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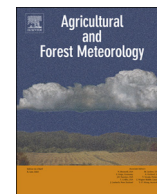
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Unraveling the impacts of droughts and agricultural intensification on the Altiplano water resources

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ABSTRACT

During the last decades, agriculture has drastically increased over the South American Andean Plateau (Altiplano), resulting in extensive changes in land cover from native vegetation to essentially Quinoa crop. Along with climatic variability, these land use changes appear as a catalyst in worsening the already existing drought events and water scarcity processes. Hence, understanding their relative contributions to the regional desertification process is crucial for sustainable water-use adaptation, but also is quite ambiguous because of water resource data scarcity over the Altiplano. Therefore, in the present study, an attempt to measure the impact of severe droughts and agricultural intensification on the water resources has been made using remote sensing datasets. The first step was dedicated to the validation of newly released CHIRPS v.2 precipitation and GLEAM v.3 potential evapotranspiration products by comparing their estimates with the results obtained from gauges data. Then, the Standardized Precipitation Index (SPI) was used to describe past hydro-meteorological drought events in terms of their spatial extent, duration, intensity and their impacts on the regional water resources. Finally, the dynamic trends in the spatial extent of the Quinoa crop and the meteorological conditions derived from CHIRPS v.2 and GLEAM v.3 were compared with the Vegetation Condition Index (VCI) and the Total Water Storage (TWS) derived from AVHRR and GRACE data respectively, to observe the respective influence of agriculture and climate variability on the regional hydrological system. A significant increase in Quinoa crop extent is observed from 2001 which corresponds to a significant decrease in regional VCI and TWS. Based on this trend, agriculture appears as a contributing factor in the water scarcity process over the Altiplano. The outcomes of this study will contribute to local decision making for a better water management and hydro-meteorological monitoring system.

1. Introduction

1.1. Pressures on water resources

Water resources are facing an unprecedented pressure due to the combined effects of population growth (Oelkers et al., 2011) and climate change (Wheater and Gober, 2015). During the 20th century, water extraction underwent a six-fold increase in order to fulfill food needs and to sustain economic levels of the growing world population (Cosgrove and Risberman, 2000). At the same time, the ongoing climate

change context that resulted in a temperature rise estimated to 0.85 °C by decades (IPCC, 2014) and a redistribution of precipitation favored the occurrence of drought events (Trenberth, 2011). Water shortages resulting from the combination of increasing water-extraction and climate change are widespread. Actually, in 2005, 35% of the world's population lived in areas with chronic water shortage (Oelkers et al., 2011) while 80% is exposed to water safety threat (Vörösmarty et al., 2010) and approximately 50% is likely to be living under severe water stress by 2071–2100 (Hanasaki et al., 2013). To address the growing need for water, water-policy improvement is urgently required to

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preserve both the environment and human life (Wheater and Gober, 2015).

1.2. The South American Andean Plateau situation

Water resources of the South American Andean Plateau region (Altiplano) are particularly sensitive to the previously described context. Between 1965 and 2012, the temperature increase was estimated between 0.15 and 0.25 °C per decade (Seiler et al., 2013; López-Moreno et al., 2015). It contributed to the regional snow and glaciers extent decrease by 43% from 1981 to 2014 period (Cook et al., 2016). Some glaciers already disappeared (Rabatel et al., 2013; Soruco et al., 2015) and others may disappear in a near future (Bradley et al., 2006). The consequences are already observable. As, for the first time, the city of La Paz, Bolivia's capital, had a restricted access to water during a five-month period (November 2016–March 2017) due to the rapid decrease in the water reservoirs related to the melting of glaciers. This situation is likely to worsen, as temperature is expected to continue to increase while precipitation may decrease by 10–30% depending on the climate projections (Bradley, 2006; Minvielle and Garreaud, 2011; Urrutia and Vuille, 2009).

Climate variability is not the only factor threatening Bolivian water security. The entire Altiplano region has gone through a drastic rise in targeted agricultural activity, such as Quinoa crop cultivation, over the last decades. While Quinoa price was multiplied by almost four, the area dedicated to Quinoa crop increased from 38,000 ha in 1980 to 70,000 ha in 2011 (Cusicanqui et al., 2013) and its production expanded from 34,000 to 84,000 tons between 2009 and 2014 (Canedo et al., 2016). In 2013, Peru and Bolivia jointly ensured more than 95% of the global Quinoa production (Canedo et al., 2016). The associated land cover change, from natural vegetation to cultivated land coupled with mechanization, shorter fallow times and poor irrigation methods, have contributed to occurrence of the regional desertification (Jacobsen, 2011, 2012). The complete drying of lake Poopó in December 2015 (i.e., second largest lake in Bolivia after lake Titicaca) is partially attributed to the agricultural activity through the evapotranspiration increase over cultivated areas (Satgé et al., 2017). The ongoing Altiplano desertification threatens agricultural activity and migrations from villages to cities are expected. Facing the combination of climate variability and human pressure, there is a need for comprehensive and reliable understanding of their relative contributions in the context of ongoing regional water scarcity.

1.3. Remote sensing datasets opportunity

The Altiplano is a transboundary watershed (Peru, Bolivia, Chile), having diverse economic and political factors that resulted in a deeply sparse and unevenly distributed hydro-climatological monitoring network. Days or years gaps in time series and hardware degradation introduce uncertainties in regional monitoring and seriously prevent accurate assessments of the impacts of both climate change and human pressure on the water resources. Canedo et al. (2016) already showed that important constraints for the Altiplano water resources assessment and agricultural development is the lack of data on climate and related water resources availability. In this context, with a global coverage in time and space, remote sensing techniques offer the opportunity to enrich traditional monitoring. The water balance controlling factors such as land cover, precipitation and evapotranspiration are intensively monitored through remote sensing. Land cover information can be retrieved from multispectral imagery such as Landsat (Hansen and Loveland, 2012) or MODIS (Friedl et al., 2010; Giri et al., 2005) for example; precipitation can be monitored from different satellite-based precipitation estimation products (SPPs) (Kidd, 2001; Maggioni et al., 2016) and evapotranspiration can be retrieved from various remote sensing datasets (Courault et al., 2005; Liou and Kar, 2014). Recently, the emergence of new long-term remote sensing products offers the

opportunity of consistent overlapping of these water balance factors, over the last 35 years. For example, the Climate Hazards Group InfraRed Precipitation with Station version 2 (CHIRPS v.2), provides precipitation estimates at spatial resolution of 0.05° (5 km) for the 1981 to near-present period (Funk et al., 2015). Evapotranspiration estimates spanning from 1980 to 2016 are available from GLEAM version 3.1 database (GLEAM v.3) (Martens et al., 2017) and, for land cover monitoring, the AVHRR Vegetation Health Product (AVHRR-VHP) released by NOAA STAR offers the opportunity to follow vegetation evolution from 1981 to near-present at 4 km spatial resolution. However, remote sensing data are indirect measurements made from satellite/sensors constellations, and are subjected to uncertainty due to the technical limitations. Many studies have already reported on the benefits/limits of remotely sensed precipitation (Maggioni et al., 2016; Sun et al., 2017) and evapotranspiration (Liou and Kar, 2014) datasets in different regions. As their reliability are subjected to significant variations in space and time, results obtained over one specific region and/or upon a specific temporal window cannot be extrapolated to the global scale (Satgé et al., 2019). Up to date no study has reported on CHIRPS v.2 and GLEAM v.3 reliability over the entire South American Andean plateau which is preventing their utilization in the regional water resource management.

1.4. Objectives

The present study aims at contributing to the understanding of the regional Andean plateau water scarcity process through different complementary steps: (i) the calibration/validation of remote sensing datasets (CHIRPS v.2 and GLEAM v.3) to support traditional precipitation and evapotranspiration measurements; (ii) the description and analysis of past meteorological drought events and their impacts on the Total Water Storage (TWS) for the 1981–2016 period and; (iii) the investigation of Quinoa crop cultivation influences on TWS. Beyond its usefulness for supporting better local decision regarding water management in the Altiplano region, the study employs tools and methods generic enough to be applied in other regions in the world where similar trends are observed (Wurtsbaugh et al., 2017).

2. Materials

2.1. Study area

The Andean Altiplano plateau is an endorheic region located in the South American high land between latitude 22°S and 14°S and longitude 71°W and 66°W (Fig. 1). The regional watershed is located at a 4000 m average altitude a.s.l. and delimited at east and west by the oriental and occidental cordilleras, respectively (Satgé et al., 2016). It is called TDPS system in relation to the specific system including the Lake Titicaca, the Desaguadero River, the lake Poopó and the two salt deserts of Coipasa and Uyuni with total area of approximately 192,390 km². The Lake Titicaca is drained by the Desaguadero River in the south and it contributes up to 65% of Lake Poopó's water input (Pillco and Bengtsson, 2006). The Lake Poopó is drained by the Lakajawira river to the Coipasa Salar. But, in the last 50 years, this river only flooded once in 1986. Consequently, the Poopó Lake can be reasonably considered as the terminal point of the endorheic Altiplano system (Pillco and Bengtsson, 2006). In 2009, around 3 million persons habit the Andean Altiplano, mainly indigenous people with low incomes, with only 20% of the population having access to water and sanitation services. The economic activity is articulated around agriculture, industry and trade, with Quinoa and potatoes being the most important crops (Canedo et al., 2016). Because of manual labor, poor soil and a regional climate context, characterized by high evaporative demands and low precipitation amounts (estimated to 1280 and 405 mm year⁻¹, respectively, over the 1981–2016 period using GLEAM v.3 and CHIRPS v.2), agricultural productivity is very sensitive to climate variability.

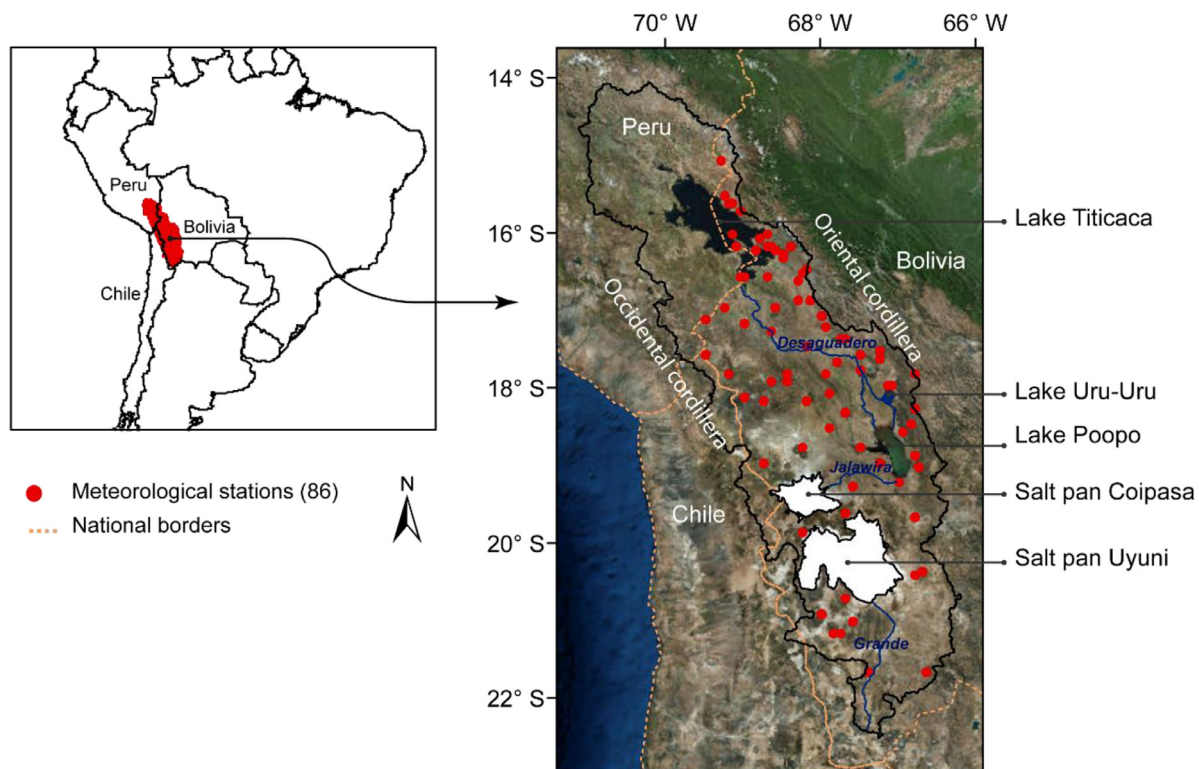


Fig. 1. Location of the Altiplano in South America (left). In this drainage area, major lakes and salt pans appear in blue and white respectively (right). The location of the meteorological stations from SENAMHI is represented using red dots.

2.2. Reference data

The Servicio Nacional de Meteorología y Hidrología (SENAMHI) oversees the Bolivian national hydro-meteorological network. In this study, data from 154 meteorological stations equipped with precipitation gauges and the required climatic sensors (temperature, air humidity and wind) for potential evapotranspiration computation for the 1960–2012 period were made available by the Bolivian SENAMHI. Meteorological stations were not available over the Peruvian region due to geopolitical tensions on the Lake Titicaca water management.

Additionally, the area dedicated to Quinoa crop was obtained from the Bolivian national institute of statistics between 1980 and 2016. It is noteworthy that the values used in this study refer to the annual superficial extent dedicated to Quinoa crop over the Bolivian territory and not only over the Altiplano. However, as most of Quinoa crops are located over the Altiplano, the temporal series dynamic is expected to be representative of the Altiplano.

2.3. Remote sensing data

2.3.1. Precipitation estimates from CHIRPS v.2

Climate Hazards Group InfraRed Precipitation with Station version 2 (CHIRPS v.2) is a product developed by the U.S. Geological Survey (USGS) and the Climate Hazards Group (CHG) (Funk et al., 2015). In a first step, precipitation estimates are derived from global Cold Cloud Duration (CCD) retrieved from the Globally Gridded Satellite (GriSat) and NOAA Climate Prediction Center (CPC) thermal infra-red dataset using a fixed threshold of 235 K. Additionally, CDD estimates are calibrated over the 2000–2013 period using the Tropical Rainfall Measuring Mission (TRMM) Multisatellite Precipitation Analysis 3B42 (TMPA-3B42) gauges adjusted version 7 (Huffman et al., 2010). In a second step, rain gauges data from world meteorological organization including the Global Historical Climate Network (GHCN), the Global Summary of the Day (GSOD), the Global Telecommunication System (GTS) and Southern African Science Service Centre for Climate Change

and Adaptive Land Management (SASSCAL) jointly with data from Mexico, Central and South America, and Sub Saharan Africa from national agencies are blended with CDD precipitation estimates, using a modified inverse distance weighting algorithm that enhances precipitation estimates accuracy. For more information, please consult (Funk et al., 2015). Data are available on daily, 10-day and monthly time step at a near global scale covering 50°N–50°S. Among all the Satellite-based Precipitation Estimates (SPPs) covering the last 35 years, CHIRPS v.2 presents the finer spatial resolution grid-cells of 0.05°. Thus, CHIRPS v.2 is of great interest to follow precipitation variation and ongoing effects on water resources and agricultural activities. In this context, CHIRPS v.2 reliability has already been reported in Brazil (Paredes-Trejo et al., 2017), Chile (Zambrano-Bigiarini et al., 2017), Africa (Dinku et al., 2018) and China (Bai et al., 2018). Even if all these studies unanimously reported an overall acceptable accuracy of CHIRPS v.2 estimates, a site-specific assessment is still needed before their use in catchment-scale hydrological studies (Zambrano-Bigiarini et al., 2017).

2.3.2. Evapotranspiration estimates from GLEAM v.3

The Global Land Evaporation Amsterdam Model (GLEAM) is a set of algorithms for the estimation of evaporation components from satellite observations. Potential evapotranspiration (ETp) is retrieved from the Priestley and Taylor equation based on surface net radiation and near-surface air temperature. Then, root-zone soil moisture and Vegetation Optical Depth (VOD) are used to apply a multiplicative evaporative stress factor on ETp to estimate Actual Evaporation (Eta). Recently, the third GLEAM version (GLEAM v.3) was made available with three datasets (GLEAM v3.1a, b and c) (Martens et al., 2017). These datasets differ in terms of spatial coverage, spatial resolution and forcing data. According to the study objectives, the dataset GLEAM v3.1a with a global coverage and a 0.25° mesh size was selected as it presents the highest spanning time among these three datasets (37 years from 1980 to 2016). Data are available on daily time step (<https://www.gleam.eu/>). GLEAM v.3 evaporation validation led over two validation sites

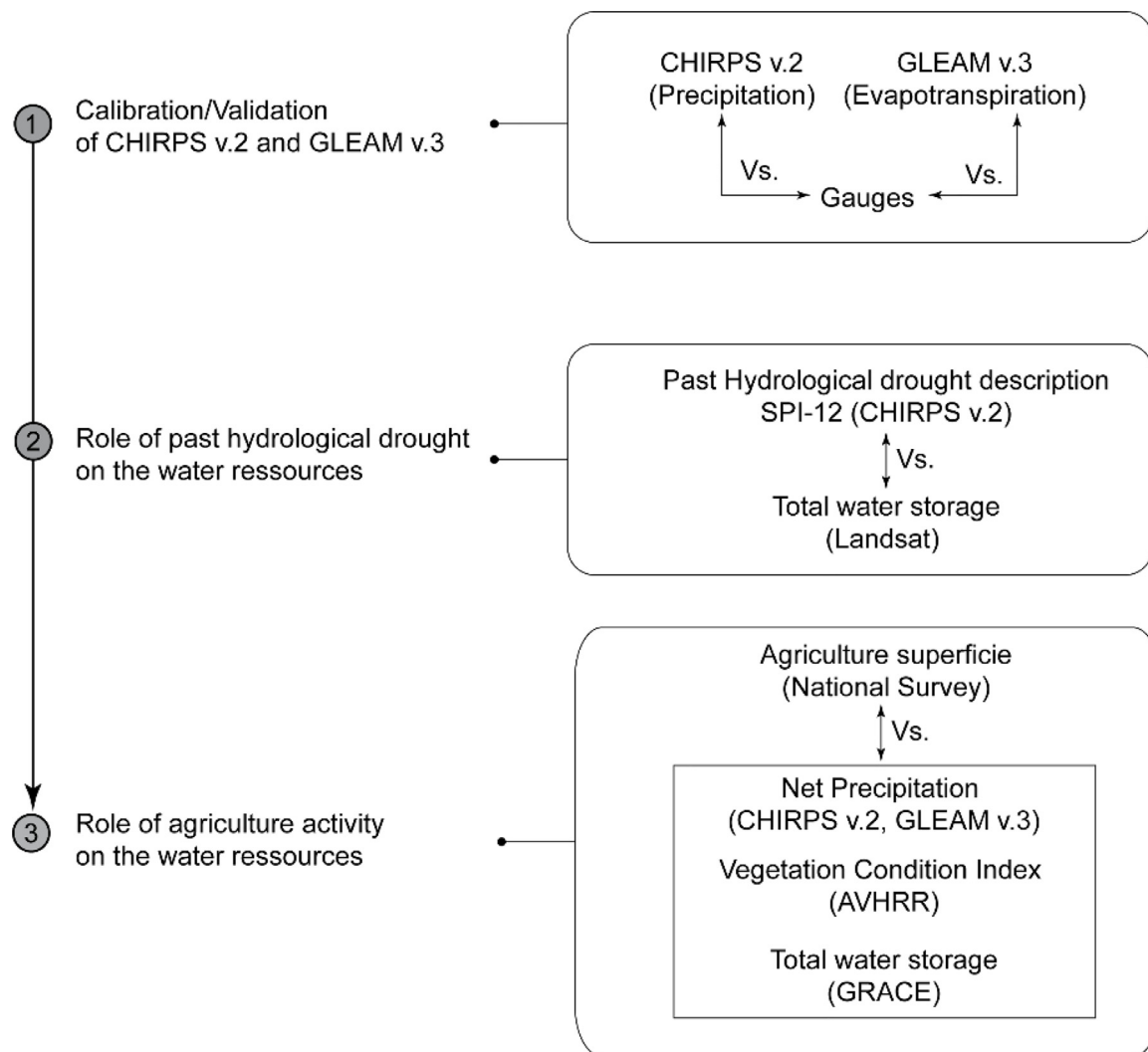


Fig. 2. Flowchart of the main study steps including: (1) the remote sensing datasets (CHIRPS v.2, GLEAM v.3) calibration/validation; (2) the identification/characterization of past hydrometeorological droughts and assessment of their impact on the regional water resources; (3) the investigation of agricultural activity influence on the regional water resources.

located in central CONUS and Australia, showed realistic evaporation estimates with no systematic errors (Martens et al., 2017). Due to the existence of limited validation sites, more validation attempts are still needed to state on GLEAM's potential utilities.

2.3.3. Vegetation Condition Index (VCI) from AVHRR

The Vegetation Condition Index (VCI) is used to observe the potential impact of drought on the Altiplano region (Eq. (1)). VCI is a normalization of the Normalized Difference Vegetation Index (NDVI) which enhances the weather component in NDVI, variations of which are mainly controlled by the factors of ecosystem (e.g., climate, soils, vegetation type and topography) (Kogan et al., 2003).

$$VCI = \frac{NDVI - NDVI_{min}}{NDVI_{max} - NDVI_{min}} \quad (1)$$

where $NDVI_{max}$ and $NDVI_{min}$ are the maximum and minimum values of a chosen NDVI over a specific period (e.g. day, week, or month). VCI's suitability in monitoring drought intensity, duration, dynamics and impact on vegetation in different regions has already been shown (Du et al., 2013; Klisch and Atzberger, 2016; Winkler et al., 2017; Zambrano et al., 2016; Zhang et al., 2017). In this study, VCI derived from the Vegetation Health Index (VHI) collected from Advanced Very High Resolution Radiometer (AVHRR) imagery is used. Data are available

online by NOAA at: <https://www.star.nesdis.noaa.gov/smcd/emb/vci/VH/vhftp.php>. Data are delivered in GEO-TIFF format at 4 km spatial resolution grid-cells at weekly time step. The VCI values were first averaged to obtain monthly time series over the period of 1982–2017.

2.3.4. Total Water Storage from GRACE

The Gravity Recovered and Climate Experiment (GRACE) is a joint mission between NASA and the German Space Agency (DLR) launched in 2002. GRACE consists in two satellites on a polar orbit separated by 220 km at an altitude of ~500 km. The distance variation between both satellites computed through the electromagnetic link in the K/K_a band is used to detect variations in Earth's gravity field. These variations are related to the movements of mass induced by the hydrological cycle, ice-loss, glacial isostatic adjustment and earthquakes (Tapley et al., 2004). GRACE data successfully helped to monitor TWS variations related to climate variability and/or human activity (Deng and Chen, 2017; Frappart et al., 2013; Khandu et al., 2016; Ndehedehe et al., 2017; Yang et al., 2015). GRACE-based TWS are commonly used to monitor groundwater storage variations (see Frappart and Ramilien, 2018 for a recent review). GRACE TWS was recently compared against groundwater volume variations using 448 wells observations over a similar arid region in Iran (Zaki et al., 2019). The authors conclude that GRACE data can be used for water resources management providing an

opportunity in data scarce regions. In this study, the monthly TWS grids from Global JPL RL06 mascons were used from August 2002 to March 2016. The data has a spatial resolution of $3^\circ \times 3^\circ$ but resampled at $0.5^\circ \times 0.5^\circ$. Compared to classical global GRACE solutions, noise is reduced applying spatial and temporal constraints during processing. For JPL mascons, spatial constraints are based on both GRACE data and geophysical models (hydrological and oceanic models) and temporal constraint is based on the application of a temporal correlation using a sequential Kalman filter with stationary process noise to account for trends in the data that are not included in the models (Wiese et al., 2016).

2.3.5. Total Water Storage “proxy” from Landsat imagery

According to hydrological dynamics, the lake Poopó can be considered as the terminal point of the endorehic Altiplano system (Pilco and Bengtsson, 2006). According to the linear relationship existing between its superficial extent and water level (Arsen et al., 2013), we proposed to use Lake Poopó's superficial extent as a “proxy” for the Altiplano TWS fluctuations. With a temporal coverage from 1984 to present, Landsat imagery offers an opportunity to follow the Lake Poopó extent over the last decades. Therefore, Lake Poopó's superficial extent was derived from Landsat imagery by using the atmospheric correction model (FLAASH) and water extraction index (WRI) previously validated over the region (Satgé et al., 2017). In order to avoid any cloud contamination in Landsat imagery, we only considered the images recorded at the end of the dry season. A total of 31 cloud free images were used to retrieve its superficial extent over the 1985–2015 period.

3. Methods

Fig. 2 is a flowchart of the main methodological part. First, precipitation (P) and ETp estimates derived from CHIRPS v.2 and GLEAM v.3 are compared against gauge measurements at the regional and point gauge scales. Secondly, CHIRPS v.2 is used to compute the Standardized Precipitation Index (SPI) that describes past hydrological drought events. Finally, the net precipitation (P_{net}), VCI and the TWS trends are confronted with the agriculture extent evolution in order to discuss the potential role of agricultural activity on the regional water resources. More details of the different parts are presented in the following sections.

3.1. Remote sensing data validation

Meteorological data quality was checked discarding non-consistent stations according to the Regional Vector Method (RVM) (Espinoza Villar et al., 2009; Rau et al., 2017; Satgé et al., 2016). RVM assumes that some consistency exists in the precipitation inter-annual variability for station located on determined climatic zone as they belong to the same precipitation regime. We divided the region into eight climatic zones and applied the RVM each zone (Satgé et al., 2016). A total of 159 stations with at least five continuous observation years for the 1960–2012 period were considered. From the analysis, only 123 stations were consistent at the inter-annual time step. The RVM was then applied for the 123 remaining stations at the monthly time step. From the analysis only 86 stations with data for the 1982–2012 periods were consistent. It is worth mentioning that only precipitation amounts were considered in the applied RVM quality check approach. It is assumed that the inconsistency found on precipitation measurement may spread to the other meteorological parameters of the considered stations. Additionally, the records were manually checked to identify potential outlier's values.

The 86 meteorological stations were spread into seventy-seven 0.05° CHIRPS v.2 grid-cells and sixty 0.25° GLEAM v.3 grid-cells, respectively. The precipitation (P_{ref}) and potential evapotranspiration ($ET_{p_{ref}}$) reference series were obtained by averaging the value derived

from all included stations in each grid-cell. To insure consistency in the analysis, only grid-cells for which the corresponding P_{ref} ($ET_{p_{ref}}$) accounted for more than 80% of monthly estimates were used to assess CHIRPS v.2 and GLEAM v.3 reliability. This corresponds to a total of 22 and 13 grid-cells for CHIRPS v.2 and GLEAM v.3, respectively.

Mean spatially-averaged monthly precipitation (CHIRPS v.2 and P_{ref}) and ETp (GLEAM v.3 and $ET_{p_{ref}}$) were obtained by aggregating the values from all the twenty-two 0.05° and thirteen 0.25° grid-cells, respectively. The FAO Penman-Monteith equation was used to retrieve $ET_{p_{ref}}$ as it performed better than Thornthwaite and Hargreaves-Samani equation when compared to lysimeters measurement (Garcia et al., 2004).

Remote sensing regional average estimates (CHIRPS v.2, GLEAM v.3) were compared to the corresponding mean regional estimates (P_{ref} , $ET_{p_{ref}}$) in terms of correlation coefficient (CC), percentage bias (%B), and root mean square error (RMSE) normalized by the standard deviation of the reference (gauges) (called NRMSE here after). CHIRPS v.2 and GLEAM v.3 estimates error's spatial distributions were assessed comparing CC, NRMSE and %B obtained at each grid-cell location separately.

Statistical scores are interpreted in terms of qualitative rating to state upon CHIRPS v.2's and GLEAM v.3's reliability, with threshold values of NRMSE, %B and CC adapted from the literature (Hussain et al., 2017; Moriasi et al., 2007; Satgé et al., 2016; Shrestha et al., 2017) (Table 2).

3.2. Hydrological drought events and impacts on the regional water resources

As recommended by the World Meteorological Organization (WMO), the Standardized Precipitation Index (SPI) (McKee et al., 1993) should be considered as a standard drought index for representing meteorological drought conditions induced by precipitation deficit (WMO, 2012). A minimum precipitation record of 30 years is advisable to compute a confident SPI (WMO, 2012). The long term precipitation record is first fitted to a gamma probability distribution and secondly transformed into a normal distribution (Almedeij, 2014; Edwards and McKee, 1997; McKee et al., 1993). Because the SPI is normalized, wet and dry climates can be depicted in the same way. Indeed, SPI presents a zero-mean value with negative and positive values indicating dry and wet periods, respectively, with corresponding intensity classes to describe them (Table 2).

SPI can be computed at any timescale but usually it is computed for 1, 3, 6, 12 and 24 months (SPI-1, -3, -6, -12, -24). The choice of time-scale is subjective and related to the objectives to be achieved. According to the WMO, meteorological and agricultural drought can be described using SP-1, and SP-1 to SP-6 respectively, whereas SPI-6 and SPI-12 are advisable to depict hydrological drought that impacts superficial and groundwater resources. However, SPI-6 is more suited for seasonal trend observation and SPI-12 for stream flows, reservoir levels, and groundwater levels (WMO, 2012). Over arid regions, the use of SPI-3 or less is not recommended as large negative or positive SPI may occur for precipitation close to the historical mean and not because of dry or wet events. To take into account (i) the typical arid context of the Altiplano and (ii) the main objective of the study, which is to observe the hydrological drought influence over TWS, we only considered SPI-12. The 12-month SPI provides a comparison of the precipitation over a specific 12-month period with the precipitation totals from the same 12-month period for all years included in the historical record (WMO, 2012).

According to (Guo et al., 2017), a drought event is characterized by a period with at least two consecutive months of negative SPI values and at least one month with SPI lower than -1 . Additionally, an identified drought can be characterized by its duration, severity and intensity. The drought duration (DD) corresponds to the number of months with negative SPI value whereas the total drought duration

(TDD) is the number of months with SPI value lower than -1 . Drought severity (DS) corresponds to the absolute sum of monthly SPI values during the drought. The drought intensity (DI) is divided into two kinds, DI1 and DI2. DI1 corresponds to the lowest SPI value recorded during the drought and DI2 is the ratio between DD and DS.

In a first step, SPI-12 was derived at the regional scale from 1982 to 2016 to identify in time the previous hydrological droughts. The identified hydrological droughts were compared in terms of DD, DS, DI1 and DI2. In a second step, we computed the percentage areas affected by moderate, severe and extreme hydrological drought conditions (Table 2) in order to assess hydrological drought spatial extents and dynamics. To do so, the ratio between (i) the number of grid-cells recording a moderate, severe and extreme hydrological drought conditions and (ii) the total number of grid-cells covering the region were computed for each month.

3.3. Agriculture and water scarcity process

From the chronological series of agricultural superficies, a drastic increase of Quinoa crop extent is observed from 2001 onward. This year was used to split the last 35 years into two periods: during the first period from 1982 to 2001 (denoted P1) the Quinoa crop area increases slightly, while it expands drastically during the second period from 2002 to 2016 (P2). During P2, the extent of Quinoa crop was multiplied by three, passing from 43,140 to 118,910 ha. The opportunity of social economic enhancement led local people to intensify their Quinoa production. Aside from converting natural vegetation to Quinoa crops, improved yields were achieved by a transition from traditional manual to mechanized agricultural methods, shorter fallow times and irrigation. The Mann–Kendall test (MK) (Burn and Hag Elnur, 2002; Mann, 1945) was applied to VCI, and TWS data for both P1 and P2 periods to quantify the influence of Quinoa crop intensification on vegetation and water resources. Additionally, the MK test was also applied to the regional net precipitation (P_{net}), to verify any trend in meteorological conditions during both P1 and P2 periods. P_{net} was computed as the difference between monthly spatial average precipitation and ETp estimates (Lee et al., 2014) derived from CHIRPS v.2 and GLEAM v.3, respectively. It is worth mentioning that the MK test was performed using the mean monthly value for each hydrological year (November–October).

4. Results and discussions

4.1. CHIRPS v.2 precipitation and GLEAM v.3 evapotranspiration assessment

Fig. 3a shows the comparison between spatially averaged monthly precipitation estimates derived from CHIRPS v.2 and the gauges in the form of scatter plot. With a CC, NRMSE and %B of 0.96, 0.29 and 3.91%, respectively (Fig. 3a), CHIRPS v.2 statistic scores correspond to an overall very good performance rating (Table 1). Therefore, CHIRPS v.2 is suitable to represent the spatially average precipitation estimates over the South American Andean plateau.

Looking at CHIRPS v.2's spatial error distribution (Fig. 3b–d), it remains well correlated to gauge data all over the region, with CC systematically superior to 0.7. Additionally, most of the grid-cells have %B between -30 to 30% which correspond to a good performance rating and only two grid-cells present unsatisfactory NRMSE score (>0.7). According to the results, CHIRPS v.2 can be used at the regional scale (Fig. 2a) to observe precipitation variability with a very good degree of confidence and at the local scale with an overall satisfactory to good degree of confidence (Fig. 3b–d).

Fig. 3e shows the comparison between the spatially averaged ETp estimates derived from GLEAM v.3 and the gauges in the form of scatter plot. A systematic negative bias of approximately $40 \text{ mm} \cdot \text{month}^{-1}$ was observed on GLEAM v.3's estimates. This bias was previously corrected

for each grid-cell and only results obtained after the bias correction are presented (Fig. 2e–h). With CC, NRMSE and %B values of 0.83, 0.66 and 0% after bias correction, respectively, the GLEAM v.3 statistic scores correspond to a good (CC, %B) to satisfactory (NRMSE) performance rating (Table 1). Therefore, GLEAM v.3 is suitable to represent the spatially average ETp estimates over the Altiplano.

Looking at GLEAM v.3's spatial error distribution (Fig. 3f–h) all the considered grid-cells present unsatisfactory NRMSE (>0.7). Therefore, GLEAM v.3 should not be used at the grid-cell level to follow local ETp dynamics. In this study, GLEAM v.3 is only used to represent the spatially average ETp estimates.

4.2. Past hydrological droughts description and impact over the water resources

Seven hydrological droughts occurred over the Andean Altiplano region (Fig. 4a, b). These droughts differed in terms of DD, TDD, DS, DI1 and DI2 (Table 3). The most severe hydrological drought (D1) occurred from December 1982 to January 1984 with the lowest and highest DI1 and DI2 values. During D1 drought, almost all the area (90%) was affected by extreme to severe dry conditions (Fig. 4c).

The two following droughts (D2, D3) occurred from April 1987 to February 1991 and from October 1991 to October 1993, respectively. They represented a continuous drought period as they were separated by only seven months during the regional dry season (March–September) with very low positive SPI values (Fig. 4a). During D2 and D3 almost 40% and 45% of the area was covered by extremely severe dry conditions, respectively.

Finally, the last four drought events (D4–D7) were less severe in terms of DI1 and DI2. During the last four drought events, there was a dominance of moderate dry conditions and no region was affected by the extreme dry conditions (Fig. 4c).

On a general point of view, drought strength and spatial extent are diminishing from D1 to D7 with decreasing TDD, DI1 and DI2 values and a relative percentage diminution of the area affected by the severe to extreme dry conditions.

Fig. 8 shows Lake Poopó's surface extent at the end of dry season during the 1985–2015 period. The lake totally dried up in 1995, 1996 and 2015 and reached very critical levels in 1994, 1998, 2008, 2009 and 2010 (Fig. 4d) (Satgé et al., 2017). These events are related to the hydrological droughts. Indeed, the 1995 and 1996 lake droughts correspond to D4 while the very critical lake extents of 1998, 2008, 2009 and 2010 correspond to D5 and D7. This confirms the influence of hydrological drought episodes on the regional water resources (Fig. 4d). It is worth mentioning that, even if D2 and D3 do not correspond to a total lake Poopó's drought, they clearly contribute to the steady lake extent decrease registered from 1987 to 1994 (Fig. 4d). During this seven-year period, the lake superficial extent passed from 3166 km^2 (1988) to 894 km^2 (1994). This trend can obviously contribute to the two successive lake Poopó droughts in 1995 and 1996. The largest lake Poopó extent was registered in 1986 (3302 km^2) as a consequence of the very high wet period which occurred from February 1984 to March 1986 (26 months, Fig. 4a, b). This wet episode was the highest wet event ever registered over the last 35 years and should have considerably contributed to recharge all hydrological components of the Altiplano (superficial and groundwater). Unfortunately, no Landsat images were available for the D1 period in order to check its influences on Lake Poopó. The D6 drought did not significantly affect the lake Poopó extent, as only a slight depletion in the lake Poopó extent was registered in 2005. This might be related to the D6 very low DS, DI1 and DI2 values in comparison with others drought events (Fig. 4a). Interestingly noticing that, the total drought which occurred to Lake Poopó in 2015 seems to be independent to the precipitation variability, as no drought events was registered for this specific period. Therefore, it suggests that other factors may have played a role in this last lake Poopó drought.

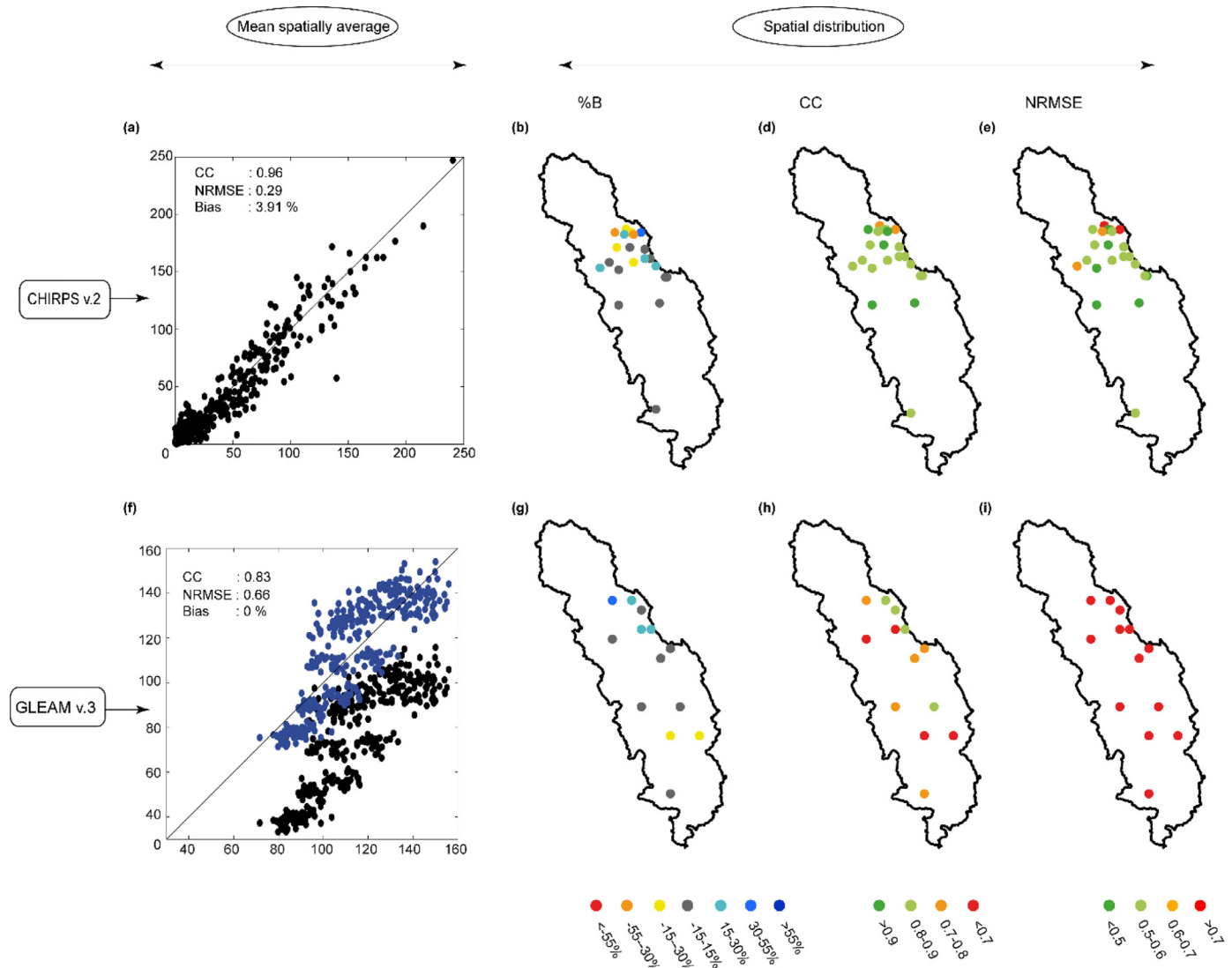


Fig. 3. Efficiency of CHIRPS v.2 and GLEAM v.3 compared with meteorological stations: (a, e) Consistency of spatially average CHIRPS v.2 and GLEAM v.3 estimates compared in the form of scatter plot; Consistency in space of CHIRPS (b, c, d) and GLEAM v.3 (f, g, h) expressed in form of %B, CC and NRMSE.

Table 1

Quality rating for CHIRPS v.2 and GLEAM v.3 assessment.

Performance rating	%B	NRMSE	CC
Very good	< ± 15%	< 0.5	> 0.9
Good	± 15%–± 30%	0.5–0.6	0.8–0.9
Satisfactory	± 30%–± 50%	0.6–0.7	0.7–0.8
Unsatisfactory	> ± 50%	> 0.7	< 0.7

Table 2

SPI classes.

Class	SPI value
Extremely wet	≥ 2
Severely wet	$1.5 \leq \text{SPI} < 2$
Moderately wet	$1 \leq \text{SPI} < 1.5$
Near-normal	$-1 < \text{SPI} < 1$
Moderately dry	$-1.5 < \text{SPI} \leq -1$
Severely dry	$-2 < \text{SPI} \leq -1.5$
Extremely dry	≤ -2

4.3. Agriculture impact over the water resources

Fig. 5 shows the temporal evolution of P_{net} , VCI, TWS and Quinoa crop extent from 1982 to 2016 with different colors being used in order to separate P1 and P2. While considering VCI, for both P1 and P2 periods (i.e. before and after the “Quinoa boom”), two distinct trends are observed. VCI increased during P1, while it decreased during P2. According to MK test, both trends are significant with MK-P value lower than 0.05 (Table 3). The VCI decay during P2 could be a direct consequence of the agricultural development and crop encroachment. High-altitude ecosystems are among the most fragile of the Earth with low recovery rates after disturbance (Rolando et al., 2017). Therefore, over the Altiplano, Barrientos et al. (2017) have estimated that once a piece of land is used for agricultural purpose, more than ten years are required for native vegetation return to its initial state. In the Altiplano region, with the fallow period reduction from 2 to 6 years to 1 or 2 years (Jacobsen, 2011), there is not enough time for native's vegetation recovery over croplands. Consequently, after the first harvest, the land is turned into bare-land, which could explain the observed VCI decay. Additionally, due to the fallow period reduction, fallow lands are not suitable for cattle grazing, this induces overgrazing of native vegetation land (Rolando et al., 2017) which could also contribute to the observed VCI decay.

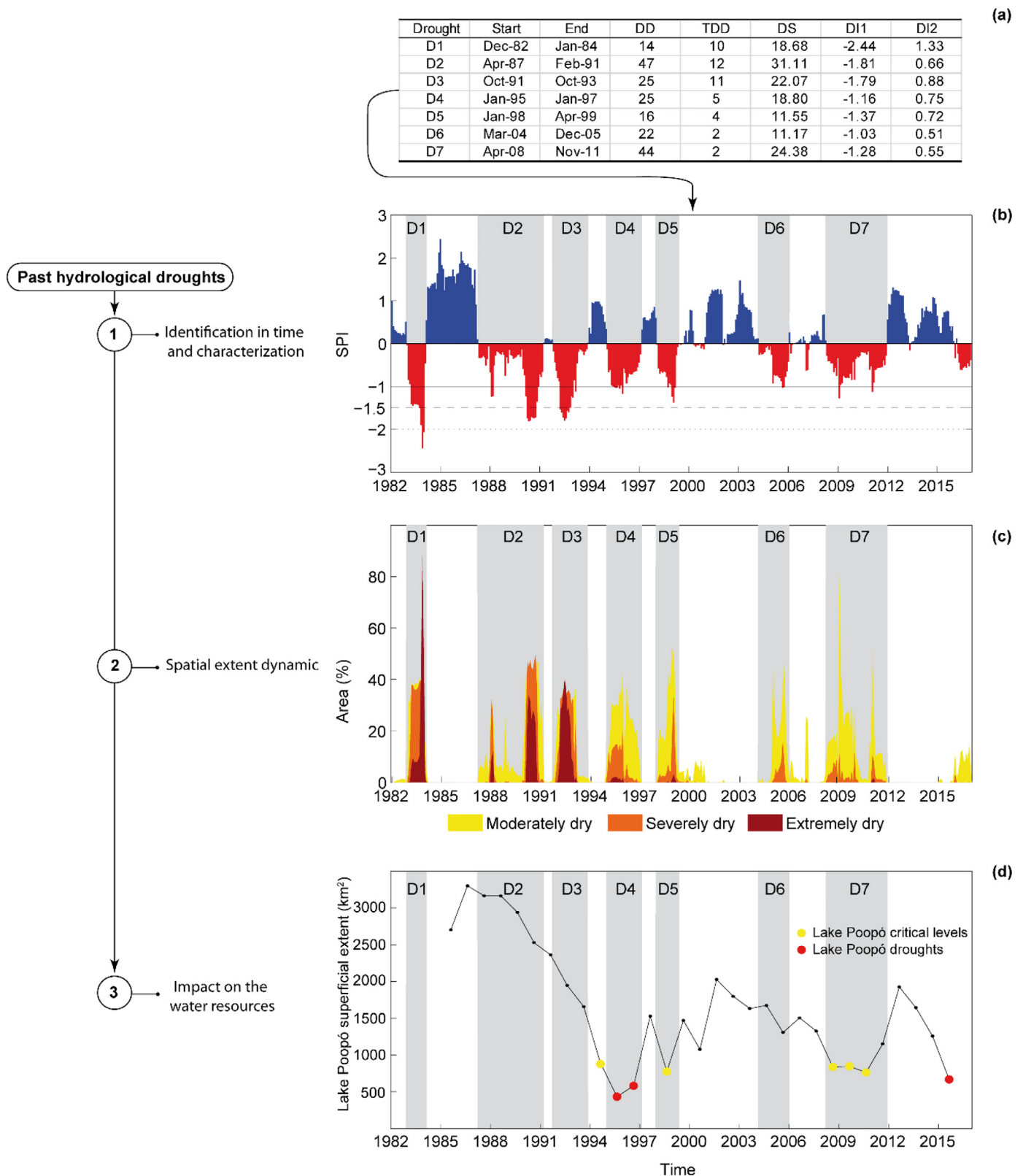


Fig. 4. Past hydrological droughts description and their influence on total water resources: (a, b) detection and characterization; (c) hydrological drought spatial dynamics over time; (d) hydrological drought influence on the Lake Poopó extent (TWS proxy).

During the P2 period, TWS also significantly decreased (Fig. 5, Table 3). Quinoa crop being rainfed agriculture, its growing cycle mobilizes direct precipitation, which reduces groundwater recharge from precipitation. Moreover, even if Quinoa is tolerant to dry

conditions, two annual precipitation amounts are required (Jacobsen, 2011; Winkel et al., 2016). Indeed, the year before the planting, the land is plowed to facilitate precipitation storage to support water deficit during the following crop's year (Giuliani et al., 2012; Winkel et al.,

Table 3
Trend and MK-P value for VCI, SPI, ETp and TWS and for P1 and P2 periods.

	P1 Trend	MK-P value	P2 Trend	MK-P value
P_{net}	-0.07	0.77	0.13	0.37
VCI	0.93	<0.05	-1.42	<0.05
TWS (GRACE)	-	-	-0.72	<0.05

2016). Regarding to the large cultivated area and the low regional annual precipitation amount, this practice should obviously reduce aquifer's recharge by precipitation input and can partially explain the TWS storage decrease. Moreover, the agriculture intensification and the crop encroachment over the native vegetation contribute to actual evapotranspiration increase (Satgé et al., 2017) supporting the observed TWS decay. Over the region, climatological conditions, represented in the form of P_{net} , remain stable with no significant trends

revealed by MKP value over both P1 and P2 periods. Therefore, Quinoa agriculture appears as a potential factor contributing to the water scarcity process in terms of vegetation and water availability as shown by VCI and TWS evolution.

However, even if no monotonic trends are observed in the meteorological conditions (P_{net}), the successive highlighted droughts (D1–D7) also contribute to the regional water resources scarcity process (see Section 3.3). Indeed, during both P1 and P2 periods the lake Poopó's extent faces significant decrease (MK-P value < 0.05). Even if Lake Poopó extent cannot be directly compared to TWS derived from GRACE, a strong relationship is expected between Lake Poopó's extent and TWS fluctuation. Therefore, the TWS decrease during P2 cannot be attributed only to the regional agricultural practices.

It is worth mentioning that the 2015 Lake Poopó's complete drying is independent from any drought event. Therefore, the TWS reach a tipping point where “normal” climatic conditions hardly compensate for agricultural activity impacts. In this context, research on the deficit

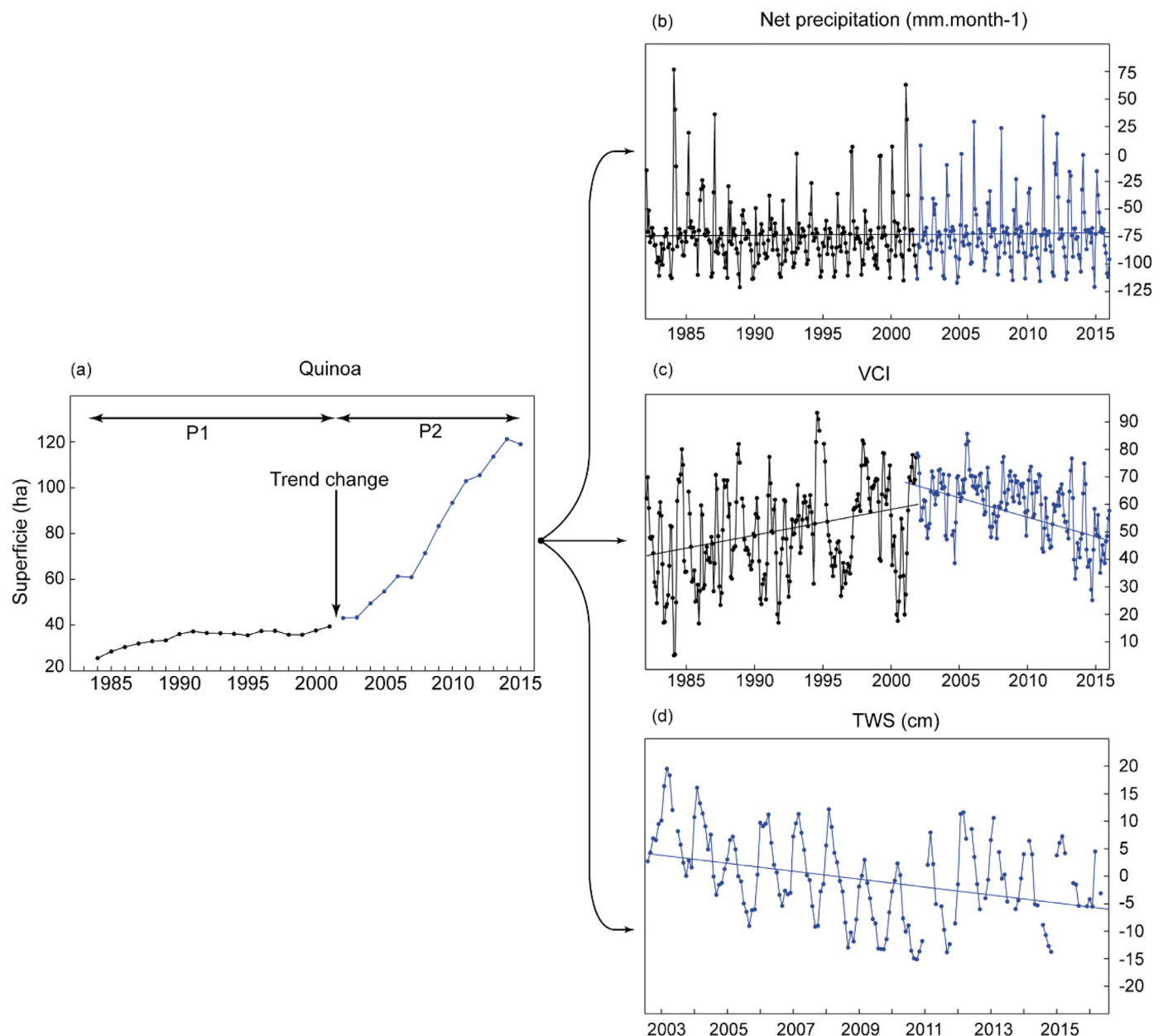


Fig. 5. Agricultural activity and total water resources: (a) temporal change in Quinoa cultivated area; (b) net precipitation; (c) VCI evolution and; (d) TWS derived from GRACE. Black and blue colors are used to differentiate 1982–2001 period from 2002 to 2016 period.

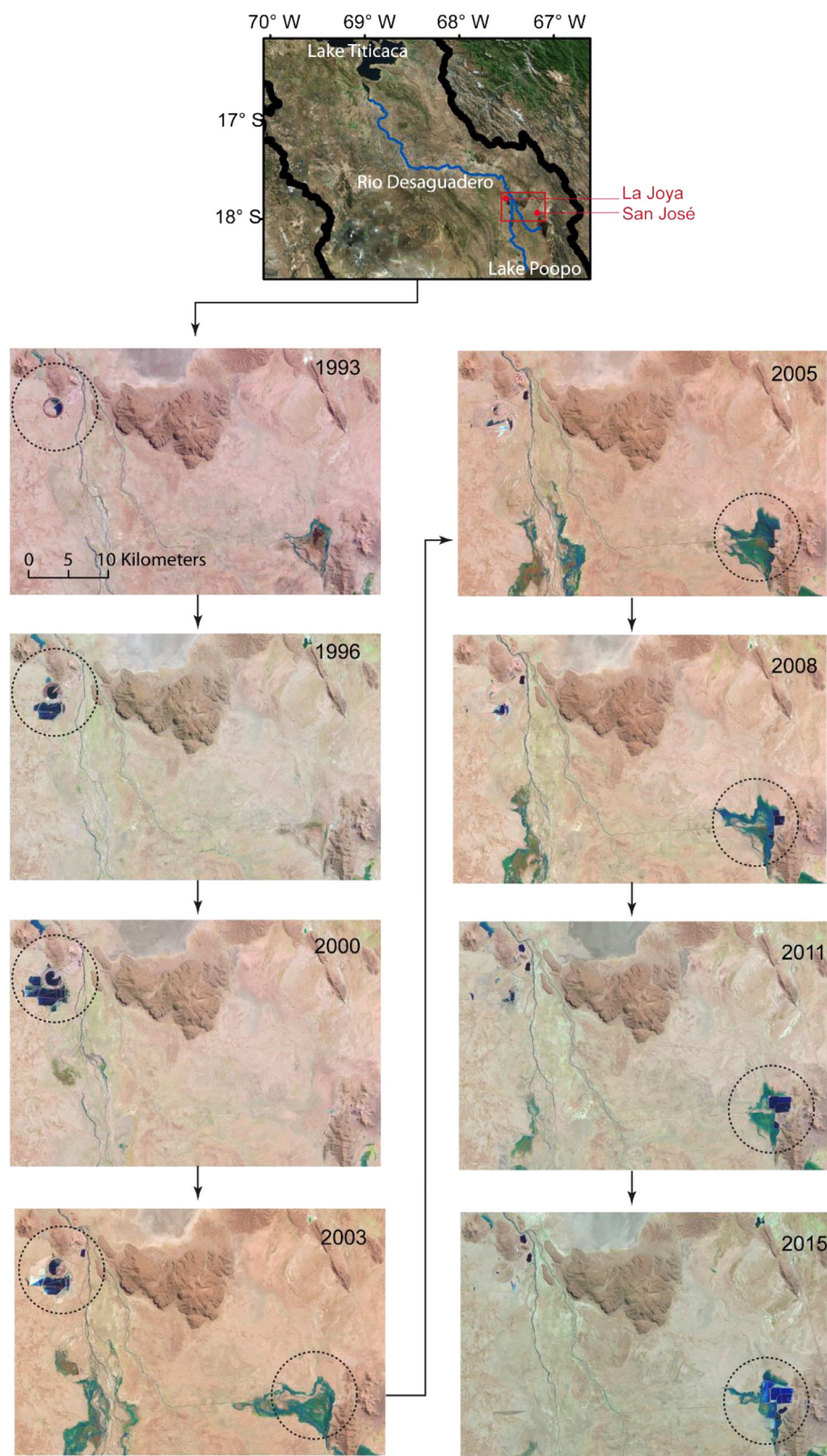


Fig. 6. Temporal evolution of two mining sites (La Joya and San José) to highlight the increased of their artificial water extents from 1993 to 2015.

irrigation aiming at optimizing water use in irrigation processes (called deficit irrigation techniques) are led over the Altiplano to support Quinoa's sustainable production (Garcia et al., 2007; Geerts, 2006; Geerts et al., 2009) and environmental quality. At term, the adoption of such practices may contribute to the sustainability of water resources.

4.4. Additional role of mining activity

Historically, the Altiplano region is known for its rich mineral resources and related mining activity, which plays a fundamental role in the local economy (Fontúrbel et al. 2011). The mineral resources have

been exploited since colonial times with the production of silver, gold, tin, tungsten, antimony, zinc, copper and lead (Ramos et al. 2012). In the last years, the global demand still stands, and extractive activity are currently intense. Mining activity contributes to the contamination of water (French et al., 2017; Salvarredy-aranguren et al., 2008), soil (Garrido et al., 2017; Hagan et al., 2011), and aerosols (Goix et al., 2016, 2011). These contaminations directly threaten the environmental stability (Guédron et al., 2017; Monroy et al., 2014) and represent a potential health hazards for the local population (Barbieri et al., 2011; Goix et al., 2016, 2011; McEwen et al., 2016; Stassen et al., 2012). Besides the contamination effects, mining activity also contributes to the decrease in TWS. Actually, the extraction process makes use of water from both superficial and/or ground reservoirs. According to our knowledge, only one report estimated mine water use over the Altiplano. The report estimated to 240 L s^{-1} the water uses of one specific mine (Huanuni) from which 66% are recycled (Van Damme, 2002). However, this value is uncertain and subject to high variation according to the extraction techniques and targeted mineral.

Even if water-supply (and quality) is a cause of conflicts between the mining sector and the population, the mining sector generally receives political support regarding water right because it represents a main national source of income (Calizaya 2009). Therefore, the access on real water use by the mining sectors is complicated. To provide a first insight on the potential role of the mining sector over the observed TWS depletion (Fig. 5d), Fig. 6 shows the temporal evolution of two main mine sectors (La Joya and Santa José) located over the study region from 1993 to 2016 as observed from Landsat Imagery.

The footprint of the superficial water bodies for La Joya and San José increase from 1993 to 2003 and from 2003 to 2015, respectively (Fig. 6). The water is directly derived from the Desaguadero River to the mine sector decreasing Desaguadero River streamflow at the inlet of Lake Poopó. At their maximum, the total water superficial extent Santa José and La Joya mine are approximated to 10 km^2 and 20 km^2 . With a mean ETp value estimated to 865 mm year^{-1} over the 1993–2016 period (GLEAM v.3), each mine could contribute to the loss of $8.65 \cdot 10^6$ and $1.73 \times 10^7 \text{ m}^3 \text{ year}^{-1}$ by evaporation from these artificial open water bodies. These amounts correspond to approximately 0.5% and 1% of the mean Desaguadero River estimated to $66 \text{ m}^3 \text{ s}^{-1}$ (Pillco and Bengtsson, 2006). However, during the dry season the Desaguadero River streamflow is low with a minimum value of $16 \text{ m}^3 \text{ s}^{-1}$ registered over the 1990–2000 period (Pillco and Bengtsson, 2006). Therefore, the impact of mining sectors during the dry season or drought events could be even worse. Even if evaporation from the different artificial water bodies is the main water output, it is worth to mention that according to the mine water balance, an additional important amount of water lockup in tailings is also lost by evaporation.

Through this simple example limited to two specific mines, the role of mining activity on TWS depletion is highlighted. Future studies should try to consider all mines which are or have been in activity over the Altiplano (Bolivia and Peru) to report on the temporal evolution of mining sector water loss and its impact on the regional TWS.

5. Conclusions

In this study, the potential role of hydrological drought events and human activities (agricultural practices and mining) over the Altiplano water scarcity process are analyzed under limited data availability conditions. The present study took advantage of several remote sensing products such as CHIRPS v.2, GLEAM v.3, AVHRR, Landsat and GRACE. Based on the new results, the dynamics of agricultural land expansion was compared to regional climatic variability (i.e. P_{net}) and to the regional vegetation and water resources state (i.e. VCI and TWS). Over the last 14 years, a significant decreasing trend in TWS is observed. With a stable monotonic trend in meteorological conditions, results bring out the potential role of rainfed Quinoa crop encroachment over native vegetation on the regional desertification process and TWS

decrease. Nevertheless, even if Quinoa seemed to play a role, it cannot be pointed out as the only factor contributing to regional desertification and water scarcity process as other crops are also experiencing large increase in terms of spatial expansions (i.e., potatoes and alfalfa). Besides agriculture, mining activity in the region also contributed to water resource depletion. In addition, the past hydrological droughts have also played an important role in the regional TWS decrease started in 1987 with the succession of severe drought event registered over the last 35 years. Therefore, this study consists in a pioneering step towards the understanding of joint combination of hydrological drought and human activity on the regional desertification and water scarcity processes.

The important no considered points in this study are: How does the precipitation deficit during hydrological droughts propagate at the longer temporal scale? Does the precipitation deficit recover between hydrological drought events? Indeed, if SPI is useful to describe punctual occurrence of past drought events, it cannot propagate drought effects over time. Moreover, which water quantity is actually consumed by the agricultural practices? Does this volume change along time, according to climate variability and water resources availability? A socio-hydrological modeling approach (Sivapalan et al., 2014) including both agricultural and hydro-geological modeling should be adapted to deal with these relevant remaining questions. As shown in this study, remote sensing is a promising tool to overcome data scarcity problems that can be used as an attractive data input choice for the socio-hydrological modeling.

Declaration of Competing interest

None.

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