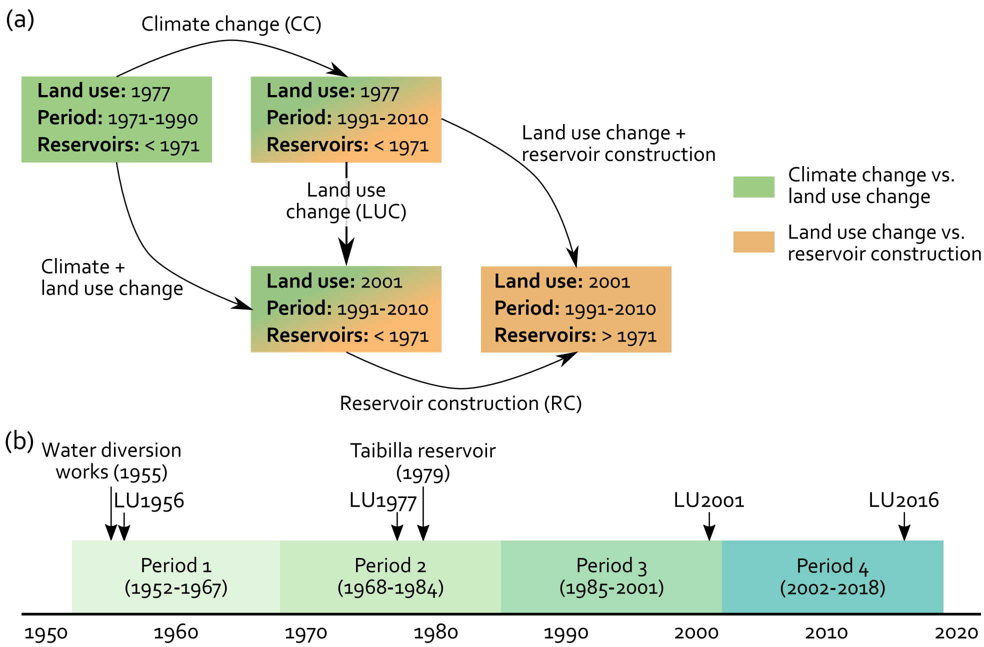

Fig. 2. (a) Scenarios applied to the Segura River catchment, with in green the scenarios on the impact of climate change and land use change and in orange the scenarios on the impact of land use change and reservoir construction. (b) The 4 periods that were considered for the long-term scenario study in the Taibilla subcatchment. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

(Table S6). See the SI for a detailed description of the calibration procedure of all model components and the optimized parameter values.

2.3. Scenarios

For the Segura River catchment, we focussed our analysis on two cases. First, we analysed how land use change and climate change impacted ecosystem services. For this analysis we used two land use maps from the Spanish Ministry of Agriculture, Fisheries and Food for the years 1977 and 2001 (Ministerio de Agricultura Pesca y Alimentación, 2010). We subdivided the climate time series into two periods of 20 years, i.e. 1971–1990 and 1991–2010. In this analysis we used three model runs (Fig. 2a; green boxes), i.e. land use 1977 for the period 1971–1990, land use 2001 for the period 1991–2010, and land use 1977 for the period 1991–2010. This allowed us to study the isolated impact of land use change, the isolated impact of climate change and the combined impact of land use and climate change. In these scenarios we only considered the reservoirs that were constructed prior to 1971.

Second, we analysed how reservoir construction impacted ecosystem services. We applied two scenarios, one with only the reservoirs constructed prior to 1971 and one with all the reservoirs that were constructed between 1971 and 2010. Furthermore, we assessed how the impact of land use change compares with reservoir construction. In this analysis we used three model runs, considering only the period 1991–2010 (Fig. 2a; orange boxes), i.e. land use 1977 without new reservoirs, land use 2001 without new reservoirs, and land use 2001 with new reservoirs. This allowed us to study the isolated impact of land use change, the isolated impact of reservoir construction and the combined impact of land use change and reservoir construction.

The combined impact of climate change, land use change and reservoir construction was obtained by comparing two model runs, i.e. land use 1977 for the period 1971–1990 without new reservoirs and land use 2001 for the period 1991–2010 with new reservoirs.

For the Taibilla subcatchment we extended our analysis with two additional land use maps, i.e. from 1956 and 2016. These two maps were obtained by aerial photo interpretation, applying a digitization scale of approximately 1:5000. The land use maps of 1977 and 2001 were revised with aerial photos for the Taibilla subcatchment. The analysis for the Taibilla catchment focusses on the combined trends in land use and climate change over the period 1952–2018. This period was subdivided into four equally sized periods of 17 years, for which we respectively used the land use maps of 1956 (1952–1967), 1977 (1968–1984), 2001 (1985–2001) and 2016 (2002–2018) (Fig. 2b).

2.4. Input data

All input maps were interpolated or resampled to the 200 m model resolution. Daily precipitation and temperature data were obtained from the SPREAD and (Serrano-Notivoli et al., 2017) and SPAIN02 (Herrera et al., 2016) daily datasets, with a 5 km and 0.11° resolution, respectively. These climate data were only available until 2012. For the last period in the Taibilla subcatchment (2002–2018) we obtained daily data from 39 precipitation and 33 temperature stations surrounding the subcatchment. These data were interpolated onto the same grid as used in the SPREAD and SPAIN02 datasets, using the interpolation techniques from Serrano-Notivoli et al. (2017) and from Herrera et al. (2016) for precipitation and temperature, respectively.

The digital elevation model was obtained from a Spanish national LiDAR dataset (Ministerio de Fomento de España, 2015) with a 5 m resolution. Soil texture and organic matter were obtained from the SoilGrids global dataset (Hengl et al., 2017) with a 250 m resolution. The organic matter map was aggregated per land use class from the land use map of 2001, to obtain average organic matter content per land use class. Subsequently, the aggregated values were applied to all land use maps to obtain specific organic matter maps. All soil data were interpolated onto the model grid using bilinear interpolation.

NDVI images were obtained from bi-monthly Moderate Resolution Imaging Spectroradiometer (MODIS) data for the period 2000–2012. No NDVI images were available for the entire historic period considered in this study. Therefore, we separated the NDVI data into inter- and intra-annual NDVI estimates, which were combined to obtain NDVI time series needed for the model runs. To determine the inter-annual NDVI we applied a land use specific log-linear model based on annual precipitation and temperature time series. The intra-annual NDVI was determined from the long-term average NDVI, also differentiated per land use class. See the SI for a detailed description of the methods applied to obtain NDVI time series.

2.5. Ecosystem services indicators

We evaluated the impact of climate change, land use change and

Table 1
Ecosystem services and related ecosystem services indicators used in this study, based on Millennium Ecosystem Assessment (2005) and Costanza et al. (2017).

Ecosystem services		Indicator
Group	Subgroup	
Supporting	Primary production	Plant water stress[a]
Provisioning	Water supply	Runoff
		Reservoir storage
	Food production	Plant water stress[b]
Regulating	Water regulation	Low flow
	Flood control	Flood discharge
	Erosion control	Hillslope erosion
		Sediment yield
		Sediment concentration
Cultural[c]		Plant water stress[d]

[a] For natural land use classes.
[b] For agricultural land use classes.
[c] Including the aesthetic and recreational subgroups.
[d] For urban land use classes.

reservoir construction on a selection of ecosystem services that are most relevant for Mediterranean environments. We quantified these ecosystem services using a number of spatially distributed and station indicators, which are related to supporting, provisioning, regulating and cultural ecosystem services (Table 1). Our modelling approach allows to evaluate the impact on a series of ecosystem services indicators for which no or limited data are available for the study area and within the study period. The spatially distributed indicators include runoff, plant water stress and hillslope erosion. Plant water stress is an indicator of the amount of stress plants experience and ranges between 0 (no stress) and 1 (fully stressed). We determined plant water stress according to the formulations by Porporato et al. (2001) and Allen et al. (1998), which account for a plant-specific stress parameter and the modelled soil moisture content (see the SI for a detailed description of the plant water stress formulations).

The station indicators include reservoir sediment yield, sediment concentration, reservoir storage, flood discharge and low flow. Reservoir sediment yield and reservoir storage were only obtained at the 13 reservoirs that were constructed prior to 1971 and were determined by the annual sum and annual average, respectively. The other three station indicators were obtained at 19 locations situated near important cities, at river bifurcations and at the catchment outlet (Fig. 1b). The sediment concentration was determined from the average daily sediment flux divided by the average daily discharge. The flood discharge and low flow were determined from the discharge time series and as the median annual maximum and minimum discharge over the 20-year simulation period, respectively.

2.6. Uncertainty analysis

To account for uncertainty, we evaluated the magnitude of change of the climate change trends and ecosystem services indicators. We applied the effect size (*ES*) to determine the magnitude of change. Unlike t-test statistics, the effect size is not affected by the sample size and therefore is an appropriate statistical test to be applied on large datasets (Lin et al., 2013). The effect size is determined according to:

$$ES = \frac{\mu_1 - \mu_2}{s} \tag{1}$$

with μ_1 and μ_2 the mean of the two populations that are being compared and s the pooled standard deviation, which is determined by:

$$s = \frac{(n_1 - 1)s_1^2 + (n_2 - 1)s_2^2}{n_1 + n_2 + 2} \tag{2}$$

with n_i and s_i the size and the standard deviation of the ith population, respectively.

In case of the spatially distributed indicators, we determined the effect size per subcatchment, where the population consisted of all the grid cells within the subcatchment. In case of the station indicators, we determined the effect size per station, where the population consisted of the annual output. An effect size value of 0.2 is considered small, 0.5 is considered medium and 0.8 is considered large (Cohen, 1988).

In the Taibilla subcatchment we evaluated the significance of the trends of the annual climate change and ecosystem services indicators using the non-parametric Mann–Kendall test (Mann, 1945; Kendall, 1975).

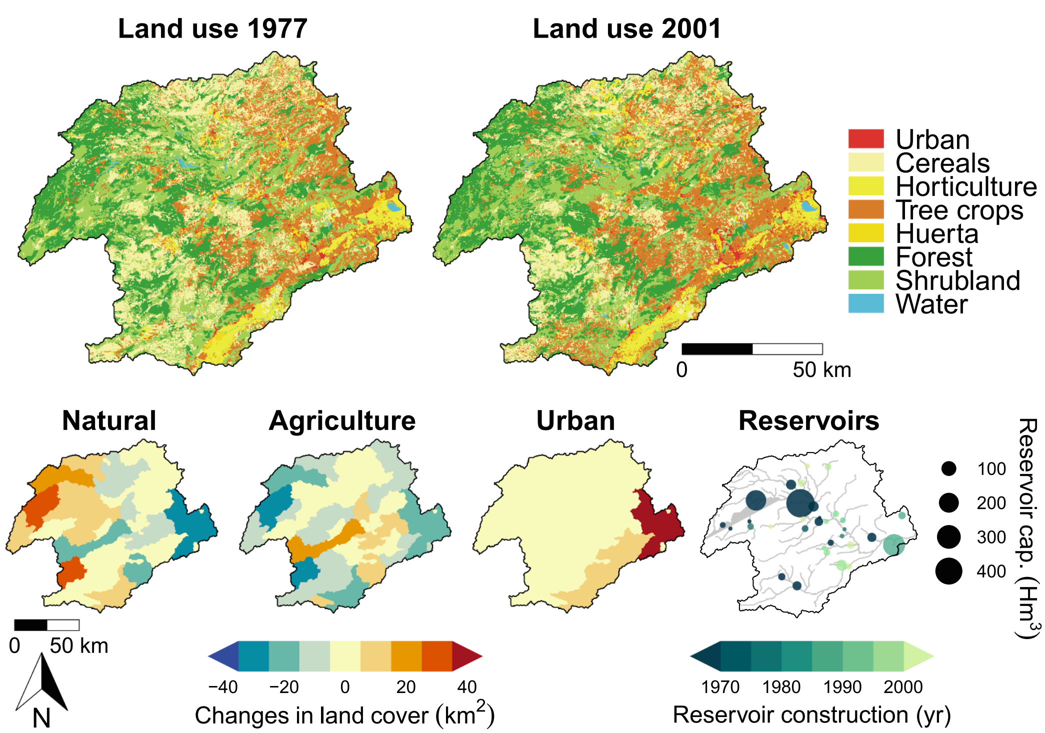

Fig. 3. Land use and land use change maps for the Segura River catchment. The upper row shows the land use maps of 1977 and 2001. The lower row shows (1) the changes in land use (km^2) for natural, agricultural and urban land uses aggregated per subcatchment, and (2) reservoir construction (yr) and their storage capacity (Hm^3).

3. Results

3.1. Land use change and reservoir construction

The land use in the Segura River catchment is dominated by natural (53%) and agricultural land use (45%) (Fig. 3, upper row, and Table S7), divided over forest and shrubland, and tree crops (almond trees, fruit trees, vineyards and olive trees) and cereals, respectively. Between 1977 and 2001, land use changed in 33.8% of the catchment and can be characterized by a transformation within natural vegetation and agricultural land uses, and urban expansion (Fig. 3, lower row). Natural vegetation increased in the headwater catchments and decreased in the central and eastern part of the catchment (Fig. S3). While on average, forest cover increased and shrubland cover decreased (Table S7), in the central part of the catchment large areas converted from forest to shrubland (Fig. S3). Apart from the most eastern part of the catchment, a change in natural vegetation coincides with an opposite change in agricultural land use. Agricultural expansion is most prominent in the central part of the catchment, mainly as a result of an increase in tree crops (fruit tree and almond tree orchards) and to a lesser extent to an increase in horticulture. On the contrary, cereal fields have been abandoned, resulting in an increase in natural vegetation, or have been replaced by other agricultural practices, such as tree crops and horticulture. As noted before, in the eastern part of the catchment the decline in natural vegetation has been the result of urban expansion.

The first reservoirs in the Segura River catchment were constructed in the beginning of the 19th century (Fig. 3, lower right, and Table S1). These reservoirs were built for a mixed use of flood defence and irrigation purposes and were mostly constructed in the headwater streams of the catchment as typical grey infrastructures. Half way the 20th century a few hydropower plants were build, with a total capacity of around 130 MW (Confederación Hidrolgráfica del Segura, 2015). Since 1971, the number of reservoirs increased from 13 to 33, leading to an increase in the reservoir capacity from 820 to 1230 Hm^3. In this period, some large reservoirs were built with the sole purpose to store water for irrigation, but most reservoirs that were built since 1971 have the purpose to reduce flood risk in the downstream parts of the catchment and, therefore, have a limited storage capacity.

3.2. Historical climate change

Annual precipitation sum decreased with 6.5% ($ES = 0.15$) within the period of consideration, from a catchment-average of 402 mm to 376 mm (Fig. 4 and Table S8). Precipitation decreased most in the headwater catchments, locally exceeding 100 mm. The largest relative decrease occurred in the central part of the catchment, with decreases up to 20%, which is also emphasized by a large effect size. Average annual precipitation increased in the southwest part of the catchment, locally up to 30%.

Extreme precipitation, obtained from the precipitation concentration index [PCI; (Martin-Vide, 2004)], which reflects the precipitation concentration at a daily time scale, also decreased in the same period, with a catchment-average decrease of 4.0% ($ES = 0.40$). Most decrease in extreme precipitation is recorded in the central part of the catchment, with a decrease up to 20%. Similar to the change in annual precipitation, extreme precipitation increased in the southwest and northwest part of the catchment, with increases up to 10%.

The average temperature has increased with 0.4 °C from a catchment-average of 14.8 to 15.2 °C ($ES = 0.18$). Most increase in temperature was recorded in the southern, eastern and northern part of the catchment, with increases locally up to 1.5 °C. The temperature decreased up to 1 °C in the central-western part of the catchment.

3.3. Impacts on ecosystem services indicators

Climate change had a bigger impact on ecosystem services than land use change over the study period (Table 2). Climate change caused a catchment-average decrease in runoff by 12.5% ($ES = 0.20$). The northern and western part of the catchment were most affected by a decrease in runoff due to climate change (Fig. 5, upper row), which coincide with the areas where precipitation sum decreased most. Land use change had a significantly smaller impact on runoff, with a catchment-average increase of 0.1% ($ES = 0.00$). Most increase is shown in the downstream part of the catchment and is mostly related to urban expansion, where the increase in paved surface caused an increase in (surface) runoff. Runoff increased also in some other parts of the catchment, related to the change in forest to shrubland (Fig. S3). The combined impact of climate and land use change clearly shows that climate change dominates the impact on runoff, although some changes as a result of land use change are visible as well.

Climate and land use change led to a catchment-average increase in plant water stress of 2.9% ($ES = 0.14$) and 1.7% ($ES = 0.11$), respectively. Climate change caused an increase in plant water stress in all except the central part of the catchment (Fig. 5, middle row), which is related to the decrease in precipitation sum and an increase in temperature in these areas. Land use change caused an increase in plant water stress on locations where agriculture expanded, while plant water stress decreased where land use changed from forest to shrubland (Fig. S3). The impact on plant water stress is amplified by the combination of climate and land use change, leading to an increase of 4.6% ($ES = 0.25$).

Climate and land use change led to a catchment-average decrease in hillslope erosion of 17.5% ($ES = 0.02$) and 2.3% ($ES = 0.00$), respectively, ultimately leading to combined decrease in hillslope erosion of 19.8% ($ES = 0.02$). While the relative change in hillslope erosion is high, especially as a result of climate change, the results are of little statistical significance because changes occurred in a limited part of the catchment, mainly in the areas where most water accumulates (Fig. 5,

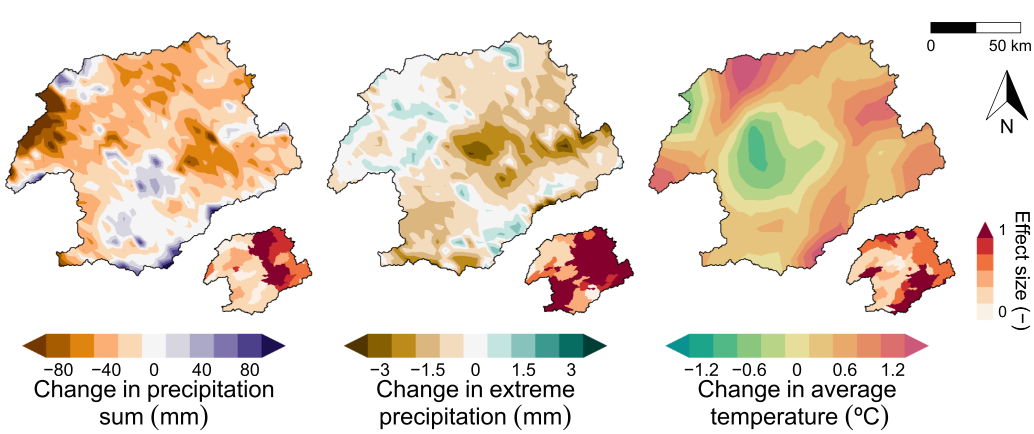

Fig. 4. Historical climate change in the Segura River catchment between the periods 1971–1990 and 1991–2010, with change in annual precipitation sum (mm), extreme precipitation (mm) and average temperature (°C). Extreme precipitation is determined with the PCI (Martin-Vide, 2004). The small maps in lower right corner show the effect size.

Table 2
Change in ecosystem services indicators as a result of climate change (CC), land use change (LUC), reservoir construction (RC) and the combination of the three (CC + LUC + RC). Runoff (mm), plant water stress (PWS; -) and hillslope erosion (Mg km^{-2} yr^{-1}) are catchment-average values of Fig. 5 and the effect size was obtained considering all grid cells of the model domain. Sediment yield (Gg yr^{-1}) and reservoir storage (Hm^3) were determined as the sum of all reservoirs that were build prior to 1971. Sediment concentration (g l^{-1}), flood discharge (m^3 s^{-1}) and low flow (m^3 s^{-1}) were obtained at the catchment outlet. The effect size of the latter five indicators was obtained from the annual time series.

Scenario	Runoff (mm)	PWS (–)	Hillslope erosion (Mg km^{-2} yr^{-1})	Sediment yield (Gg yr^{-1})
CC	−15.10 (−12.5%)	0.02 (2.9%)	−5.20 (−17.5%)	−390.62 (−11.7%)
	ES = 0.20	*ES* = 0.14	*ES* = 0.02	*ES* = 0.28
LUC	0.17 (0.1%)	0.01 (1.7%)	−0.68 (−2.3%)	−79.64 (−2.4%)
	ES = 0.00	*ES* = 0.11	*ES* = 0.00	*ES* = 0.04
RC	0.00 (0.0%)	0.00 (0.0%)	−0.22 (−0.7%)	−1045.00 (−31.3%)
	ES = 0.00	*ES* = 0.00	*ES* = 0.00	*ES* = 0.15
CC + LUC + RC	−14.93 (−12.4%)	0.03 (4.6%)	−6.09 (−20.6%)	−1515.26 (−45.4%)
	ES = 0.20	*ES* = 0.25	*ES* = 0.02	*ES* = 1.40
	Sediment concentration (g l^{-1})	Reservoir storage (Hm^3)	Flood discharge (m^3 s^{-1})	Low flow (m^3 s^{-1})
CC	−0.052 (−3.5%)	−13.46 (−6.2%)	−23.95 (−15.3%)	−4.13 (−19.4%)
	ES = 0.17	*ES* = 0.36	*ES* = 0.34	*ES* = 0.44
LUC	0.003 (0.2%)	−1.69 (−0.8%)	0.93 (0.6%)	−0.13 (−0.6%)
	ES = 0.01	*ES* = 0.04	*ES* = 0.02	*ES* = 0.01
RC	−1.152 (−78.3%)	0.00 (0.0%)	−47.98 (−30.7%)	5.78 (27.2%)
	ES = 5.00	*ES* = 0.00	*ES* = 0.89	*ES* = 0.44
CC + LUC + RC	−1.20 (−81.6%)	−15.15 (−6.9%)	−70.99 (−45.5%)	1.53 (7.2%)
	ES = 4.69	*ES* = 0.40	*ES* = 1.08	*ES* = 0.14

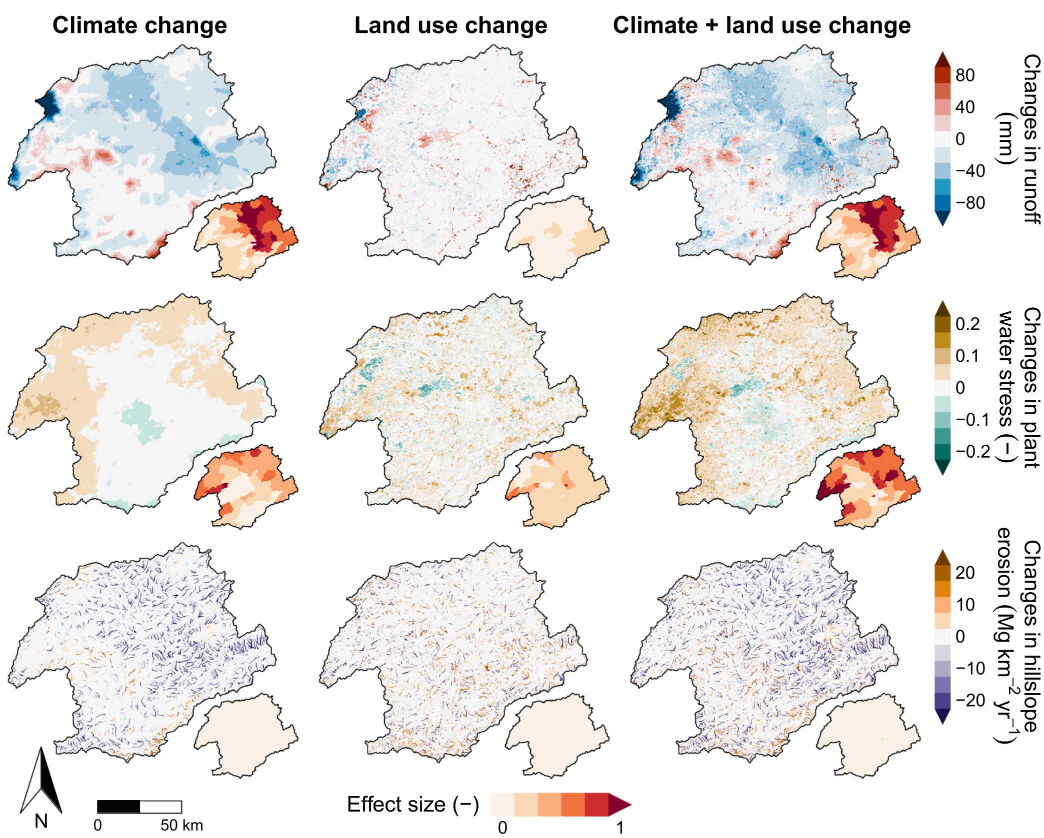

Fig. 5. Change in ecosystem services indicators as a result of climate change, land use change and the combined climate and land use change. The results are shown for runoff (mm), plant water stress (-) and hillslope erosion (Mg km^{-2} yr^{-1}). The small maps in the lower right corner show the effect size per subcatchment.

lower row). This is particularly the case for the change in hillslope erosion caused by climate change. The areas where hillslope erosion increased (or decreased) due to climate change largely coincide with the areas where runoff increased (or decreased). Land use change had a mixed impact on hillslope erosion. Hillslope erosion increased in locations where land use changed from natural to agricultural land use or where cereals were replaced by tree crops. Hillslope erosion decreased in areas of reforestation and urban expansion.

Climate and land use change led to similar relative changes in reservoir sediment yield as hillslope erosion (Table 2), although with higher statistical significance. The sediment concentration at the catchment outlet decreased as a result of climate change, which is the combined result of a decrease in runoff and hillslope erosion. While land use change resulted in a small increase in catchment-average runoff, reservoir storage decreased due to land use change. The reason for this is that most reservoirs constructed prior to 1971 are located in the headwater

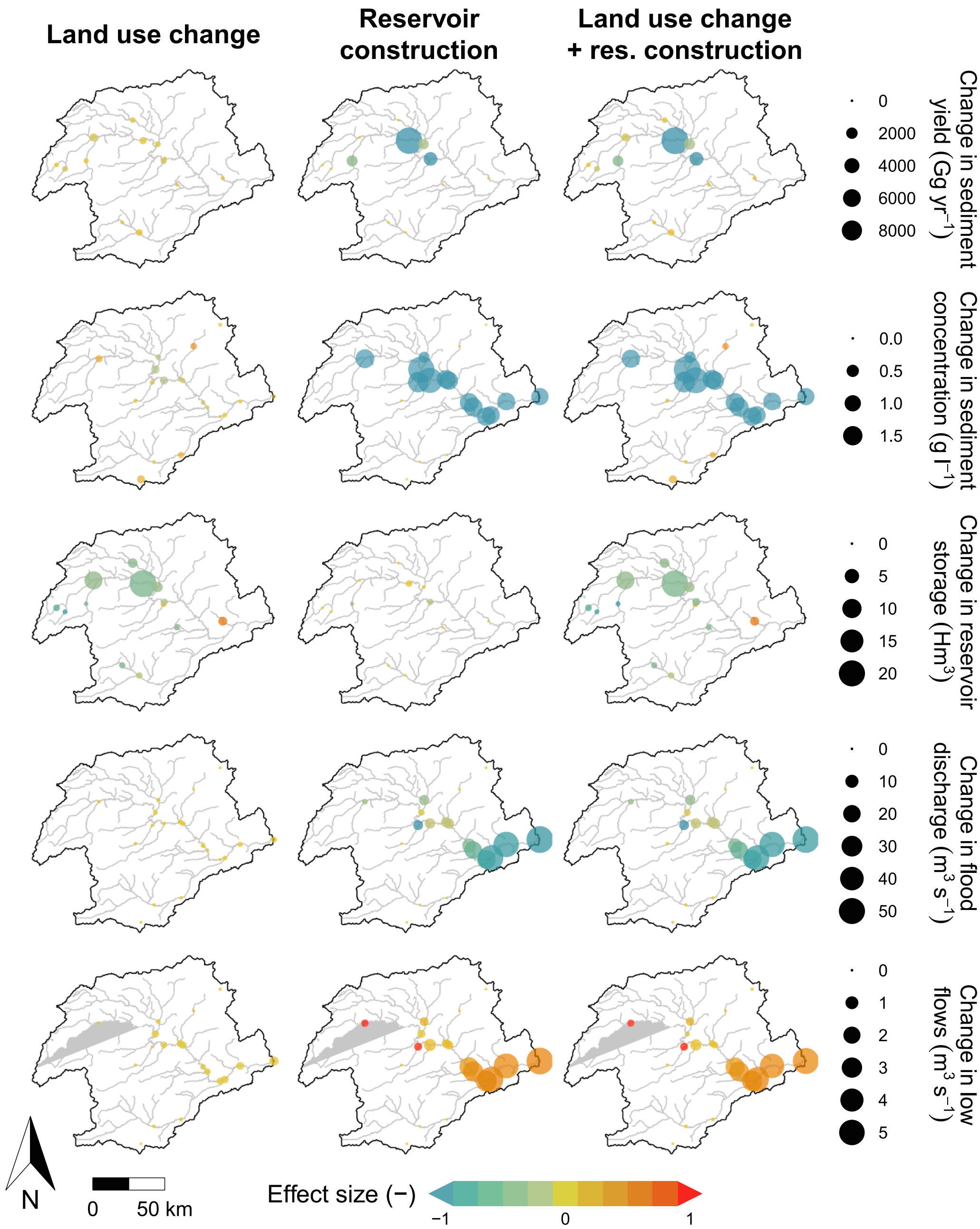

Fig. 6. Change in ecosystem services indicators in the stations and the reservoirs constructed prior to 1971. Changes are shown as a result of land use change, reservoir construction and the combination of land use change and reservoir construction. The results are shown for reservoir sediment yield (Gg yr^{-1}), sediment concentration (g l^{-1}), reservoir storage (Hm^{-3}), flood discharge (m^{-3} s^{-1}) and low flow (m^{-3} s^{-1}). The size of the circles indicates the amount of change in the ecosystem services indicators. The colors indicate the direction of change, with blue a decrease and red an increase. The intensity of the colors indicates the effect size related to the change. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

catchments where runoff decreased as a result of land use change (Fig. 5). Flood discharge and low flow at the catchment outlet show similar results as compared to catchment-average runoff, with a decrease caused by climate change and a minor change caused by land use change.

Reservoir construction led to a decrease in sediment yield in the reservoirs constructed prior to 1971 and sediment concentration at the stations, while land use change only led to minor changes (Fig. 6). Reservoir storage is the only indicator where land use change had a bigger impact than reservoir construction, because since 1971 only a few reservoirs were built upstream of existing reservoirs. Reservoir construction notably impacted flood discharge and low flow, especially in the most downstream located stations, with a decrease in flood discharge and an increase in low flow.

The combined impact of climate change, land use change and reservoir construction resulted in distinct responses among the ecosystem services indicators (Table 2). The decrease in runoff (−12.4%; *ES* = 0.20) was dominated by the changes caused by climate change. Plant water stress increased with 4.6% (*ES* = 0.25), which is the combined effect of climate change and land use change. All three factors of change attributed to the decrease in hillslope erosion (−20.6%; *ES* = 0.02) and sediment yield (−45.4%; *ES* = 1.40). The decrease in sediment concentration (−81.6%; *ES* = 4.69) is mainly attributed to reservoir construction and the decrease in reservoir storage (−6.9%; *ES* = 0.40) to climate change. Flood discharge decreased with 45.5% (*ES* = 1.08) due to the combined effect of climate change and reservoir construction. The decrease in low flow due to climate change is counteracted by the increase in low flow due to reservoir construction, ultimately resulting in an increase of 7.2% (*ES* = 0.14).

3.4. Long-term impacts in a mountainous subcatchment

Land use has significantly changed in the period from 1956 to 2016 in the Taibilla subcatchment (Fig. 7). Most notably is the increase in forested area, from 28% in 1956 to 61% in 2016. The new forested areas have covered land previously used for cereal cultivation and shrubs, which decreased from 25% to 7% and from 46% to 27%,

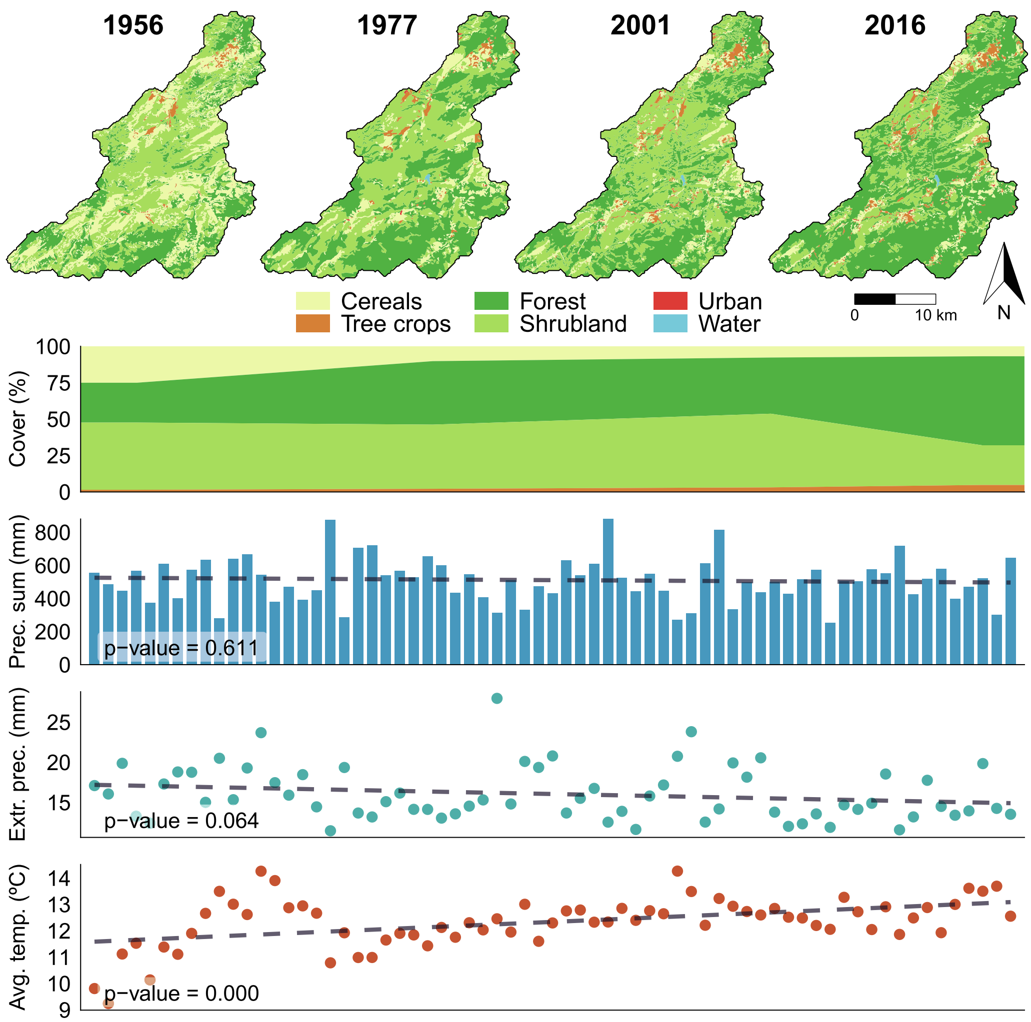

Fig. 7. Land use maps and the temporal evolution of land use and the climate signal for the Taibilla subcatchment. The upper row shows the land use maps of 1956, 1977, 2001 and 2016. The second row shows the temporal change in land use (%) from 1952–2018. The last three rows show the temporal change in annual precipitation sum (Prec. sum; mm), extreme precipitation (Extr. prec.; mm) and average temperature (Avg. temp.; °C). Extreme precipitation is determined with the PCI (Martin-Vide, 2004). These three time series include the linear trend over the study period (dashed line), where the p-value shows the significance of these trends obtained from the Mann–Kendall test.

respectively. Annual precipitation has not significantly changed in the study period. Extreme precipitation decreased from around 17 mm to 15 mm. Average temperature significantly increased with on average 0.23 °C per decade.

The interplay between climate and land use change in the Taibilla subcatchment has resulted in a significant decrease in runoff and hillslope erosion and a significant increase in plant water stress (Fig. 8). Considering that annual precipitation sum hardly changed over the 67-year study period, runoff was mainly impacted by the increase in temperature and increase in forest cover, which resulted in an increase in evapotranspiration and, consequently, an increase in plant water stress. The decrease in hillslope erosion is mostly caused by the sharp increase in forest cover, although this can also be attributed to the decline in extreme precipitation. The Taibilla reservoir, located in the central part of the catchment, was built in 1979, hence, due to the relatively short period, climate and land use change did lead to significant trends in sediment yield and reservoir storage yet. However, sediment concentration at the subcatchment outlet significantly decreased, which is mainly caused by the construction of the water diversion work in 1955 and the Taibilla reservoir in 1979. The decrease in flood discharge is mostly in line with the decrease in runoff.

4. Discussion

4.1. Positive and negative impacts on ecosystem services

The changes that have taken place in the Segura River catchment from the 1950's onwards can be characterized by reforestation and agricultural land abandonment in the headwater catchments, and reservoir construction, agricultural intensification and urban expansion in the central and downstream part of the catchment (Fig. 9). In the headwater catchments, agricultural land abandonment and increased forest cover resulted in an overall greening-up (Boix-Fayos et al., 2007; García-Ruiz and Lana-Renault, 2011; Pérez-Cutillas et al., 2018; van Leeuwen et al., 2019). These changes led to a decrease in runoff and hillslope erosion, comparable with results obtained in other mountainous Mediterranean catchments (e.g. López-Moreno et al., 2011; Lasanta and Vicente-Serrano, 2012; Buendia et al., 2016). Runoff significantly decreased in the Taibilla subcatchment, even though annual precipitation sum did not significantly change for this subcatchment between 1952 and 2018 (Fig. 7). This finding confirms the studies of Boix-Fayos et al. (2007) and Pérez-Cutillas et al. (2018), who showed from observations that runoff measured at the Taibilla reservoir significantly decreased in the last 70 years, heavily impacting catchment-scale fluvial morphology and channel erosion processes. The decrease in runoff led to a decrease in reservoir storage in many headwater reservoirs (Fig. 6). Plant water stress increased in the headwaters, which is mostly the result of decreasing annual precipitation and increasing temperatures in parts of the headwaters (Figs. 4 and 5), but also as a result of an increase in forest cover, due to higher evaporative water demand of forest as compared to arable land. Hillslope erosion in the headwaters decreased on average, due to the increase in natural vegetation at the expense of arable land. The land use changes that have occurred in the headwaters had a positive impact on regulating

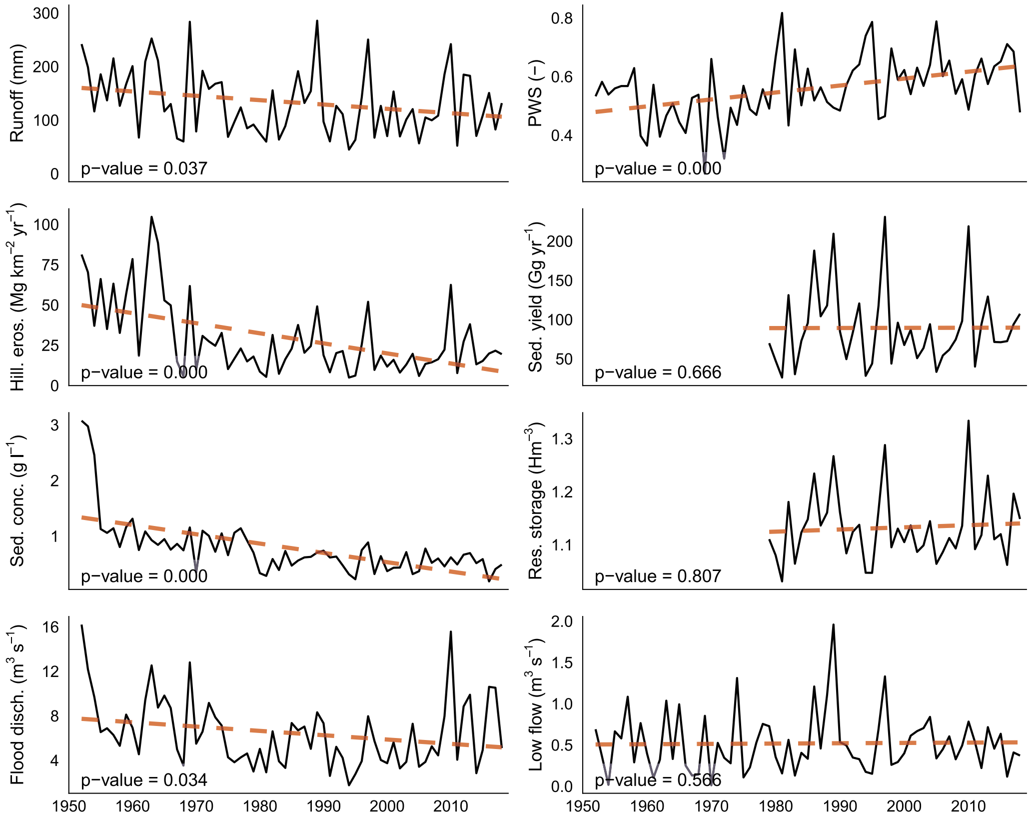

Fig. 8. Temporal change in the in ecosystem services indicators for the Taibilla subcatchment. Runoff (mm), plant water stress (PWS; -) and hillslope erosion (Mg km^{-2} yr^{-1}) are catchment-average values. Sediment yield (Gg yr^{-1}) and reservoir storage (Hm^3) were determined at the Taibilla reservoir. Sediment concentration (g l^{-1}), flood discharge (m^3 s^{-1}) and low flow (m^3 s^{-1}) were obtained at the subcatchment outlet. The time lines include the linear trend over the study period (dashed line), where the p-value shows the significance of these trends obtained from the Mann–Kendall test.

ecosystem services such as erosion control, but with trade-offs affecting other ecosystems services such as water supply due to decreased runoff and primary production due to increased plant water stress (Table 3), confirming previous findings on the complex interactions and potential trade-offs that arise after land use change (Doblas-Miranda et al., 2017; Requena-Mullor et al., 2018).

Land use changes in the central and downstream areas are characterised by an increase in intensive agriculture (horticulture and tree crops) and urban expansion, replacing rainfed cereals and natural vegetation. Similar land use changes have occurred in other Mediterranean areas (e.g. Lasanta and Vicente-Serrano, 2012; Requena-Mullor et al., 2018). The urban expansion in the downstream areas of the Segura River catchment follows the general trend of the increase in urban areas in the Mediterranean basin over the last decades (Underwood et al., 2009). While urban expansion had a positive impact on water supply and erosion control (Table 3), it is likely that these changes contributed to an increase in flood discharge and hillslope erosion downstream from new urban areas (Trimble, 1997; Nelson and Booth, 2002). Urban expansion reduces infiltration due to a significant increase in paved surface. This causes an increase in plant water stress (a function of soil moisture, see SI), negatively affecting cultural ecosystem services, such as aesthetics and recreation. This illustrates the importance of accounting for interactions between hydrological processes and societal changes (Montanari et al., 2013). The increase in

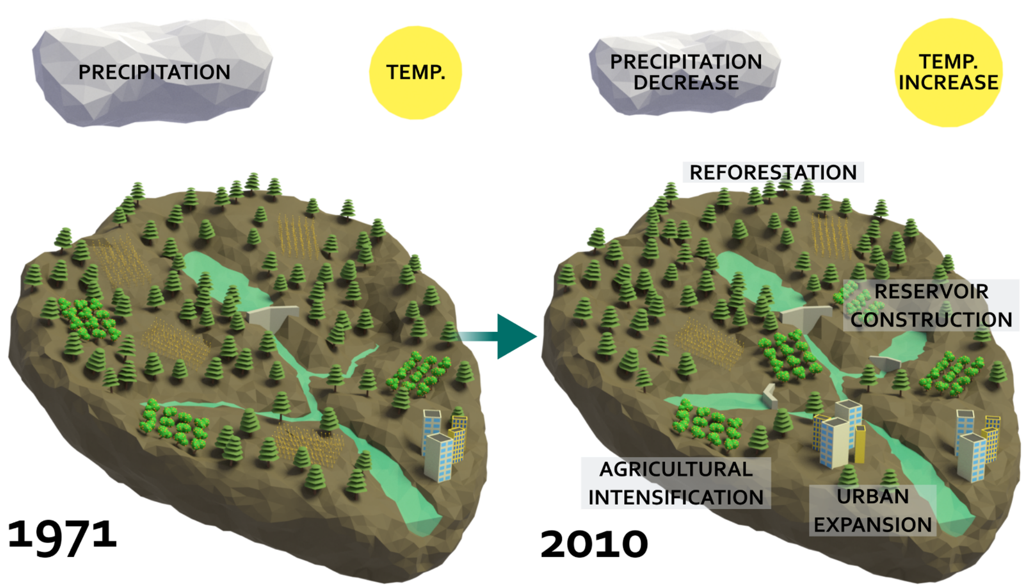

Fig. 9. Summary of the changes that have occurred between 1971 and 2010 in the Segura River catchment.

Table 3
Significant changes in indicators and related ecosystem services per factor of change identified in this study.

Factors of change	Positive impacts		Negative impacts	
	Indicators	Ecosystem services	Indicators	Ecosystem services
Climate change[a]	Flood discharge ↓	Flood control	Reservoir storage ↓ Low flow ↓	Water supply Water regulation
Reforestation[b]	Hillslope erosion ↓ Sed. concentration ↓ Flood discharge ↓	Erosion control Flood control	Runoff ↓ Plant water stress ↑	Water supply Primary production
Agricultural intensification[c]			Plant water stress ↑	Food production
Urban expansion[c]	Runoff ↑	Water supply	Plant water stress ↑	Cultural
Reservoir construction[a]	Hillslope erosion ↓ Sed. concentration ↓ Flood discharge ↓ Low flow ↑	Erosion control Erosion control Flood control Water regulation		

[a] Based on Table 2 for changes with an $ES > 0.2$.
[b] Based on Fig. 8 for changes with a $p < 0.05$.
[c] Based on Table S9 for changes with an $ES > 0.2$.

agriculture in the central part of the catchment resulted in an increase in plant water stress (Fig. 5), which is the result of a change in plant-specific parameters describing stress (depletion fraction). Agricultural intensification is most likely related to the construction of reservoirs in this area (Di Baldassarre et al., 2018), although the main purpose of these reservoirs is flood control, rather than irrigation water provision. Reservoir construction resulted in several positive impacts, like a significant increase in low flow and a significant decrease in flood discharge and sediment concentration (Fig. 6), positively affecting several ecosystem services, such as erosion and flood control, and water regulation. These positive consequences converge with the objectives of the hydrological control works by building a large network of grey infrastructures consisting of check dams in several parts of the catchment since the 1970's to regulate floods and control erosion (Boix-Fayos et al., 2008; Quiñonero-Rubio et al., 2016).

4.2. Green infrastructure vs. grey infrastructure

Due to continuing population growth, urbanization and an increase in intensive irrigated agriculture, water demand and soil erosion are likely to intensify in the Mediterranean Basin in the future (Underwood et al., 2009; Cooper et al., 2013). Moreover, climate change is expected to cause a decrease in annual precipitation sum and an increase in temperature and extreme precipitation in the Mediterranean region, leading to an additional increase in plant water stress and hillslope erosion (García-Ruiz et al., 2011; Eekhout et al., 2018). In the same study area, Eekhout et al. (2018) showed that the projected increase in extreme precipitation may lead to a redistribution of water within the catchment, from green water (water stored in the soil) to blue water (water stored in reservoirs). This will most likely promote the shift from rainfed to irrigated agriculture, and increase the dependency on reservoir storage and significantly increases the pressure on groundwater resources (Aeschbach-Hertig and Gleeson, 2012).

Integrated and catchment-wide measures based on interdisciplinary assessments are needed to mitigate the expected negative impacts of future land use and climate change on provisioning and regulating ecosystem services, such as water supply, and flood and erosion control (e.g. Ceola et al., 2016). Measures are broadly subdivided into two categories: (1) green infrastructures, such as reforestation/afforestation and sustainable land management, and (2) grey infrastructures, such as construction of reservoirs and check dams. In the past, a combination of green and grey infrastructures has been implemented in many Mediterranean catchments. For example, it is still a common practice in headwater catchments to combine reforestations with the construction of erosion control works, such as check dams, to decrease sediment yield in downstream located reservoirs. Grey infrastructures are, however, often short-lived (e.g. < 30 years for check dams; (Quiñonero-Rubio et al., 2016), due to the trapping of sediment and damage during floods. Besides, on the medium-long run they may lose their effectiveness by provoking downstream channel erosion (Boix-Fayos et al., 2007). A recent study performed in the Taibilla subcatchment confirmed the preference of green infrastructures over grey infrastructures considering costs and benefits of optimizing the provision of several ecosystem services (i.e. climate regulation, moderation of extreme events, soil condition and erosion control; Boix-Fayos et al., 2020). The construction of large reservoirs has similar effects as check dams by retaining large volumes of sediments, affecting downstream hydrological and channel morphological processes (Vericat and Batalla, 2006; Syvitski and Saito, 2007; Syvitski et al., 2009; Syvitski and Kettner, 2011; Bergillos et al., 2016).

Here we show that reservoir construction, as a representative example of grey infrastructure, has a potential positive impact on several ecosystem services such as flood and erosion control, and water regulation (Table 3). Nevertheless, past research has shown that grey infrastructure, like reservoirs and check dams, is often not a sustainable solution to address ecosystem services (Lorenzo-Lacruz et al., 2012; Di Baldassarre et al., 2018). The short-term effect of reservoir construction is an increase in irrigation water availability but with important socioeconomic and environmental trade-offs. This increased availability often increases the demand and the dependency on water stored in reservoirs (Di Baldassarre et al., 2018). Specifically, this dependency on water from reservoirs could cause water users to lose the notion of climate change, where reservoirs cause a delay in the impact of climate change on water availability (Lorenzo-Lacruz et al., 2012). In the Segura River catchment this is even more amplified by the inter-basin water transfer from the Tagus River (central Spain) (Lorenzo-Lacruz et al., 2010), which compensates for exceeding the water demand in the Segura River catchment by more than 200% (Sabater and Tockner, 2009). Additionally, when water from reservoirs is not available due to prolonged periods of drought or due to loss of capacity by sedimentation, the water demand will be satisfied with groundwater extractions, leading to significant aquifer depletion (Gleeson et al., 2012; Taylor et al., 2013), a process that is already unfolding in the study area (Rupérez-Moreno et al., 2017; Pellicer-Martínez and Martínez-Paz, 2018). In the long term, reservoir capacity is threatened by sediment trapping (Vörösmarty et al., 2003), affecting channel morphology (Vericat and Batalla, 2006) and river delta morphodynamics (Syvitski et al., 2009; Ibánez et al., 2019). Reservoirs also reduce the connectivity within the stream network, seriously affecting habitat conditions for stream ecology (Kondolf et al., 2006). Hence, the positive impacts are counterbalanced by negative impacts, such as an increase in water demand and habitat loss, which are more difficult to quantify, but are equally important.

The results from the Taibilla subcatchment demonstrate that a significant increase in green infrastructure (i.e. forest cover through reforestation) can have significant positive impacts on ecosystem services, such as erosion and flood control, quantified here by a decrease of hillslope erosion, sediment concentration and flood discharge (Fig. 8). Reforestation also leads to trade-offs in water supply and primary production ecosystem services, quantified here by decreased runoff and increased plant water stress, respectively, affecting downstream water availability. The negative impacts may be even stronger under future climate change, due to longer periods of drought and higher temperatures (García-Ruiz et al., 2011). Climate change will also affect the development of native Mediterranean vegetation and ecosystems (Bussotti et al., 2014; Riordan et al., 2015; Prieto and Querejeta, 2019; León-Sánchez et al., 2020), which urges to invest in drought and heat resistant species to be used in future reforestation projects.

We argue that integrated land-use planning and spatial optimisation of green infrastructures, such as sustainable land management (SLM) and nature-based solutions, are needed to mitigate the negative impact of climate change on ecosystem services (World Bank, 2008; Chabbi et al., 2017; Sanz et al., 2017). Where land use planning refers to changes in land use, SLM includes a range of technologies, policies and activities that aim to integrate biophysical, socio-cultural and economic needs and values to support the long-term productivity of ecosystems (Dumanski and Smyth, 1993; Schwilch et al., 2009). Plot-scale and laboratory experiments have demonstrated that SLM enhances on-site soil quality and, consequently, reduces runoff and soil erosion and increases water use efficiency and food production (Verhulst et al., 2010; Maetens et al., 2012; Delgado et al., 2013; Palm et al., 2014; Almagro et al., 2016). SLM also affects the off-site impact on ecosystem services indicators, such as a decrease in flood discharge and reservoir sediment yield (Azari et al., 2017; Eekhout and de Vente, 2019). Well-balanced implementation of spatial land use planning and application of SLM practices in collaboration with stakeholders is needed to optimise the delivery of ecosystem services.

A previous study in the Segura River catchment demonstrated that SLM applied in rainfed agriculture can offset the negative impact of future climate change on ecosystem services, positively affecting both on-site and off-site indicators (Eekhout and de Vente, 2019). But similar to reforestation, large-scale implementation of SLM also leads to a decrease in runoff and, subsequently, a decrease in downstream water availability. However, since green infrastructure increases water stored in the soil (green water), it also decreases the dependency on reservoir storage (blue water), arguably making the green infrastructures more sustainable in time. While a wide range of green infrastructures exist that can all have different impacts, we argue that due to the principal role of vegetation in green infrastructures similar effects can be expected on the interaction between green and blue water in Mediterranean environments. Furthermore, the benefits of green infrastructure over grey infrastructure and the possible trade-offs between up and downstream water availability supports the claim for integrated land-use management, in which balanced decisions are taken to optimize the spatial configuration of land use and management practices, considering interactions with future climate projections and expected land use change.

5. Conclusions

Here we present a study on the impacts of climate and land use change and reservoir construction on ecosystem services in a large Mediterranean catchment in the southeast of Spain. The changes that have occurred over the past 70 years can be considered typical for many Mediterranean catchments. Climate change in this period is characterized by a decrease in precipitation and an increase in temperature. Land use change is characterized by two opposing trends. Natural vegetation increased in the headwaters due to agricultural land abandonment and reforestation, while at the same time agriculture intensified in the central part of the catchment, which most likely is related to the construction of reservoirs and increased demand for food production. The downstream part of the catchment is characterized by urban expansion. While land use changed in more than 30% of the catchment, most impact on ecosystem services can be attributed to climate change and reservoir construction. This can in part also be explained by the contrasting impacts of greening-up and agricultural intensification at different locations in the catchment. Greening-up that dominated land use change in the headwaters resulted in a decrease in hillslope erosion, flood discharge and sediment concentration, but also in an increase in plant water stress, due to higher water demands of reforested areas as compared to rainfed agriculture. At the same time, agricultural intensification and urbanisation in the central and downstream parts of the catchment resulted in opposite impacts for most indicators.

All these changes had positive and negative impacts on ecosystem services relevant for Mediterranean environments, like water supply, and flood and erosion control. The positive impacts were quantified by a decrease in hillslope erosion, sediment yield, sediment concentration and flood discharge, which are all mainly attributed to climate change and reservoir construction. The negative impacts were quantified by an increase in plant water stress and a decrease in reservoir storage, for which the latter is mainly attributed to reforestation in the headwater catchments. The decrease in low flow caused by land use change was counteracted by an increase in low flow by reservoir construction. Grey infrastructure (i.e. reservoir construction) may have had a relatively positive impact on some of the ecosystem services considered here, however, it also increases the dependency on reservoir storage and may have attributed to an increase of irrigated agriculture, increasing the pressure on reservoir storage and groundwater resources. We argue that a shift is needed to green infrastructure, such as reforestation and sustainable land management, which may lead to similar benefits on ecosystem services, but without the negative impacts caused by grey infrastructure, such as habitat loss and an increase of water demand. Furthermore, green infrastructures decrease the dependency on reservoir storage. This supports the claim for integrated land-use management as strategy for climate change adaptation, optimizing the delivery of ecosystem services to society in Mediterranean environments.

CRediT authorship contribution statement

J.P.C. Eekhout: Conceptualization, Methodology, Software, Formal analysis, Writing - original draft, Visualization, Funding acquisition. **C. Boix-Fayos:** Conceptualization, Writing - review & editing, Funding acquisition. **P Pérez-Cutillas:** Resources, Writing - review & editing. **J. de Vente:** Conceptualization, Writing - review & editing, Supervision, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, athttps://doi.org/10.1016/j.jhydrol.2020.125208.

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