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Implications of agricultural production, policy and land use changes on water resource assessment

TESIS DOCTORAL

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This water blindness is quite fascinating. In view of the rapidly growing world population, and the warnings from environmentalists that large-scale expansion of irrigation will be unacceptable, it is absolutely essential to focus on the invisible water in the soil that is necessary for plant production.

Malin Falkenmark & Johan Rockström

Balancing water for humans and nature: the new approach in ecohydrology, 2004.

A mi familia

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Summary

Achieving a more efficient and equitable water management at catchment scale does not only rely on the water resource itself, but also on other policies and scientific knowledge. There is a growing consensus that, in addition to consideration of changing climate conditions, integration with research areas such as agronomy, land use planning and economics and political science is required to meet sustainably the societal and environmental water demands. The Common Agricultural Policy (CAP) is a main driver for trends in rural landscapes and agricultural systems, but environmental deterioration is now a principal concern. One of the most relevant changes has occurred with the expansion and intensification of olive orchards in Spain, taking place mainly with new irrigated areas or with the conversion from rainfed to irrigated systems. Moreover, changing climate conditions might exert a major role on water yield trends, but it remains unclear the role that ongoing land use and land cover changes (LULCC) might have on observed river flow trends.

This thesis aims to improve the understanding of the effects of agricultural production, policies and LULCC on water quality conditions, hydrological response and human water appropriation. Firstly, the study determines the existing trends for nitrates and suspended solids in the Guadalquivir river basin's surface waters (south Spain) during the period from 1998 to 2009. From a policy perspective, the research tries to assess with panel data analysis the main drivers, including the 2003 CAP reform, which are having an influence on both water quality indicators. Secondly, water appropriation and nitrate pollution level originating from the production of olive oil in Spain is determined with a water footprint (WF) assessment, considering a spatial temporal variability across the Spanish provinces and from 1997 to 2008 years. Finally, the thesis

analyzes the effects of the LULCC on the observed negative trends over the period 1973-2008 in the Upper Turia basin, headwaters of the Júcar river demarcation (east Spain), with ecohydrological modeling.

In the Guadalquivir river basin about 20% of monitoring stations show significant trends, linear or quadratic, for each water quality indicator. Most significant trends of nitrates are augmenting than decreasing, and most significant quadratic terms of both indicators exhibit U-shaped patterns. The panel data models show that the most important drivers that are worsening nitrates and suspended solids in the basin are biomass intensification and exports of both water quality indicators from upland regions. In regions that agricultural abandonment and/or de-intensification have taken place the water quality conditions have improved. For nitrates, the decoupling of agricultural subsidies and the reduction of the amount of subsidies to irrigated land underlie the observed reduction of nitrates concentration. Measures of irrigation modernization and establishment of vulnerable zones to nitrates ameliorate the concentration of nitrates in subbasins showing an increasing trend. However, the effect of nitrates load from upland areas, intensification of biomass and crop prices present a greater weight leading to the final increasing trend in this subbasins group, where annual crops dominate. For suspended solids, there is no clear evidence that decoupling process have influenced negatively or positively. Nevertheless, greater values of subsidies still linked to production, particularly in irrigated regions, lead to increasing erosion rates. Although agricultural production has augmented in the basin and water efficiency in the agricultural sector has improved, the issue of high erosion rates has not yet been properly faced.

The water footprint (WF) assessment reveals that for 1 L Spanish olive oil more than 99.5% of the WF is related to the olive fruit production, whereas less than 0.5% is due to other components i.e. bottle, cap and label. Over the studied period, the green WF in rainfed and irrigated systems represents about 72% and 12%, respectively, of the total WF. Blue and grey WFs represent 6% and 10%, respectively. The olive production is concentrated in regions with the smallest WF per unit of product. The olive oil production has increased its apparent water productivity from 1997 to 2008 incentivized by growing trade prices, but also did the amount of virtual water exports. In fact, the largest producing areas present high water use efficiency per product and apparent water productivity as well as less nitrates pollution potential, but this enhances the pressure on the available water resources. Increasing groundwater abstractions related to olive oil exports may add further pressure to the already stressed Guadalquivir basin. This shows the need to balance the market forces with the available local resources.

Concerning the effects of LULCC on the Upper Turia basin's streamflow, LULCC play a significant role on the water balance, but it is not the main driver underpinning the observed reduction on Turia's streamflow. Increasing mean temperature is the main factor supporting larger evapotranspiration rates and streamflow reduction. In fact, LULCC and climate change have had an offsetting effect on the streamflow generation during the study period. While streamflow has been negatively affected by increasing temperature, ongoing LULCC have positively compensated with reduced evapotranspiration rates, thanks to mainly shrubland clearing and forest degradation processes. The research provides insight for strengthening the interdisciplinarity between hydrological and spatial planning, highlighting the need to include the implications of LULCC in future hydrological plans. These findings are valuable for the management of the Turia river basin, as well as a useful approach for the determination of the weight of LULCC on the hydrological response in other regions.

Resumen

Una gestión más eficiente y equitativa del agua a escala de cuenca no se puede centrar exclusivamente en el recurso hídrico en sí, sino también en otras políticas y disciplinas científicas. Existe un consenso creciente de que, además de la consideración de las cambiantes condiciones climáticas, es necesaria una integración de ámbitos de investigación tales como la agronomía, planificación del territorio y ciencias políticas y económicas a fin de satisfacer de manera sostenible las demandas de agua por parte de la sociedad y del medio natural. La Política Agrícola Común (PAC) es el principal motor de cambio en las tendencias de paisajes rurales y sistemas agrícolas, pero el deterioro del medio ambiente es ahora una de las principales preocupaciones. Uno de los cambios más relevantes se ha producido con la expansión e intensificación del olivar en España, principalmente con nuevas zonas de regadío o la conversión de olivares de secano a sistemas en regadío. Por otra parte, el cambio de las condiciones climáticas podría ejercer un papel importante en las tendencias negativas de las aportaciones a los ríos, pero no queda claro el papel que podrían estar jugando los cambios de uso de suelo y cobertura vegetal sobre las tendencias negativas de caudal observadas.

Esta tesis tiene como objetivo mejorar el conocimiento de los efectos de la producción agrícola, política agraria y cambios de uso de suelo y cobertura vegetal sobre las condiciones de calidad del agua, respuesta hidrológica y apropiación del agua por parte de la sociedad. En primer lugar, el estudio determina las tendencias existentes de nitratos y sólidos en suspensión en las aguas superficiales de la cuenca del río Guadalquivir durante el periodo de 1998 a 2009. Desde una perspectiva de política agraria, la investigación trata de evaluar mediante un análisis de datos de panel las

principales variables, incluyendo la reforma de la PAC de 2003, que están teniendo una influencia en ambos indicadores de calidad. En segundo lugar, la apropiación del agua y el nivel de contaminación por nitratos debido a la producción del aceite de oliva en España se determinan con una evaluación de la huella hídrica (HH), teniendo en cuenta una variabilidad espacial y temporal a largo de las provincias españolas y entre 1997 y 2008. Por último, la tesis analiza los efectos de los cambios de uso de suelo y cobertura vegetal sobre las tendencias negativas observadas en la zona alta del Turia, cabecera de la cuenca del río Júcar, durante el periodo 1973-2008 mediante una modelización ecohidrológica.

En la cuenca del Guadalquivir cerca del 20% de las estaciones de monitoreo muestran tendencias significativas, lineales o cuadráticas, para cada indicador de calidad de agua. La mayoría de las tendencias significativas en nitratos están aumentando, y la mayoría de tendencias cuadráticas muestran un patrón en forma de U. Los modelos de regresión de datos de panel muestran que las variables más importantes que empeoran ambos indicadores de calidad del agua son la intensificación de biomasa y las exportaciones de ambos indicadores de calidad procedentes de aguas arriba. En regiones en las que el abandono agrícola y/o desintensificación han tenido lugar han mejorado las condiciones de calidad del agua. Para los nitratos, el desacoplamiento de las subvenciones a la agricultura y la reducción de la cuantía de las subvenciones a tierras de regadío subyacen en la reducción observada de la concentración de nitratos. Las medidas de modernización de regadíos y el establecimiento de zonas vulnerables a nitratos reducen la concentración en subcuencas que muestran una tendencia creciente de nitratos. Sin embargo, el efecto de las exportaciones de nitratos procedente de aguas arriba, la intensificación de la biomasa y los precios de los cultivos presentan un mayor peso, explicando la tendencia creciente observada de nitratos. Para los sólidos en suspensión, no queda de forma evidente si el proceso de desacoplamiento ha influido negativa o positivamente. Sin embargo, los mayores valores de las ayudas agrarias aún ligadas a la producción, en particular en zonas de regadío, conllevan un aumento de las tasas de erosión. Aunque la cuenca del

Guadalquivir ha aumentado la producción agrícola y la eficiencia del uso del agua, el problema de las altas tasas de erosión aún no ha sido mitigado adecuadamente.

El estudio de la huella hídrica (HH) revela que en 1 L de aceite de oliva español más del 99,5% de la HH está relacionado con la producción de la aceituna, mientras que menos del 0,5% se debe a otros componentes, es decir, a la botella, tapón y etiqueta. Durante el período estudiado, la HH verde en secano y en regadío representa alrededor del 72% y 12%, respectivamente, del total de la HH. Las HHs azul y gris representan 6% y 10%, respectivamente. La producción de aceitunas se concentra en regiones con una HH menor por unidad de producto. La producción de aceite de oliva ha aumentado su productividad del agua durante 1997-2008, incentivado por los crecientes precios del aceite, como también lo ha hecho la cantidad de exportaciones de agua virtual. De hecho, las mayores zonas productoras presentan una eficiencia alta del uso y de productividad del agua, así como un menor potencial de contaminación por nitratos. Pero en estas zonas se ve a la vez reflejado un aumento de presión sobre los recursos hídricos locales. El aumento de extracciones de agua subterránea relacionadas con las exportaciones de aceite de oliva podría añadir una mayor presión a la ya estresada cuenca del Guadalquivir, mostrando la necesidad de equilibrar las fuerzas del mercado con los recursos locales disponibles.

Los cambios de uso de suelo y cobertura vegetal juegan un papel importante en el balance del agua de la cuenca alta del Turia, pero no son el principal motor que sustenta la reducción observada de caudal. El aumento de la temperatura es el principal factor que explica las mayores tasas de evapotranspiración y la reducción de caudales. Sin embargo, los cambios de uso de suelo y el cambio climático han tenido un efecto compensatorio en la respuesta hidrológica. Por un lado, el caudal se ha visto afectado negativamente por el aumento de la temperatura, mientras que los cambios de uso de suelo y cobertura vegetal han compensado positivamente con una reducción de las tasas de evapotranspiración, gracias a los procesos de disminución de la densidad de matorral y de degradación forestal. El estudio proporciona una visión que fortalece la interdisciplinariedad entre la planificación hidrológica y territorial, destacando la necesidad de incluir las implicaciones de los cambios de uso de suelo y cobertura vegetal en futuros planes hidrológicos. Estos hallazgos son valiosos para la gestión de la cuenca del río Turia, y el enfoque empleado es útil para la determinación del peso de los cambios de uso de suelo y cobertura vegetal en la respuesta hidrológica en otras regiones.

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AEM: Agri-environmental measures

AR(1): First order autocorrelation

AWP: Apparent water productivity

CAP: Common Agricultural Policy

CC: Climate change

CCC: Contemporaneous cross-correlation

CN: Curve Number

CEIGRAM: Research Centre for the Management of Agricultural and Environmental Risks (Centro de Estudios e Investigación para la Gestión de Riesgos Agrarios y Medioambientales)

DEM: Digital elevation model

DPS: Direct Payment Scheme

EAFRD: European Agricultural Fund for Rural Development

ET: Evapotranspiration

EU: European Union

FDC: Flow duration curve

GAEC: Good agricultural and environmental condition

GRB: Guadalquivir river basin

GRBA: Guadalquivir River Basin Authority

HDPE: High-density polyethylene

HET: Heteroskedasticity

HI: Harvest index

HRU: Hydrological response unit

LULCC: Land use and land cover changes

MA: Moving average

PBIAS: Percent Bias

PET: Polyethylene terephthalate

PP: Polypropylene

RPR: Residue to product ratio

RMSE: Root Mean Square Error

RSR: RMSE-observations standard deviation ratio

SPS: Single Payment Scheme

UPM: Technical University of Madrid (Universidad Politécnica de Madrid)

UTB: Upper Turia basin

SS: Suspended solids

SWAN: Sustainable Water Action

SWAT: Soil and Water Assessment Tool

VWE: Virtual water exports

WF: Water Footprint

WFD: Water Framework Directive

1. General introduction

1.1. Water, agriculture and environment: a global and local challenge

In the Mediterranean environment, water resources are unevenly distributed and very often in poor quality conditions. The main goal of water resources management is to reconcile in a sustainably manner multiple users' demands with a limited and variable supply of water, in time and space. However, achieving a more efficient and equitable water management at catchment scale is not only related to the water resource itself, but also influenced by other policies and scientific knowledge. Warmer and drier conditions for the Mediterranean region are expected in the coming decades with an inter-annual variability leading to a greater occurrence of extremely high temperature events (Gibelin, 2003; Giorgi and Lionello, 2008). This region considered as a primary hot spot for climate change (Giorgi, 2006). Climate conditions will probably increase evapotranspiration rates and water stress, but the impact on available water resources is difficult to predict and subject to considerable uncertainty (García-Ruiz et al., 2011).

Globally, agriculture is the most important driver of land use change (Foley et al., 2011) and is responsible for 70% of the total water withdrawals (surface and groundwater), used to produce food, feed and fibers (FAO, 2011). Between 1970 and 2010 worldwide agricultural land (including pastureland) has increased from about 3,988 to 4,888 million ha (The World Bank, 2014). But in Europe, the agricultural area (including pastureland) dropped from 205 to 189 million ha. In 2010 the European agricultural production (without including livestock) amounts to 316,540 million USD (*ibid*). Between 1970 and 2010, oilseed crops (i.e. rapeseed, olives, sunflower and soybeans) have increased from 15.1 to 32.2 million ha, whereas cereals and vegetables decreased from 185.6 to 109.7 million ha and from 5.9 to 4.2 million ha, respectively.

Besides this reduction area, cereals (from 351 to 405 million tons), grains (from 187 to 199 million tons), vegetables (from 79 to 94 million tons) and oil crops (from 7 to 21 million tons) mark a clear increasing production trend. Moreover, the area equipped for irrigation has risen from 2.9% to 5.4% of the agricultural area in Europe (FAO, 2014; *ibid*). These trends indicate both ongoing abandonment and agricultural intensification processes. On one hand, rainfed agriculture has become less dominant in the Mediterranean region during the second half of the twentieth century and forested areas have expanded in higher altitudes. This trend is likely to continue given the possibility of further abandonment of agricultural land and existing subsidies for reforestation of arable land. On the other hand, further intensification of agriculture is probably driven by market incentives, irrigation water availability and future agricultural policy and rural development (Nainggolan et al., 2012). Climate change conditions will also influence croplands distribution and production. For instance, climate change is likely to increase cereal crop yields in northern Europe and diminishes yields in southern Europe (IPCC, 2014). Climate change will also intensify irrigation requirements in Europe and future irrigation will be constrained by reduced runoff, demand from other sectors, and economic costs (*ibid*). The integration of water, agricultural and land policies is considering as a strategy for addressing competing water demands.

Water quality conditions also play a critical role in present and forthcoming water sustainability issues. In Europe by 2015, almost half of surface and 10% of groundwater bodies will be in poor ecological status, as defined by the Water Framework Directive (WFD) (EEA, 2012). The pressures from agricultural diffuse pollution represent a growing concern (*ibid*). A reduction of nitrates in groundwater and surface water during 2008-2011 in comparison to 2004-2007 has been observed in the most recent Nitrates Directive report (European Commission, 2013a), although pressures from intensive livestock and horticultural crops have not yet been sufficiently addressed. In the Mediterranean area, several studies have focused on the repercussions of agricultural activities on diffuse pollution caused particularly by nitrogen (De Paz et al., 2009), phosphorus (Torrent et al., 2007; Silgram et al., 2008; Alexakis et al., 2012) and/or sediments (Altin et al., 2009; Koklu et al., 2009; Bangash et al., 2013; Dechmi and Skhiri, 2013), but mitigating these sources of contamination is still a complex issue.

1.2. The role of the European Agricultural Policy on environmental changes

1.2.1. Evolution of the Common Agricultural Policy

The Common Agricultural Policy (CAP) was incepted in 1958 as part of the Treaty of Rome, at a time when Europe was in deficit with most food products. The CAP was initially based on a strong production support provided through a system of high support prices to farmers, combined with border protection and export subsidies (European Commission, 2012a). By the 80s improved crop genetics, increased inputs use and better management practices led to growing output production, accompanied by a relatively stagnant demand (European Commission, 1980). The MacSharry reform in 1992 began a process to shift market support through prices to direct aid payments that links support to production. The direct aids stimulate production growth and thus encourage intensification of production techniques (European Commission, 1991). The reform aimed to improve the competitiveness of the EU agriculture, stabilize the agricultural markets, diversify production and protect the environment, as well as control the EU budget expenditure (European Commission, 2012a). However, it was recognized that, in addition to surplus production concerns, the greater intensity and further agricultural production could put the environment at risk (European Commission, 1991).

In 2003, the CAP introduced a radical reform that became effective in the agricultural season 2006/2007 in Spain. Farmers were offered a Single Payment Scheme (SPS) (Pillar I), decoupled from production and conditional on cross-compliance, and the rural development program (Pillar II) was further developed. Cross-compliance is an instrument that links direct payments to fulfillment by farmers with basic standards concerning the environment, food safety, animal and plant health and animal welfare, as well as the requirement of sustaining land in good agricultural and environmental condition (GAEC) (European Commission, 2014). The cross-compliance involves 18 statutory management requirements (e.g. the Birds Directive and the Nitrates

Directive) and a number of measures for ensuring GAEC (e.g. control of soil erosion and soil organic matter content). Besides SPS, farmers can participate in agri-environmental measures (AEM) under the Pillar II (OJEC, 1999). After 2007, the European Agricultural Fund for Rural Development (EAFRD) offered farmers additional subsidies for implementing agricultural practices beyond the minimally required GAEC e.g. organic farming, crop and farming extensification and set aside (OJEU, 2005).

Regarding the decoupling process, the reaction of farmers can vary significantly between the European countries and regions due to the different climatic and soil conditions, historical traditions of farming and incidence of support, being their decisions likely to be influenced by a range of socioeconomic, demographic and farm structural factors (Gorton et al. 2008; Lobley & Butler 2010). Regions with less favorable conditions for agriculture, inefficient marketing and sales structures, and with high dependence on direct aid payments would be most severely affected by the abolishment of direct payments (Uthes et al., 2011). Renwick et al. (2013) indicates that in Spain, under the hypothesis of removal of all Pillar I payments (coupled or decoupled), the greatest decreased of agricultural area would occur for the livestock sectors, particularly those specialized in cattle rearing, grazing and dairying with negatives changes between 11 to 13.7%. For crops, the specialization in cereals, oilseeds and proteins crops group, as well as olives orchards would cause the largest impact with a reduction of about 6% agricultural area for each crops group. Spanish agriculture is beyond doubt crucially dependent on the support mechanisms of the CAP.

The most recently CAP reform 2014-2020 aims to be more effective and to contribute to a more competitive and sustainable agriculture. This includes three long term CAP objectives: viable food production, sustainable management of natural resources and climate action and balanced territorial development (European Commission, 2013b). This reform includes a simplified and compulsory cross-compliance, plus the introduction of the Green Direct Payment that accounts for the 30% of the national direct payment. Both are included in the Pillar I. Additionally, the rural development program continues to gain more weight on the achievement of CAP environmental objectives with 30% of its budget dedicated to them. Measures related to agri-environmental-climate, organic farming, Areas of Natural Constraint, Natura 2000 areas, forestry and animal welfare are considered as beneficial for the environment and/or climate (OJEU, 2013a; 2013b; 2013c).

1.2.2. CAP's environmental implications

The CAP is one of the main drivers of land use change in rural areas and in the European agriculture (Viaggi et al., 2013), with consequences of abandonment in some regions and of specialization and intensification processes in others (Stoate et al., 2009). There is also a spontaneous abandonment affecting all European mountain areas, driven by either a progressive or sudden collapse of the rural economy in regions with little agricultural productivity, as well as in semiarid areas as a consequence of soil deterioration (García-Ruiz and Lana-Renault, 2011). The abandonment in mountain areas will likely lead to some environmental concerns, i.e. shrubland encroachment, simplification of the landscape, decline of biodiversity in low intensity habitats, increase in soil erosion and intensified fire risk (Oñate et al., 2007; Duarte et al., 2008; Murua et al., 2013). Both processes, abandonment and expansion, can present a higher probability of occurrence in dry, warm and accessible areas near main cities (Hatna and Bakker, 2011). In Spain, for instance, large modernized farms chose to expand in dry and warm locations with high potential productivity because they had irrigation and water rights. In the same area, small holders without access to irrigation are forced to de-intensify or quit farming because they cannot compete with the irrigated farms (Bakker et al., 2011).

Intensification of agricultural production, especially under the stimulus of the CAP, becomes a major cause of damage to the rural environment, particularly through the loss of habitats for biodiversity, the damage to valuable landscapes, and pollution from fertilizers and pesticides (Hodge, 2013). Some studies have also identified soil erosion as a consequence of intensification promoted by the direct agricultural subsidies coupled to production (Boardman et al., 2003b) and particularly for traditional olive orchards on sloppy land (de Graaff and Eppink, 1999). A clear example of intensification is the transformation of the olive oil system in the Mediterranean region in less than 20 years. Until 2004 larger olive oil production led to greater subsidies for farmers (Scheidel and Krausmann, 2011). Nevertheless, soil erosion worsened in olive orchards because of their expansion towards soils with steeper slopes (Gómez-Limón and Riesgo, 2012). In Andalusia (Spain) olive orchards on slopes greater than 15% requires conservation tillage practices or no tillage (BOJA, 2005). However, farmers do not easily change their traditional farming practices because of the negative gross margins that reduced tillage or cover crop entails, and many olive orchards are not handled properly to control soil erosion (Francia Martínez et al., 2006; de Graaff et al., 2010). Although 2003 CAP reform promoted subsidies decoupled from production, the SPS is still favoring the more intensive olive systems, since the amount of this payment at farm level is still dependent on the historical production during the period 1999-2002 (de Graaff et al., 2011). For the period 2005-2030, de Graaff et al. (2008) projected that sloping and mountainous olive production systems will shift towards more intensive plantations and in some areas towards organic production systems in Granada, Córdoba and Jaén (south Spain), and only the orchards on the steepest slopes were likely to be abandoned. In Andalusia (south Spain) olive orchards exert 262% more pressure on the environment than a virtual farm with the same level of net income, related to a lack of technical efficiency (Gómez-Limón et al., 2012). Traditional plain groves offer the best trade-off between economic and environmental concerns, whereas mountain groves producing the lowest environmental pressures in absolute terms, their inferior economic returns make them the least eco-efficient type of olive orchard system (*ibid*).

After years of stimulating increased production for the olive orchards, the SPS cross-compliance rules and the AEM policy instruments represent a chance to improve olive-growing practices and to generate positive effects on soil conservation, water use, wildfire control, biodiversity conservation and landscape value (Duarte et al., 2008; Fleskens and Graaff, 2010). EU subsidy regime and cross compliance measures are a positive institutional development toward the avoidance of intensification of production at the cost of environmental health. However, the penalties for farmers not meeting cross-compliance requirements are not calculated on the basis of the cost of the environmental damage caused and thus may represent only a portion of this cost (European Court of Auditors, 2014). Relevant environmental concerns such as pollution in intensive systems are not addressed and the trade of agricultural commodities remains a relevant driver for land use and land cover changes (LULCC) (Fleskens and Graaff, 2010; Scheidel and Krausmann, 2011).

Different impacts on the status of water resources can be expected depending on the measures applied across CAP reforms. Several studies have projected the efficacy of the 2003 CAP reform and the decoupling process on diffuse pollution depending on the agricultural and economic context of the study region. Martínez and Albiac (2006) concluded that in the Flumen-Monegros irrigation area (Ebro basin, Spain) the reform and further trade liberalization would cause both a more intensive use of irrigation water and the abandonment of farmland, depending on the availability of human and capital resources in agricultural regions. As a result of decoupling, in central Italy farmers adopt more extensive management practices with decreased chemical input and water use (Cortignani and Severini, 2012), which might result in biodiversity gains because of reduced pressure on the environment (Overmars et al., 2013). Belhouchette et al. (2011) did not appreciated a change of nitrate leaching in the Midi-Pyrenees (France) at farm level, because of the crop pattern change occurred (wheat and maize replaced by soya, pea and sunflower) and the low value of the economic penalties in the case of no compliance. Under the scenario of total CAP subsidies abolishment, Giannoccaro and Berbel (2013) indicate that farmers' decision in south Europe moves towards a reduction in chemical and water use rates for permanent crops (i.e. olives). In the United States of America, Broussard et al. (2012) found a significant correlation between the concentration of nitrate in rivers and total government farm payments per ha of watershed. Nevertheless, in Europe the relationship of the CAP reforms with a retrospective analysis of measured water quality parameters in a typical agricultural watershed has not yet been assessed.

Chapter 3 of this thesis studies the possible implications of the 2003 CAP reform on observed surface water quality in the Guadalquivir river basin for the period 1999-2009. This basin comprises one of the most important agricultural areas in Spain, where LULCC and accelerated intensification during the last years has caused environmental concerns because of the expansion of irrigated areas and particularly of olive orchards.

1.3. Human water appropriation: the water footprint assessment and virtual water trade

Related to the definition of virtual water, Chapagain and Hoekstra (2004) introduced the Water Footprint (WF) concept for a country as the total volume of water consumed, directly or indirectly, in order to produce all products and services within the country. The methodology was further developed (Hoekstra et al., 2009; 2011b) distinguishing between the WF of a process, product, group of consumers or delimited geographic area. The WF computes only the consumptive, or non-reusable, water associated with a specific use or process. It also gives some guidance about the volumes needed to reduce pollution caused during a process (Hoekstra et al., 2011). Several studies have addressed the WF of a variety of crops products e.g. sugar-containing water beverages (Ercin et al., 2010), a pair of jeans (Chico et al., 2013) and soy products (Ercin et al., 2012), for geographical areas i.e. populations within nations (Chapagain and Hoekstra, 2004; van Oel et al., 2009; Esteban et al., 2010; Vanham and Bidoglio, 2013; Zhan-Ming and Chen, 2013) and basins (Salmoral et al., 2011; Dumont et al., 2013; Zeng et al., 2012). The WF provides a broad perspective on the water management of the system, and allows for a deeper understanding of water appropriation (Čuček et al., 2012).

Three water color components are distinguished in the WF assessment: the blue water (surface and groundwater), green and grey water. The blue water system maintains aquatic ecosystems and constitutes the water resource directly available to and consumed by humans (Falkenmark and Rockström, 2004). The green water, defined as rainfall stored as soil moisture and available for plant evapotranspiration, sustains terrestrial ecosystems as well as rainfed crop production. The grey WF refers to the volume of freshwater required to assimilate a load of pollutants based on existing ambient water quality standards (Hoekstra et al., 2011). For EU-28, Vanham and Bidoglio (2013) estimate that 58% of the precipitation is consumed as green water, whereas 7% is abstracted for human water use i.e. municipalities, agriculture, manufacturing industries and cooling water for electricity generation. Salmoral et al. (2011) and Dumont et al. (2013) estimate that green water consumption represents 81% of the annual water balance in the Guadalquivir river basin (south Spain). Within agricultural production, green water also constitutes the largest proportion. According to Vanham and Bidoglio (2013) green water comprises 93% of the total water consumption in croplands areas in EU-28, whereas Thenkabail et al. (2010) estimate 80% for the global croplands areas. Water policy and traditional water planning has only taken into account the blue water component, although green water may comprise a critical role for a more efficient and equitable water and land policies (Falkenmark, 2003; Falkenmark and Rockström, 2006). A wider perspective for water and land management is afforded with the consideration of the green water.

Virtual water is defined as the water embedded in agricultural products, which generates a “virtual” flow of water through trade of mentioned products (Allan, 1997; 1999). The WF and virtual water trade represent an innovative approach to show the situation of the world's freshwater resources and the weight of consumption patterns, for instance introducing the issue of water resources equity through virtual water trade or emphasizing the role of consumers and the impacts of their choices (Hoekstra and Mekonnen, 2012; Ercin et al., 2013). Spain is a net importer of virtual water embedded in crops with an average volume of 32,000 hm^3 $year^{-1}$ in 2008, mainly through imports of cereals and industrial crops, primarily used for feed production (Chico and Garrido, 2012). Virtual water exports represent 9,000 hm^3, and the olive orchards comprises nearly 20% of the share of economic value of the exports (*ibid*). Andalusia and its main drainage area, the Guadalquivir river basin, stands out as an unstable net virtual water exporter, owing mostly to olive oil production (Velázquez, 2007; Vanham, 2013). Since Andalusia is a net virtual water exporter under semi-arid climatic conditions, questions have been raised about the sustainability of changes in land use with the expansion of irrigated olive orchards in the region.

Recently, a water label has been proposed as a new way of certificating the efficient use of water resources (European Commision, 2011). Work is undergoing on a standard (ISO 14046) for calculating water footprints that will promote efficient measurement and management of this scarce resource. The standard will help organizations for reporting and setting an international benchmark for water use (ISO, 2012). However, the WF also requires to be complemented with additional analyses or indicators in order to achieve integrated policy options (Vanham and Bidoglio, 2013).

As impacts of water quality and availability are typically local or regional, the definitions of the WF with a spatially explicit impact provide a more robust way to address responsible water use and management (Launiainen et al., 2014). Although the use of three distinguished color components adds a new perspective to explain the source of water and the level of pollution, some authors (Launiainen et al., 2014; Pfister and Ridoutt, 2014) consider that the comparability of a hypothetical "pollution volume" (grey WF) with water consumption volumes (blue and green WFs) is questionable. This is because such sums are not environmentally meaningful or informative. Moreover, the WF just represents the quantity of water appropriation without a measurement of the related environmental impacts i.e. due to water scarcity (Jeswani and Azapagic, 2011). This has been overcome with the WF sustainability assessment (Hoekstra et al., 2011), which has already applied a blue water scarcity index (Pfister et al., 2009; Hoekstra et al., 2012; Chico et al., 2013; Pfister and Bayer, 2013) and water pollution level (Liu et al., 2012). Some studies (Aldaya et al., 2009; Garrido et al., 2010; Salmoral et al., 2011; Aldaya and Llamas, 2012; Dumont et al., 2013) have also highlighted the need of an economic valuation of water in the agricultural sector in order to consider a more efficient water allocation, since the largest proportion of blue water resources is often allocated to produce low value crops. Chapter 4 analyses geographically the explicit green, blue and grey water footprints of olives and 1 L of olive oil, an economic valuation of water and the related virtual water exports of olive oil over the period 1997-2008 in Spain. The aim of this study is to provide an overview of the water color components (blue, green and grey) during the supply chain of olive oil, but not to support decisions making of future olive oil sustainability, which require studies at local scale considering social, economic and environmental indicators.

1.4. The effects of land use changes on river flow response

Some authors whose research focuses on WF assessment (Ridoutt and Pfister, 2010; Jeswani and Azapagic, 2011) disregard the green component in water impact assessments, since it is only accessible through the occupation of land. Instead, they propose the consumption of green water in agricultural systems in the context of land use impact category. Nevertheless, modifications in the green water flow in a catchment as well as other LULCC can affect the fraction of the precipitation that becomes run-off (Hoekstra et al., 2011). Since the second half of the twentieth century decrease in streamflow has been reported in several studies in the Mediterranean region (Giakoumakis and Baloutsos, 1997; Cigizoglu et al., 2005; Lespinas et al., 2009; García-Ruiz et al., 2011), and particularly in Spain (Ceballos-Barbancho et al., 2008; Morán-Tejeda et al., 2011; Lorenzo-Lacruz et al., 2012; Martinez-Fernandez et al., 2013). Besides climate change, the most significant drivers influencing streamflow response are LULCC (Delgado et al., 2010; Gallart et al., 2011), invasion of exotic species (Little et al., 2009; Vose et al., 2011), damming, interbasin water transfers and intensive water use particularly for irrigation (Milliman et al., 2008; Aus der Beek et al., 2011).

In Spain studies have reported that up to 25% of the streamflow decrease during the last decades has been caused by ongoing LULCC (Beguería et al., 2003; Gallart and Llorens, 2004; Willaarts et al., 2012). Most of the observed LULCC across the Iberian basins include an expansion in forest cover across the middle sections and headwaters after agricultural abandonment, and the intensification of the lowlands for agriculture and urban development (Poyatos and Llorens, 2003; Pinto-Correia and Vos, 2004; Lasanta-Martínez et al., 2005; Plieninger and Schaar, 2008; Rescia et al., 2010;

Martinez-Fernandez et al., 2013). The drivers of these landscape trends include a variety of circumstances like the progressive abandonment of rural areas, industrial development, urbanization trends, changes of agricultural and environmental policies and ongoing globalization of food markets (Verburg et al., 2006; Westhoek et al., 2006; García-Ruiz, 2010; Lasanta and Vicente-Serrano, 2012; Viaggi et al., 2013).

The identification of the effects of climate change and LULCC on streamflow trends is complex because of the influence of natural hydrological variability and human activities on flow discharges (Estrela et al., 2012; Floury et al., 2012). Attributable changes have been studied in paired catchments (a treated cathment against a control catchment) using time series analysis (Zhao et al., 2010; Chappell and Tych, 2012), flow duration curves (Brown et al., 2013) and regressions analysis for separating the effects of climate and vegetation on streamflow (Zhao et al., 2010).

But the assessment of vegetation change on hydrology in paired catchments limits the analysis to specific experimental locations, and the approach is generally not feasible for large-scale watersheds as it becomes difficult to locate suitable controls (Fohrer et al., 2005). Therefore, several studies have selected a group of catchments with specific climatic conditions to carry out their analysis, but without considering a catchmet control. For instance, Zhang et al. (2012) created a model through the parametrization of the flow duration curve and predicted the change following a modification in forest cover. Similarly, Lane et al. (2005) fitted a sigmoidal function to characterize the impact of plantation growth on the flow duration curve. Farley et al. (2005) showed the effect of original vegetation type, plantation species, plantation age, and mean annual precipitation on the change in runoff.

The hydrological response has also being analyzed using regressions models of river flow against climate data and including a time variable that represents LULCC to determine the effects of several LULCC. i.e. agricultural land abandonment and forest regeneration (Beguería et al., 2003; Morán-Tejeda et al., 2010), forest plantation expansion and changes in tree species composition (Iroumé and Palacios, 2013) and drought-induced tree die-off (Guardiola-Claramonte et al., 2011). Finally, physically based hydrologic modelling has also been used to assess among others the water yield impact of agricultural abandonment and forest cover increase (Delgado et al., 2010; Gallart et al., 2011) and fire events (Lane et al., 2010).

From a policy perspective, managing both green and blue water is of critical importance to move forward with the integration of water and land policies. In the catchment management not only the projection of climate drifts needs to be considered but also the hydrological consequences of land use planning (Gallart and Llorens, 2003). Nevertheless, knowledge about local and national government regarding the implications of LULCC on river flow is still scarce. Along the second half of the twentieth century an overall streamflow reduction has been observed in the headwaters of the Júcar river demarcation (eastern Spain) (Estrela et al., 2012; Lorenzo-Lacruz et al., 2012; Martinez-Fernandez et al., 2013). Changing climate conditions have been reported as major drivers, but it remains unclear the role that ongoing LULCC might have caused on the basin's hydrological response. Chapter 5 of this thesis studies the influence of LULCC on the recorded trend of streamflow reduction over the period 1973-2008 in the semi-arid Upper Turia basin, headwaters of the Júcar river demarcation.

2. Research framework, objectives and outline

2.1. Research framework

Water resource allocation in the Mediterranean region is constrained by the dominant semiarid conditions and the growing competition between water users. Projected increase of evapotranspiration rates caused by climate change (García-Ruiz et al., 2011) at the same time as expected land use and land cover changes (LULCC) (Delgado et al., 2010; Gallart et al., 2011) can further compromise available water resources. Moreover, water resources can be affected by external drivers such as globalization of food markets and consumption patterns (Ercin and Hoekstra, 2014). Recently, the virtual water trade is identified as a tool for improving overall water efficiency on the comparative advantages of certain water uses in particular regions, and could become therefore a significant part of the solution of the global water challenge (Martínez-Santos et al., 2014). Nevertheless, globalization of food markets has associated an impact on local water resources (Duarte et al., 2014), which opens the debate of whether food production is environmentally sustainable following only market forces.

There is a growing consensus that in addition to the consideration of changing climate conditions, pursuing efficient and equitable water resource management faces challenges on the integration of the agricultural policy and land use planning (See Figure 2.1.). In 2000 the Water Framework Directive (WFD) (OJEC, 2000) introduced a framework for EU water policy based on natural geographical and hydrological formations: river basins. River basin management plans are the tools for implementing the WFD and achieving the good ecological status of water bodies. The WFD is complemented by other legislation regulating specific aspects of water use e.g. the Groundwater Directive (OJEU, 2006) and the Environmental Quality Standards

Directive (OJEU, 2008). The Nitrates Directive (OJEC, 1991a) forms an integral part of the WFD and Common Agricultural Policy (CAP) in Europe, aiming to protect water quality by preventing nitrates from agricultural sources and by promoting the use of good farming practices. Recently, the Blueprint Report (European Commission, 2012b) emphasized the need for better implementation and increased integration of water policy objectives into the CAP. However, a missing opportunity of gaining interdisciplinarity between the WFD and the CAP has occurred not introducing in the good agricultural and environmental condition (GAEC) for post CAP 2014 an obligation to control irrigation i.e. water abstraction permits, water meters and reporting on water use (European Court of Auditors, 2014).

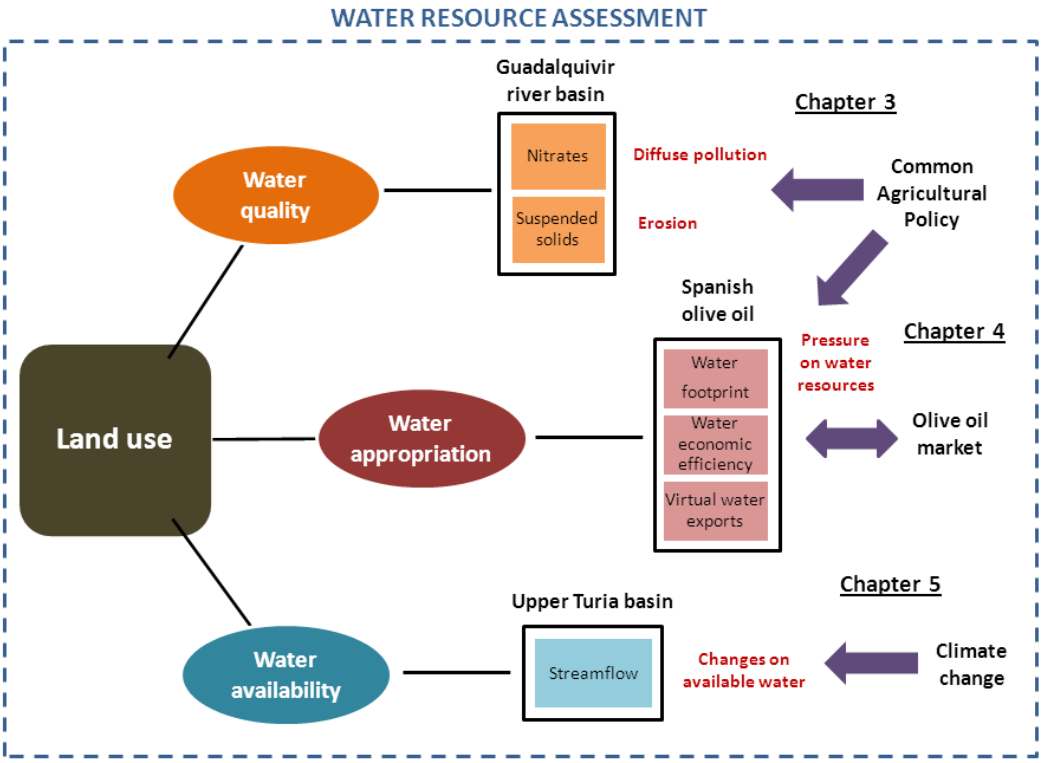

Figure 2.1. Research framework.

Soil erosion is a major problem in many European countries, and mitigation procedures need to receive greater emphasis in policy (Stoate et al., 2009). Nevertheless, there is no specific EU legal basis for soil protection and the Soil Framework Directive has been finally withdrawn, although the European Commission is working on the Thematic

Strategy for Soil Protection (European Commission, 2006) e.g. soil protection as an integral part of the GAEC and aids for rehabilitation of contaminated land (European Commission, 2012c). In Spain high erosion risk occurs in some regions, particularly for olive orchards in the Guadalquivir basin (Gómez et al., 2009; Taguas et al., 2011; Taguas et al., 2013). For the 2011/2012 agricultural season 50% of the olive oil world production was produced in Spain, totaling 1.6 million tons (IOC, 2014). Moreover, the European Union is the first olive oil consumer in the world with 58% during this agricultural season (*ibid*). The olive orchards production represents 15% of 17 million ha Spanish cropland area and in economic terms 7% of 25 billion euros crop production (MAGRAMA, 2014). Andalusia Autonomous Community produces 85% of the national olive oil production, and particularly within the Guadalquivir river basin (*ibid*). Besides market forces, the olive orchards expansion took place in Spain thanks to agricultural subsidies coupled to production, leading to intensification of the olive oil sector and the erosion risks (de Graaff and Eppink, 1999).

Concerns regarding diffuse pollution from the Spanish agricultural sector are known (Martínez and Albiac, 2006; Gallego-Ayala and Gómez-Limón, 2009; De Paz et al., 2009; Custodio et al., 2012), but need still to be tackled. LULCC can have an impact on local resources i.e. diffuse pollution or reduced availability of water resources, depending on the LULCC trends and sustainability of agricultural practices. However, neither water policy nor land management considers soil water in their assessments. LULCC processes (e.g. shrubland encroachment, reforestation) can significantly influence on the green water flow at catchment scale, and consequently have an impact on generation of runoff downstream.

2.2. Objectives and research outline

The overall aim of this thesis is to improve the understanding of the effects of agricultural production, policies and LULCC on water quality conditions, hydrological response and human water appropriation. The CAP is a main driver for trends in rural landscapes and agricultural systems. However, a retrospective analysis of the implications of CAP reforms on water quality status has not been carried out. Moreover, the effect of water appropriation and pollution level in the olive orchard agricultural production needs to be put into context of the Spanish water resources and agricultural sector. Finally, changing climate conditions are exerting a major role on the hydrological response in many basins, but the role that ongoing LULCC might have on observed decreasing trends in the Iberian Peninsula remains unclear.

The specific objectives of the thesis can be summarized as follows:

1. To determine whether the 2003 CAP reform has had an influence, and in which direction, on the nitrates and suspended solids conditions of the surface water in the Guadalquivir river basin in order to enhance coherence between agricultural and water policies (Chapter 3).

2. To achieve a better understanding in the Guadalquivir river basin, with explicit reference of temporal and spatial analysis, of the complex relationships between the observed water quality indicators with the natural environment, agricultural sector, urban sector and economic policy factors (Chapter 3).

3. To assess the human water appropriation (green and blue water footprints) and assimilation of nitrates pollution (grey water footprint) for one of the most important agricultural products in Spain, the olive oil (Chapter 4).

4. To consider the economic efficiency of the water footprint for 1L of olive oil across Spain and the related virtual water exports with the aim to provide a more robust way to address responsible water use and management (Chapter 4).

5. To identify drivers of change that exert an effect on the run-off response and to determine the magnitude and sign of LULCC versus climate change effects on observed negative changes in the Upper Turia basin (Chapter 5).

6. To assess the underlying LULCC trends that are influencing on the hydrological response of the Upper Turia basin in order to provide further knowledge on the integration of water management with land use processes, as a strategy for addressing competing water demands (Chapter 5).

The relationships between the specific objectives and the respective chapters throughout the thesis are represented in Figure 2.2, which also indicates the covered period and geographic scale of each research focus.

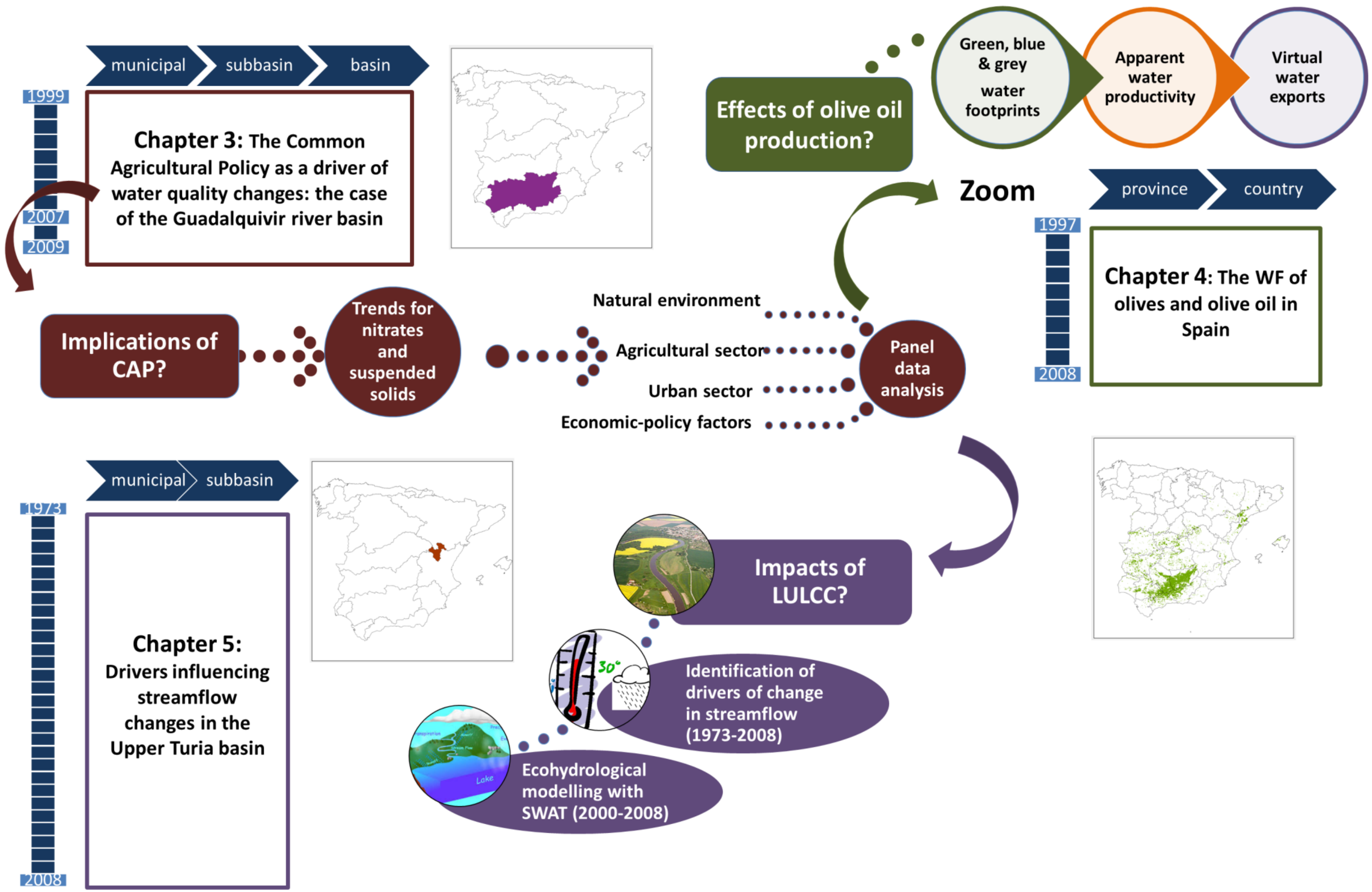

Figure 2.2. Relation between specific objectives and corresponding chapters throughout the thesis.

2.3. Thesis outline

The structure of the thesis is organized as follows. Chapter 3 presents the implications of the 2003 CAP reform on the existing surface water quality conditions in the Guadalquivir river basin during the period 1999-2009. Time trend analysis and panel data regressions of water quality indicators against variables that explain natural environment, agricultural and urban sectors, and economic-policy indicators are used. Chapter 4 applies a water footprint assessment for olives and oil in Spain. Chapter 5 studies the drivers of change of the existing negative river flow trends found in the Upper Turia basin. Finally, chapter 6 provides the main conclusions of this thesis and suggests some lines for further research.

2.4. Thesis development and research stages

The research has been funded within the grants between the Joint Research Center CEIGRAM (Research Centre for the Management of Agricultural and Environmental Risks) at the Technical University of Madrid (UPM) and the Botín Foundation. Since beginning 2010 until the end of 2013, annual grants have provided the framework for the research projects titled “Extensions Analysis of the Water Footprint and Virtual Water Trade in Spain”, “Water and Food Security in Spain” and “Water and Food Security in Latin America”. The Botín Foundation has among its work programs the Water Observatory, which is supported by two teams: one in CEIGRAM and the other in the Faculty of Earth Sciences at the Complutense University of Madrid. The main goal of the Water Observatory is to enhance the quality and relevance of political decision making with regard to water issues in Spain and around the world. The Water Observatory has shown novel approaches such as the extended WF as a tool to reveal a more equitable and efficient water allocation (Salmoral et al., 2011; Aldaya and Llamas, 2012; Dumont et al., 2013), the influence of unanticipated consequences and co-benefits in Spanish irrigation modernization (Lopez-Gunn et al., 2012) and the importance of food trade in the water resources management of a region (Aldaya et al., 2009; Novo et al., 2009; Garrido et al., 2010).

The thesis was initiated in March 2010. My research area during the first year was the analysis of the water footprint and virtual water trade of agricultural products, particularly for olive oil in Spain, and also evaluated at catchment scale. Between September 2010 and June 2011, I got a Master's Degree in Agro-Environmental Technology for Sustainable Agriculture in the Technical School of Agricultural Engineering at the UPM. The master dissertation's title was "The water footprint within the hydrological cycle: green, blue and grey water analysis of the bottom part of the Turia river basin".

In July 2011, I began the study about the impacts of the 2003 CAP reform on the surface water quality status in the Guadalquivir river basin. In summer 2012 (1 June-31 August) I visited the Department of Hydrology and Water Resources Management at the University of Kiel (Germany), hosted by the Professor Nicola Fohrer and Dr. Björn Guse in order to gain knowledge and skills in hydrologic and water quality modeling with the ecohydrological model SWAT (Soil & Water Assessment Tool) (Arnold et al., 1998). This research stay was supported by a student mobility grant as part of the PhD training studies with Quality Mention of Excellence, granted by the Ministry of Education of Spain.

Thanks to the interest of the Júcar River Basin Authority, a new research study was initiated in April 2013 to determine whether changes in land use could be responsible for the negative trends in river flows identified in the last 50 years in the Júcar river demarcation. This study was initiated at the Department of Hydrology and Water Resources, University of Arizona (USA), hosted by Professor Peter Troch (1 April - 31 July). The study focused on assessing the impacts of land use changes on the water balance of the Upper Turia basin for the period 1973-2008 applying also the ecohydrological model SWAT. This research stay was financed by the grants of internationalization for PhD studies by the Social Council (Consejo Social) at the UPM. The research stay at the University of Arizona gave me the opportunity to collaborate with the project SWAN. It aims at the creation of a transatlantic dialogue on water, involving five EU Member States (Bulgaria, France, Netherlands, Spain and United

Kingdom) and the University of Arizona team from the Hydrology and Water Resources Department. This project responds to the need of performing interdisciplinary and multiregional collaboration regarding water issues. My role in this project involves developing an innovative framework for the integration of physical disciplines such as climate and ecohydrologic modeling with human-centric approaches such as ecosystem services, societal metabolism and water footprint assessments.

2.5. Thesis publications

Chapter 3 gathers the paper under review:

Salmoral, G., Garrido, A. The Common Agricultural Policy as a driver of water quality changes: the case of the Guadalquivir river basin (south Spain). Submitted to *Journal of Environmental Management.*

Chapter 4 has been published in:

Salmoral, G., Aldaya, M.M., Chico, D., Garrido, A., Llamas, M.R., 2011. The water footprint of olives and olive oil in Spain. *Spanish Journal of Agricultural Research* 9 (4), 1089-1104.

Chapter 5 has been published in:

Salmoral, G., Willaarts, B., Troch, P., Garrido, A., 2014. Drivers influencing streamflow changes in the Upper Turia basin, Spain. *Science of the Total Environment*, in press.

Other author's publications co-authored with members of the Water Observatory are:

Willaarts, B., **Salmoral**, G., Farinaci, J.S., Sanz-Sánchez, M.J., 2014. Trends in land use and ecosystem services in Latin America, in: Bárbara A. Willaarts, Alberto Garrido, M. Ramón Llamas (Eds.), *Water for Food and Human Well-being in Latin America: Status and Challenges in a Globalized World*. Earthscan, pp. 55-80.

Dumont, A., **Salmoral**, G., Llamas, R., 2013. The Water Footprint of a river basin with a special focus on groundwater: the case of Guadalquivir basin (Spain). *Water Resources and Industry* 1-2, 60-76.

Custodio, E., Garrido, A., Coleto, C., **Salmoral**, G., 2012. The Challenges of Agricultural Diffuse Pollution, in: Lucia De Stefano and M. Ramón Llamas (Eds.), *Water, Agriculture and the Environment in Spain: can we square the circle?* Taylor & Francis, pp. 153-164

Dumont, A. & **Salmoral**, G., 2012. The extended water footprint of the Guadalquivir Basin, in: Lucia De Stefano and M. Ramón Llamas (Eds.), *Water, Agriculture and the Environment in Spain: can we square the circle?* Taylor & Francis, pp. 105-114.

Salmoral, G. & Chico, D., 2012. Lessons learnt from the analyses of the tomato and olive oil water footprint in Spain, in: Lucia De Stefano and M. Ramón Llamas (Eds.), *Water, Agriculture and the Environment in Spain: can we square the circle?* Taylor & Francis, pp. 123-136.

Salmoral, G., Dumont, A., Aldaya, M.M., Rodríguez-Casado, R., Garrido, A., Llamas, R., 2011. *The Extended Water Footprint Analysis of the Guadalquivir basin*. SHAN nº 1. Observatorio del Agua. Fundación Marcelino Botín.

Chico, D., Aldaya, M.M., Garrido, A., Llamas, M.R. **Salmoral**, G., 2010. *The water footprint and virtual water exports of Spanish tomatoes*. Papeles de Agua Virtual nº 8. Observatorio del Agua. Fundación Marcelino Botín.

3. The Common Agricultural Policy as a driver of water quality changes: the case of the Guadalquivir river basin (south Spain)

3.1. Introduction

The 2003 Common Agricultural Policy (CAP) reform in Europe, which was in force from 2006/2007 to 2013/2014 agricultural seasons, established the Single Payment Scheme (SPS) (Pillar I) decoupled from production and conditional on cross-compliance. It also further developed the rural development programs (Pillar II). "Cross-compliance is a mechanism that links direct payments to compliance by farmers with basic standards concerning the environment, food safety, animal and plant health and animal welfare, as well as the requirement of maintaining land in good agricultural and environmental condition (GAEC)"[1]. Besides SPS, farmers could participate in agri-environmental measures (AEM) (e.g. organic farming, crop and farming extensification and set aside) under the Pillar II (OJEC, 1999). A primary objective of the 2003 reform, and of the 2014 CAP reform too, was to promote a more market-oriented and sustainable agriculture. Thus, it is essential to determine whether the environmental goals of the CAP are fulfilled based on recorded indicators.

The 2003 reform has been associated on one hand with a declining of traditional agricultural activities necessary for the preservation of landscapes, particularly in low intensive areas (Brady et al., 2012). Risk of abandonment is high in these regions and negative environmental consequences are likely to ensue i.e. scrub encroachment, simplification of the landscape, decline of biodiversity, increase in soil erosion and intensify fire risk (Oñate et al., 2007; Duarte et al., 2008; Murua et al., 2013). With

[1] EU Commission: http://ec.europa.eu/agriculture/envir/cross-compliance/index_en.htm

decoupled income support, farmers also tend to adopt more extensive management practices, which results in biodiversity gains because of reduced pressure on the environment (Piorr et al., 2009; Cortignani and Severini, 2012; Overmars et al., 2013). On the other hand, decoupled direct subsidies together with years of higher agricultural prices can stimulate intensification and crop development linked to trade liberalization (Martínez and Albiac, 2006; Sieber et al., 2013), which makes the attribution of policy reforms on observed environmental effects difficult to establish.

In Europe there is no specific EU legal basis for soil protection and the Soil Framework Directive has been finally withdrawn, although the European Commission is working on the Thematic Strategy for Soil Protection since its adoption (European Commission, 2006) e.g. soil protection as an integral part of the GAEC and aids for rehabilitation of contaminated land (European Commission, 2012c). In Europe, Mediterranean and mountainous regions generally have higher sediment yield values (>0.4 t ha^{-1} and >2 t ha^{-1} for 85% and 50% of the observed sediment yield data, respectively) than the temperate and relatively flat regions of Western, Northern and Central Europe (Vanmaercke et al., 2011). Within agricultural areas, some studies have identified soil erosion in Spain as a consequence of intensification promoted by subsidies coupled to crop production (Boardman et al., 2003) and particularly for traditional olive orchards (de Graaff and Eppink, 1999). The 2003 CAP reform encouraged subsidies decoupled from production, but the SPS is still favoring more intensive olive orchards since the amount of this payment at farm level was until 2014 dependent on the historical production during the period 1999-2002 (de Graaff et al., 2011). At the long term erosion can influence on the economic sustainability of olive orchards, since positive gross margins in some olive-growing areas in southern Spain will likely disappear after about 100 years, attributable to the soil loss (Ibáñez et al., 2014).

By 2015, almost half of Europe's surface water bodies are likely to be in poor ecological status, as defined by the Water Framework Directive (WFD), with pressures from agricultural non-point pollution becoming a growing concern (EEA, 2012). Although the WFD lacks on guidance for achievement of good ecological status specifically for sediment standards (Rickson, 2014), recently, the Blueprint report (European Commission, 2012a) emphasized the need for better implementation and deeper integration of water policy objectives into the CAP. Also a missed opportunity of gaining political synergies between the WFD and the CAP has occurred, because the GAEC of the new CAP 2014 did not establish to control irrigation i.e. water abstraction permits, water meters and reporting on water use (European Court of Auditors, 2014).

The implementation of the Nitrates Directive (whose fulfillment also required for cross-compliance) reveals a general reduction of nitrates in groundwater and surface water during the last decade in Europe, but pressures from intensive livestock and horticultural crops have not been fully addressed (Bouraoui and Grizzetti, 2011; van Grinsven et al., 2012; European Commission, 2013). The predicted effects of 2003 CAP reform measures on nitrates mitigation have shown several outcomes. Volk et al. (2009) highlighted that the most effective nitrate pollution mitigation would be for management practices that change from conventional farming to eco-farming practices, as well as for the conversion of arable land to pasture land. No significant changes have been found on water quality status when crop pattern changes do not differ significantly in nutrients requirements after the reform, such as in the Midi-Pyrenees (France) (Belhouchette et al, 2011). But the partial decoupling can promote more diversified production patterns with the introduction of less nitrogen intensive crops and reducing plant area of most nitrogen intensive crops, which would lead to improvement of groundwater nitrate pollution in irrigated areas (Gallego-Ayala and Gómez-Limón, 2009). No amelioration of nitrates concentration would occur if the reduction of subsidies in case of non-compliance is low (Belhouchette et al, 2011). In fact, the penalties for farmers not meeting cross-compliance requirements are not calculated on the basis of the cost of the environmental damage caused and thus may represent only a portion of this cost (European Court of Auditors, 2014).

In the Guadalquivir river basin (GRB), with 90% of its territory in the Andalusian region (South Spain), 42% of the surface water bodies are at risk of failing to meet the environmental objectives of the WFD by 2015. The main pressure in surface water bodies is diffuse pollution (50%), followed by point pollution (37%) (GRBA, 2012). Emerging water quality problems related to diffuse pollution are related to land and water use in the basin i.e. agricultural production on hillsides aggravates water runoff and soil erosion, promoting the deterioration of stream quality and siltation of reservoirs (Blomquist et al., 2005). In Europe, an ex-post evaluation of the implications of the CAP implementation on observed surface water quality has not yet been performed. Considering the failure to meet the WFD requirements in the GRB and the existing environmental concerns, this study aims to determine the influence that the 2003 CAP reform may have had on nitrates and suspended solids concentrations from the agricultural season 1998/1999 to 2008/2009. This leads to a better understanding of the relationships between the water quality parameters with agricultural sector, urban sector, natural environment and economic policy factors. The chapter is structured as follows. Section 3.2 describes the material and methods, highlighting the monitoring stations not meeting the good physicochemical status before and after the reform for nitrates and suspended solids. Lineal and quadratic equations are fitted to determine the existing time trends of both water quality indicators throughout the basin. Then a panel data statistical analysis is detailed to determine the effects of the changes CAP support, together with other geophysical, land-use and socio-economic variables, on these water quality indicators. Section 3.3 reports the trends of GRB's nitrates and suspended solids measurements and the results of the panel data analysis, and Section 3.4 summarizes the main findings of the study.

3.2. Materials and methods

3.2.1. Site of study

The GRB covers 57,530 km^2 and the climate in the basin is Mediterranean with a mean annual precipitation that ranges from 289 mm to 743 mm over the period 1998-2009. The hydrographic network is configured around the axis of the Guadalquivir River with 655 km length and 7,022 hm^3 mean annual discharge. Downstream estuarine water bodies are found, with the Doñana National Park, a high biodiversity area and a wildlife shelter for migratory European and African birds (Fernández-Delgado, 2005) (Figure 3.1).

In 2009 croplands comprised about 2,650,780 ha (not including pastureland), with 31% of the area under irrigation. Olive orchards are the principal crop with 56% of the total cropland area, followed by wheat (13%) and sunflower (9%) (MAGRAMA, 2012a). New irrigated olive orchards have been expanded in the basin thanks to improving the efficiency of irrigation, but ameliorating the effect of water consumption savings (Martin-Ortega, 2011). Moreover, for the upper and middle part of the GRB, 20% of the olive orchards extension present erosion rates that exceeds 50 t ha^{-1} $year^{-1}$ (RGA, 2008).

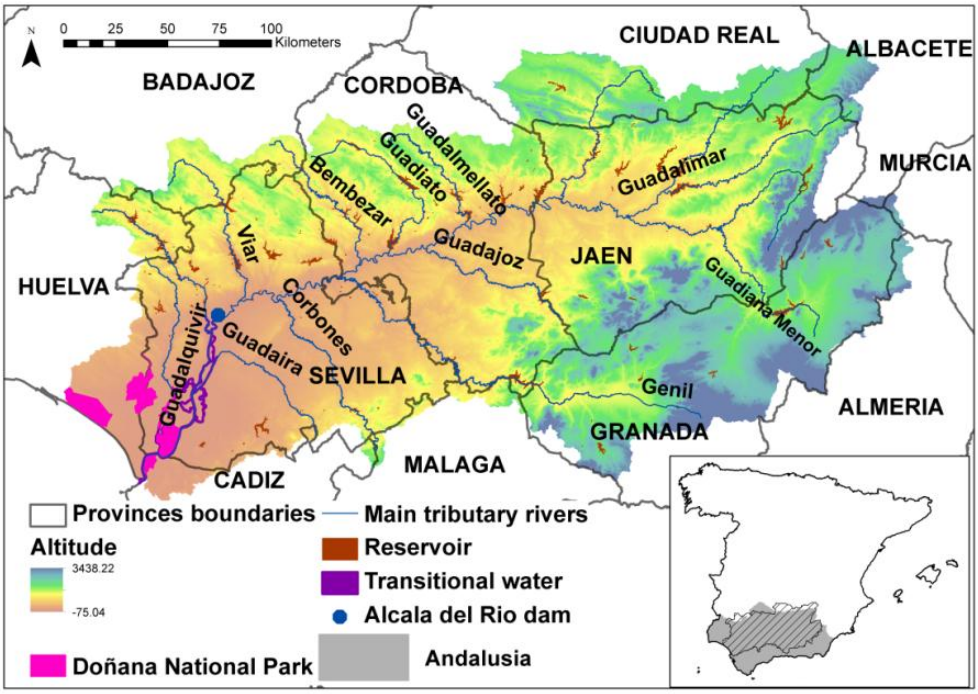

Figure 3.1. Guadalquivir river basin.

3.2.2. Analysis of water quality data and subbasins delineation

The water quality database consists of monthly observations of nitrates and suspended solids for 89 sampling stations collected during 9/1998 - 8/2009 (GRBA, 2011), though the series are not complete in all stations. Each water quality sampling station is assigned to its drainage area based on a digital elevation model (CNIG, 2011), comprising a total study area of 12,619 km^2 that represents 22% of the GRB. The 89 delineated subbasins are classified depending on the dominant irrigated agricultural type, according to criterions of geographic proximity, production orientation and economic importance (RGA, 2010a): olives orchards (‘Olives’), mountain areas (‘Mountain’), intensive crops at coast areas (‘Coast’) and semi-intensive crops (‘Semi intensive’) (Figure 3.2).

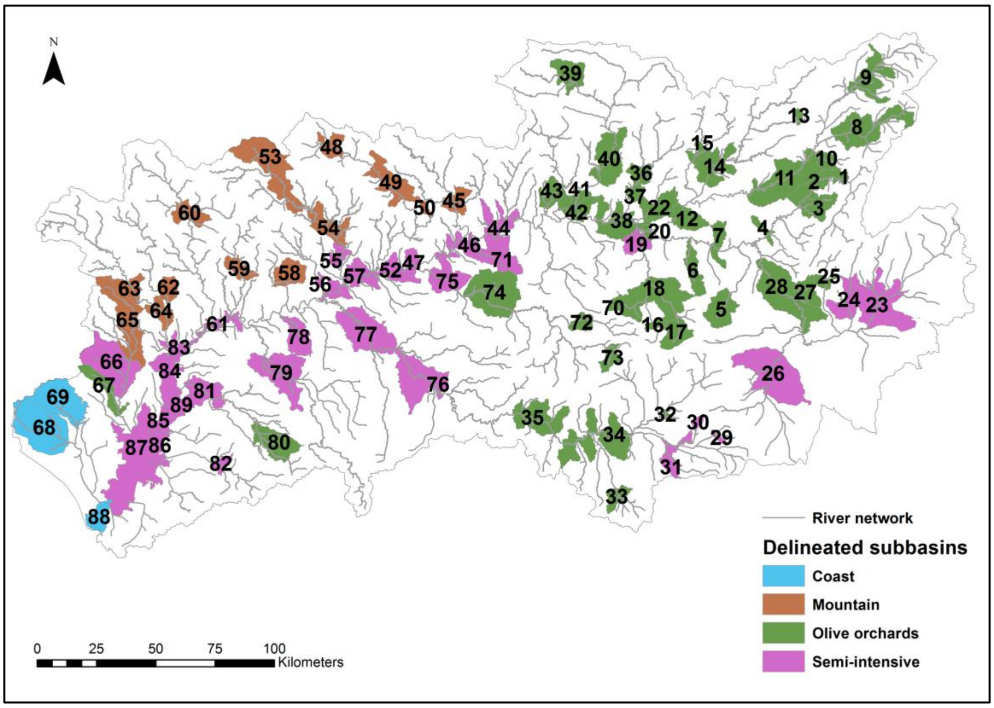

Figure 3.2. Subbasins of study and classification by dominant agricultural type. Source: own elaboration based on location of monitoring stations (GRBA, 2011), digital elevation model (CNIG, 2011) and main irrigated regions (RGA, 2010a).

Firstly, the physicochemical status for monthly nitrates (NO_3) and suspended solids (SS) observation is assessed as the percentage of exceedance before and after the agricultural reform, based on the established references value given by legislation. Water quality concentrations above the reference values of 25 mg NO_3 L^{-1} and 35 mg SS L^{-1} indicate that the good physicochemical status is not met (OJEC, 1991b; GRBA 2007; BOE, 2008).

Secondly, water quality trends are used to identify any turning point over the study period or the predominance of improvement, worsening or stability of selected parameters. The monthly dataset is summarized into annual median values (p50) for both indicators, in order to minimize the potential effect of seasonality and to account only for inter-annual changes. In our study, annual scale means the period that comprises the agricultural season from beginning September until end of August. Using the annual median values, we fit linear and quadratic time trends of both indicators in each of the 89 monitoring stations. The linear trend provides the slope (*b)* of the time variable (*H)*, with:

$$y_t = a + bH_t + \varepsilon_t \qquad [3.1]$$

Where y_t is either nitrates or suspended solids concentration, H_t is the time trend ($t=1,...,T$) and ε_t is the error term. To check for convex or concave trend behavior we also fit a quadratic trend equation:

$$y_t = a + bH_t + cH_t^2 + \varepsilon_t \qquad [3.2]$$

Based on the sign of the *b* and *c* coefficient estimators, six different time trends over the study period can be analyzed (Table 3.1).

Table 3.1. Linear and quadratic time trend with polynomial regressions (*b* and *c* as coefficient estimators).

	Coefficient	Time trend
Linear	b>0	Increase
	b<0	Decrease
Quadratic	*b*>0, *c*>0	Exponential increase (convex)
	b<0, *c*<0	Exponential decrease (concave)
	b>0, *c*<0	With a maximum (convex)
	b<0, *c*>0	With a minimum (concave)

3.2.3. Subbasins characterization

Considering the main factors involved in the water quality status of freshwater, each subbasin *i* and agricultural season *t* is characterized with a total of 16 variables over the period from 9/1998 to 8/2009 following: 1) climatic and physical environmental characteristics (*Precipitation, Slope,* $Export_{NO3}$ *and* $Export_{SS}$), 2) urban point sources (*Population density*), 3) agriculture structure and productivity measures ($Biomass_{rainfed}$, $Biomass_{irrigated}$, *Shannon, % Drip* and N_{cons}), and 4) economic and policy indicators (*Change CAP reform,* $Subsidies_{rainfed}$, $Subsidies_{irrigated}$, *% Coupling, ratio VZ* and *Crop price index*) (See Table 3.2 for further details). ArcGIS 9.3.1 (ESRI, 2009) is used to adapt geographical information from administrative (i.e. municipal, province) to subbasin level.

The mean annual precipitation and slope were calculated for each subbasin based on raster information (CNIG, 2011; SGPYUSA, 2012). Exports of nitrates ($Export_{NO3}$) and suspended solids ($Export_{SS}$) from upland were also considered as a constraint of the existing environment. They were included as the concentration of the water quality indicator of the closest headwater subbasin. Although this study focuses on diffuse pollution, we also estimated the point source pollution as the population density, based on annual population by municipality (INE, 2013).

Regarding agriculture structure and productivity measures, the total aboveground biomass in dry weight is used as an indicator of agricultural intensification since it allows for aggregating all agricultural biomass generated within a region. 129 crop species are found within the study area. The total aboveground biomass is calculated separately for rainfed ($Biomass_{rainfed}$) and irrigated ($Biomass_{irrigated}$) systems, summing up the agricultural (t) and residual (t) production and dividing by the subbasin area (ha). The agricultural production comprises the economic or agricultural parts (grain, fiber, fruit or tuber). The residual production refers to the crop residues that remain in the field after the crop is harvested. Annex A gathers in detail the calculation process for crop area and total aboveground biomass, distinguishing between rainfed and irrigated systems and annual and woody crops. Secondly, the crop diversity is measured with the Shannon Index (*Shannon*) (Shannon and Weaber, 1949), which characterizes spatially and temporally the allocation of crops area within the different species. A larger value of *Shannon* represents more diversified agricultural areas. Thirdly, modernization of irrigation system is included as the percentage of drip irrigation (*% Drip*) per type of agricultural classification ('Olives', 'Semi intensive, 'Coast' and 'Mountain'), obtained from RGA (2010c). Interpolation and extrapolation of *% Drip* is carried out between 1997 and 2008 to obtain the annual observations for the period 1999-2009. Finally, data about consumption of nitrogen at subbasin level are not available. The average consumption of nitrogen (N_{cons}) at subbasin scale is estimated by multiplying the average nitrogen rates in Spain (kg ha^{-1}) (MAGRAMA, 2012b) by the total cropland area per subbasin (ha) and dividing by the total area of each subbasin (ha).

Table 3.2. Characterization of subbasins per agricultural season over the period from 9/1998 to 8/2009.

Variable classification	Variable	Units	Available data
Climate and physical environment	*Precipitation*	mm	- Raster monthly data (SGPYUSA, 2012)
	Slope	%	- Digital elevation model of 100 m x 100 m grid cell size (CNIG, 2011)
	Exports of NO_3 and SS from upland *(Export$_{NO3}$, Export$_{SS}$)*	mg NO_3 L^{-1} mg SS L^{-1}	- Concentration of the water quality indicator of the closest headwater subbasin (GRBA, 2011)
Urban point sources	*Population density*	inhabitants km^{-2}	- Annual population by municipality (INE, 2013)
Agriculture structure and productivity measures	Total aboveground biomass for rainfed (*Biomass$_{rainfed}$*) and irrigated crops (*Biomass$_{irrigated}$*)[1]	t ha^{-1}	- Land use maps for years 1999, 2003 and 2007 (RGA, 2010; MARM, 2009). - Irrigated crops inventories of Andalusia for years 1996 and 2002 (RGA 1999; 2003) - Irrigated crops location in 2010[3] - Crop yields (MARM, 2012) - Harvest index (See Annex A Table A1) - Residue to Product Ratio (See Annex A Table A1)
	Crop diversity (*Shannon*)		- Municipal data of crop area (MAGRAMA, 2012a)
	Modernization of irrigation system *(% Drip*)	%	- Area of irrigation method (surface, sprinkler and drip) by classification of irrigated agriculture ('Olives', 'Semi intensive, 'Coast' and 'Mountain') for years 1997 and 2008 in Andalusia (RGA, 2010b).
	Consumption of nitrogen (*N_{cons}*)[1]	kg ha^{-1}	- Average national nitrogen consumption rates (MAGRAMA, 2012b)
Economic and policy indicators	*Change CAP reform*		- Dummy variable with value of 1 after 2006/2007 agricultural season
	Agricultural subsidies for rainfed (*Subsidies$_{rainfed}$*) and irrigated crops (*Subsidies$_{irrigated}$*)[1,2]	€ ha^{-1}	- Agricultural subsidies per unit of production (€ t^{-1}) or cultivated area (€ ha^{-1}) before 2006/2007 agricultural season (See Annex B Table B1) - Percentage of decoupled payments, reference period and subsidies per unit of production (€ t^{-1}) or cultivated area (€ ha^{-1}) after the agricultural season 2006/2007 (See Annex B Table B2) - Crop yields (MARM, 2012)
	Percentage of coupled subsidy (*% Coupling*)[1]	%	- Percentage of decoupled payments (See Annex B Table B2)
	Ratio of vulnerable zone to nitrates (*Ratio VZ*)[1]		- The vulnerable zone to nitrates area and crops affected by the Nitrates Directive (BOJA 1999; BOJA 2001)
	Crop price index[1,2]		- National crop prices (MAGRAMA, 2012b)

[1] Municipal data of crop area (MAGRAMA, 2012a) also required for the calculation.
[2] One year lag was also considered as explanatory variable.
[3] Provided by the Guadalquivir River Basin Authority.

Regarding economic and policy indicators, the study distinguishes among the effects of CAP, Nitrates Directive and crop prices. Agricultural subsidies ($Subsidies_{rainfed}$ and $Subsidies_{irrigated}$, in € ha^{-1} agricultural area) are used as the first CAP's policy indicator. 32 crops are entitled to receive subsidies. One-year lag for both variables ($L.Subsidies_{rainfed}$ and $L.Subsidies_{irrigated}$) is also considered, since farmers' behavior might also be influenced by subsidies from the previous agricultural season. The calculation differs between the Direct Payment Scheme (DPS) (1999-2006) and the SPS (2007-2009). The second CAP policy indicator is the average percentage of coupled subsidy (*% Coupling*). Calculations for both CAP's indicators are detailed in Annex B.

The GAEC in Spain are applicable for cross-compliance since 2006/2007 agricultural season, however there is not available information at our analysis scale regarding the level of adherence to SPS support recipients. Since we cannot characterize in quantitative terms the GAEC, a dummy variable (*Change CAP reform*) characterizes the agricultural policy implementation after the 2006/2007 agricultural season.

The implementation of the Nitrates Directive is included as the ratio of vulnerable zone (*Ratio VZ*) to nitrates per subbasin area. The vulnerable zone extension is divided by the subbasin area and multiplied by the proportion of crops area affected by the Nitrates Directive within the vulnerable zone. Finally, a *crop price index* is calculated based on 2000 current prices and weighted by the crop production. One-year lag for crop price index (*L.crop price index*) is also considered, since agricultural practices might also rely on prices from the previous agricultural season.

3.3.3. Fitting panel data models

A. Model formulation

The application of panel data analysis aims at explaining the variation of the median (p50) of both physicochemical indicators taking into account the variables that characterize the water quality status, described in Section 3.2.3. The analysis of panel data allows us to model time series processes while accounting for heterogeneity across geographical units (subbasins, characterized by monitoring stations). The dataset has a panel data structure since data appear on N units (subbasins) over T periods (agricultural seasons). The general regression model for analyzing panel data is formulated as follows:

$$y_{it} = z_i'\alpha + x'_{it}\beta + \varepsilon_{it} \qquad i=1,\ldots,N;\ t=1,\ldots,T \qquad [3.3]$$

Where X_{it} is the ith observation of each explanatory variable. The heterogeneity is controlled by the intercept $z_i'\alpha$, where z_i includes a constant term and a set of individual or group specific variables (Greene, 2012). The error component model includes the unobservable unit effects (λ_i) and the remainder disturbance (u_{it}):

$$\varepsilon_{it} = \lambda_i + u_{it} \qquad [3.4]$$

The fixed effect (FE) panel data model assumes λ_i to be fixed parameter and u_{it} independent and identically distributed (IID) (0, σ^2_u). This model requires estimating N separate λ_i that together with the intercept $z_i'\alpha$, comprise a dichotomous variable (v_i) for each unit. FE only analyzes the impact of variables that vary over time, since time-invariant variables are absorbed by v_i. The random effect (RE) model assumes λ_i random where λ_i ~ IID (0, σ^2_λ), u_{it}~ IID (0, σ^2_u) and the λ_i are independent of the u_{it} (Baltagi, 2008).

However, FE or RE regressions residuals often show attributes that ordinary least squares (OLS) cannot handle. Feasible Generalized Least Squares (FGLS) or Panel Corrected Standard Errors (PCSE) can be used to deal with heteroskedasticity[2] (HET), contemporaneous cross-correlation[3] (CCC) and first order autocorrelation[4] (AR (1)). Driscoll and Kraay Standard Errors (DKSE) is applied when the autocorrelation is a moving average type (MA) (Hoechle, 2007), which represents the average value of a variable over a given period of time. We chose the PCSE or DKSE models since our dataset does not always present the same number of observations per subbasin and FGLS require rectangularized datasets.

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[2] The variance of the errors of each panel is not constant.
[3] Observations of some panels are correlated with other panels during the same period of time. It refers to the correlation of the errors of at least two or more panels.
[4] Autocorrelation occurs when errors are not independent with regard to time.

B. Model selection

Statistical software STATA 12 (StataCorp, 2011) was used for fitting the models. We set up models at three different scales: 1) at the scale of the whole basin including all subbasins ('Total'), 2) selecting a group of subbasins based on the dominant irrigated agricultural type: 'Olives', 'Semi intensive', 'Coast' and 'Mountain', and 3) selecting subbasins by existing water quality time trends: 'Increasing', 'Decreasing', 'Minimum' and 'No trend'. Regressions for the 'Maximum' classification were not carried out because of the lower number of observations (less than 10). Since the analysis comprises two water quality parameters and different subbasins classification, we used the following abbreviation: [water quality parameter] [subbasin classification].

Variables were transformed with logarithm as *ln(variable+1)* and standardized to have a mean of zero and standard deviation of one. Explanatory variables with a pairwise correlation coefficient larger than 0.6 were excluded from the analysis. Regressions were firstly run for pooled OLS and we did not allow a subset of explanatory variables with variance inflation factor greater than 4. Then panel RE and FE OLS were run. After running the RE model, we performed the Breusch and Pagan Lagrange multiplier test (Breusch and Pagan, 1980) to verify the absence of RE with the null hypothesis that error variance across units is zero. So under the null hypothesis pooled OLS is consistent. After running FE regressions, we test the null hypothesis that all dichotomous variables are zero (Ho: $v_1 = v_2 = v_i = 0$) with a F test (Snedecor and Cochran, 1983). The selection between RE or FE depends on whether the individual error component (u_{it}) and explanatory variables are correlated, which is identified with the Hausman test (Hausman, 1978). If u_{it} and explanatory variables are correlated FE model is required.

We chose the PCSE model when disturbances were assumed to be heteroskedastic across panels or heteroskedastic and contemporaneously cross-correlated across panels, with or without AR(1). Heteroskedasticity was detected in the FE residuals with the modified Wald statistic for groupwise heteroskedasticity (Greene, 2012). To test the cross sectional correlation we performed the Pesaran test (Pesaran, 2004) after running FE with the null hypothesis that the error term is independent across sections. Finally, autocorrelation was tested under the null hypothesis of no serial autocorrelation (Wooldridge, 2002). We selected the DKSE model when heteroskedasticity, cross correlation panels and MA was found. MA was identified if coefficient of determination presented larger values for DKSE model than for PCSE. We only kept variables with significant coefficient estimators ($p<0.05$). With the Wald test, we checked if the removed variables are not significant in the final panel model regression with the null hypothesis that coefficients are zero.

3.3. Results and discussion

3.3.1. Crop changes by agricultural subbasin classification

Considering the 89 subbasins under study, both processes of agricultural abandonment and agricultural intensification prevail throughout the basin. On one hand, the rainfed agriculture area has decreased from 480,840 to 412,230 ha between 1999 and 2009. The largest crop area reductions occurred under rainfed conditions for olives (20,015 ha in ‘Olives’), wheat (13,880 ha and 9,500 ha in ‘Semi intensive’ and ‘Olives’ areas, respectively) and fodder (11,240 ha in ‘Mountain’). On the other hand, irrigated crops has increased from 213,560 to 258,715 ha, mainly due to the expansion of olives (43,135 ha in ‘Olives’ and 21,365 ha in ‘Semi intensive’) on existing rainfed areas or occupying new regions. Nevertheless, a significant decrease under irrigation took also place for industrial crops i.e. cotton (13,510 ha), sunflower (6,655 ha), sugar beet (2,280 ha) and rape (1,250 ha), mostly in ‘Semi intensive’ (Figure 3.3).

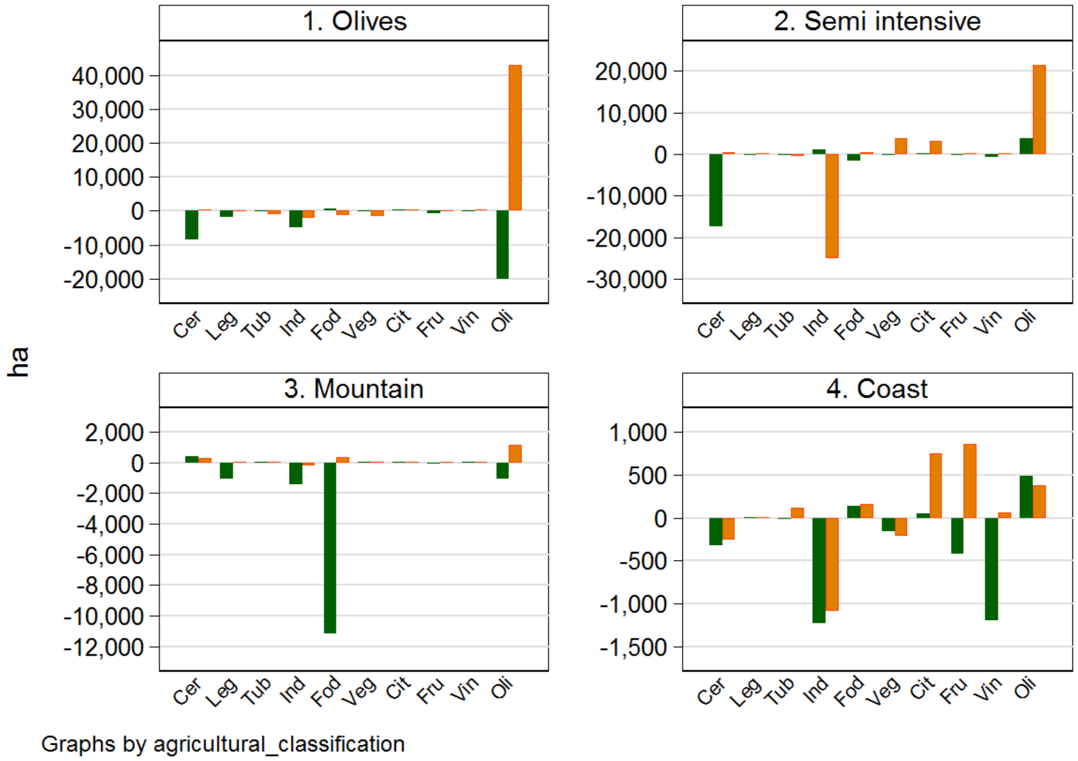

Figure 3.3. Difference of crop area (ha) between 1999 and 2009 under rainfed (green left bar) and irrigated (orange right bar) conditions by agricultural subbasins classification (Olives, Semi intensive, Mountain and Coast) and main crop groups (Cer: cereals, Leg: leguminous, Tub: tubers, Ind: industrial crops, Fod: fodder, Veg: vegetables, Cit: citrus tree, Fru: fruit tree, Vin: vineyard, Oli: olive tree).

3.3.2. Water quality assessment in the GRB

Different physicochemical properties are found in the GRB depending on the type of dominant agriculture, with values of nitrates above 25 mg L^{-1} in 90th percentile within ‘Semi intensive’ and ‘Coast’ subbasins groups. The high concentration of suspended solids (>35 mg L^{-1}) is an indication of intense erosion processes reducing water quality throughout the basin, with the exception of ‘Mountain’ classification (Table 3.3).

Table 3.3. Descriptive statistics for monthly observations in 89 sampling stations over the period 9/1998-8/2009 and by subbasin classification according to dominant irrigated agricultural type.

		Nitrates				Suspended solids (mg L^{-1})			
Subbasin classification	No. Sampling stations	No. Observations	p10	p50 (mg L^{-1})	p90	No. Observations	p10	p50 (mg L^{-1})	p90
Mountain	13	916	1	2	7	827	3	8	30
Olives	41	2201	1	5	21	2551	4	25	156
Coast	3	158	1	5	26	210	11	41	191
Semi intensive	32	2307	2	12	33	2377	9	53	280

*p10: 10th percentile; p50: 50th percentile (median), p90: 90th percentile.

The percentage of non-compliance for good physicochemical status for the entire GBR's monitoring stations does not vary significantly before and after the 2003 reform (2006/2007 agricultural season), although nitrates slightly increase (from 11.5% to 12.8%) and suspended solids diminish (45.7% to 44.3%). For nitrates, almost the same number of sampling stations does not comply before and after the reform, with 41 and 40 subbasins, respectively. Failure to meet the standards for suspended solids is more widespread throughout the basin, but the number of subbasins not meeting the standard value has decreased from 85 to 65 after 2007 year (Table 3.4).

Table 3.4. Number and percentage of monthly observations of nitrates (NO_3) and suspended solids (SS) that do not meet the good physicochemical status before and after the 2003 CAP reform (2006/2007 agricultural season).

Period	Total Obs.	No. Subbasins with NO_3 >25 mg L^{-1}	No. Observations with NO_3>25 mg L^{-1}	% observations with NO_3>25 mg L^{-1}
Before CAP reform	4046	41	466	11.5%
After CAP reform	1536	40	197	12.8%
Period	**Total Obs.**	**No. Subbasins with SS > 35mg L^{-1}**	**No. Observations with SS > 35mg L^{-1}**	**% observations with SS > 35mg L^{-1}**
Before CAP reform	4991	85	2279	45.7%
After CAP reform	974	65	431	44.3%

Table 3.5 and Figure 3.4 show the time trends for the median annual values of nitrates and suspended solids for the period 9/1998 - 8/2009. For nitrates, 13.5% of subbasins had significant linear trends ($p<0.05$ of coefficient *b* in equation 3.1), among them 9% are increasing (b>0). Significant quadratic trends ($p<0.05$ of coefficients *b* and c in equation 3.2) represent 7.9% of subbasins, with 6.7% showing an inverted U-shaped curve. Subbasins that exhibit a U-shaped curve (convex) began worsening from

February 2003 and March 2006. As a result, we can affirm that among the existing trends prevail those that worsen nitrates concentration. The group of nitrate increasing trends (linear or with a minimum) takes place mostly within 'Semi intensive' areas in the south-eastern part (subbasins 23, 26, 28, 29 and 31), middle Guadalquivir (subbasins 56, 74 and 80) and estuary (subbasins 68, 81, 87 and 89). In the southeast of the basin (subbasins 23, 26, 28, 29 and 31), dominate olives, cereals and vegetables. Under irrigation between 1999 and 2009, olives, cereals and vegetables were expanded (by 5,350 ha, 850 ha and 615 ha, respectively) whereas industrial crops area decreased (by 880 ha), with the peculiarity that irrigation system has scarcely been modernized in this region (Gómez-Limón et al., 2013). The Guadaira river (subbasin 80 and 81) has been identified as a source of urban pollution, whereas the intermediate zone of the estuary (subbasin 87 and 89) are mostly affected by agricultural activities and partially by treated urban effluents (López-López et al., 2011). However, high nitrogen concentrations in the estuary present a seasonal pattern that matches with the timing of local fertilization practices (González-Ortegón & Drake, 2011). Moreover, in the estuary land uses and regulation upstream can have an influence on the water quality, not only because of the contribution of pollutants, but also caused by the loss of dilution and transport capacity of the current flow regime, much lower than the natural flow regime of the Guadalquivir (Contreras, 2012).

Table 3.5. Linear and quadratic trends regressions with significant coefficients ($p<0.05$) out of 89 monitoring stations for annual median values of nitrates (NO_3) and suspended solids (SS) concentrations.

		NO_3	**SS**
	Trends	**No. Subbasins**	**No. Subbasins**
Linear	**Increasing: b>0**	8	3
	Decreasing: b<0	4	5
	Total	**12**	**8**
Quadratic	**Maximum: b>0 & c<0**	1	1
	Minimum: b<0 & c>0	6	14
	Total	**7**	**15**

Regarding the behavior of suspended solids, most trends were fitted with a quadratic component (16.9%), and particularly for U-shaped curve with 15.7%. For linear equations, 9% of sampling stations present statistically significant trends. However, despite the acute suspended solids concentrations in the basin, only 5.6% of sampling stations reduced the suspended solids concentrations. Decreasing suspended solids concentration occur in the east of the basin (subbasin 26), upper part (subbasins 13 and 27) and middle part (subbasin 74), within 'Semi intensive' and 'Olives' areas with the exception of subbasin 69 in 'Coast'. The reduction of suspended solids can be related to the 38% total agricultural area decrease between 1999 and 2009 within these subbasins, mainly caused by the reduction of planted area of cereals (7,860 ha wheat, 2,960 ha barley and 1,200 ha oat) and industrial crops (5,000 ha sunflower and 1,765 ha flax) under rainfed conditions. But olives have expanded under both rainfed (2,970 ha) and irrigated (2,940 ha) systems. Worsening of suspended solids concentration (both linear and quadratic) takes places particularly throughout 'Olives' subbasins classification, and in 'Semi intensives' areas along the Guadalquivir river axis. The most significant crop area change under both trends has been a significant expansion of olives (7,795 ha) probably on soils with steeper slopes (Gómez-Limón and Riesgo, 2012), at the same time that rainfed wheat (5,005 ha) and irrigated industrial crops (2,035 ha cotton and 1,015 ha sunflower) areas were reduced.

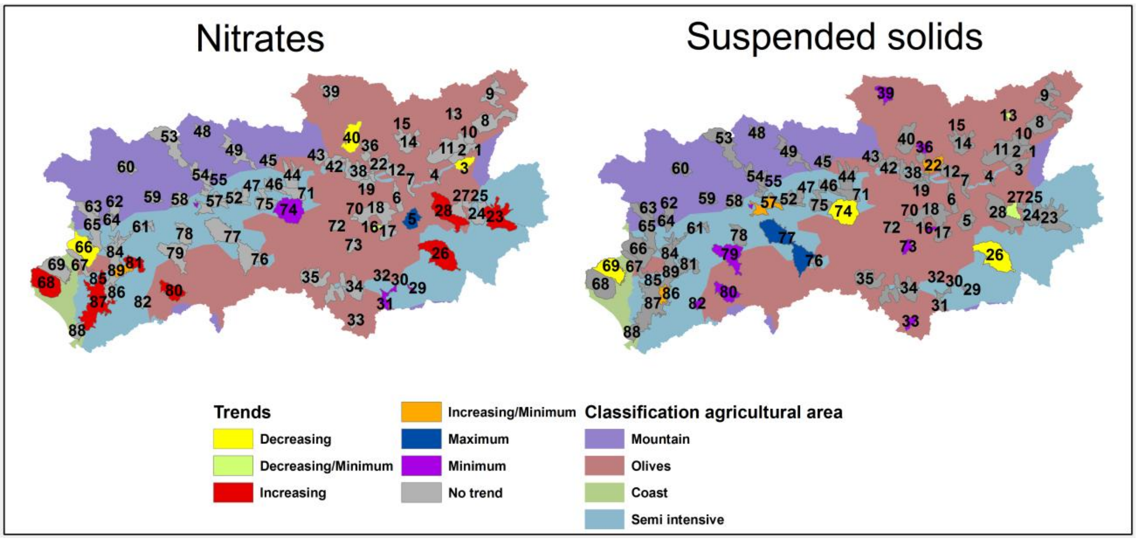

Figure 3.4. Linear and quadratic trends regressions with significant coefficients ($p<0.05$) for median values of nitrates and suspended solids concentrations. Classification of dominant agricultural areas across the basin is also shown.

3.3.3. Panel data analysis

Tables 3.6 and 3.7 report the model fits for median annual measurements of nitrates and suspended solids, respectively. The modeling results indicate whether panel data analysis is required instead of pooled regression, when random effects or fixed effects are relevant between subbasins. Models that are fitted with fixed effects (‘NO_3 Minimum’, ‘SS Olives’, ‘SS Decreasing’ and ‘SS Increasing’) requires a dichotomous variable for each entity, that controls for all time-invariant differences between subbasins. Therefore, only the net effect of predictors changing over time can be assessed.

Overall, one of the most significant coefficient regressor for nitrates concentration is the nitrate export from upland ($Export_{NO3}$) with positive sign ($p<0.001$). Export from upland subbasins can be a major contributor of excess nutrients (King and Balogh, 2011), being upstream land use crucial in water chemistry for larger streams (Mouri et al., 2011). In the GRB, agricultural intensification of both rainfed ($Biomass_{rainfed}$) and irrigated ($Biomass_{irrigated}$) systems and point sources (*population density*) represent also the most significant variables ($p<0.001$) with positive sign.

For the total of subbasins (NO_3 Total), nitrates concentration increases with biomass intensification, more diversity of crops (*Shannon*), larger subsidies to irrigated land from the previous season ($L.Subsidies_{irrigated}$) and after the CAP reform (*Change CAP reform*). Similarly, in ‘Olives’ subbasins group, nitrates concentration increases with intensification and *Shannon*, but *Change CAP reform* is not significant. Greater *Ratio VZ* and larger subsidies to rainfed crops from the previous season ($L.Subsidies_{rainfed}$) also increase nitrates concentration in ‘Olives’. Intensification of biomass per agricultural area has taken place in ‘Total’ (from 4.4 to 4.6 t ha^{-1}) and particularly in ‘Olives’ (from 3.1 to 3.6 t ha^{-1}) after the reform. De Graaff et al. (2008) highlighted intensification as a common trend in the olive oil sector in Andalusia (Spain), where semi-intensive and low input orchards may gradually turn into semi-intensive and high input and intensive orchards. The positive influence of *Shannon* on the nitrates concentrations is probably related to greater fertilization requirements under a larger variety of crops. The greater probability of runoff or leakage for nitrates in the vulnerable zones can also explain the positive sign of *Ratio VZ* in 'Olives' (as well as in 'Minimum' and 'No trend'). In fact, in hilly and mountainous olive orchards, the cross-compliance and AEM policy instruments had positive environmental outcomes i.e. soil conservation, but in intensive systems relevant environmental concerns such as water pollution are not sufficiently addressed (Fleskens and Graaff, 2010).

In 'Mountain', larger nitrates concentration takes place under greater $Biomass_{irrigated}$, lower $Subsidies_{irrigated}$ and before the agricultural reform. The negative sign of the dummy variable *change CAP reform* hide drivers not directly related to CAP measures, like the reduction of nearly 30% of agricultural areas during the study period (from 41,675 ha to 29,130 ha). Low productivity and aging of the population can stimulate farmland abandonment in marginal areas and lowering the effectiveness of agricultural subsidies to halt abandonment in the mountainous areas (Navarro and Pereira, 2012). De-intensification has also occurred in the area with a decrease of irrigated biomass from 6.1 to 5.7 t ha^{-1} after the reform. This de-intensification takes place particularly on industrial crops (from 7.7 to 4.9 t ha^{-1}) and cereals (from 12.2 to 10.4 to ha^{-1}). Although, after the reform intensification has also showed up for fodder (from 11.5 to 17.1 t ha^{-1}) as well as with the expansion of irrigated olive orchards (from 390 to 1,520 ha). Processes of abandonment, intensification and expansion have been taking place at the same time in 'Mountain', where all of them can present a higher probability of occurrence in dry, warm and accessible areas (Hatna and Bakker, 2011). Technical progress i.e. irrigation in Spanish farmers made possible to maintain or even increase crop yields in areas with less propitious physical conditions and greater distances from markets (Bakker et al., 2011).

On one hand, water quality of receiving water bodies are improved in 'NO_3 Increasing' with the modernization of irrigation system (*% Drip*), since nitrogen losses in the irrigation return flows are reduced with the development of higher irrigation efficiency systems (Lecina et al., 2010; Barros et al., 2012). Similarly, the negative sign of vulnerable zones areas (*Ratio VZ*) is related to the convenience of considering the whole alluvium area as vulnerable zone to nitrates in mitigation measures (Arauzo et al., 2011; Arauzo and Valladolid, 2013). On the other hand, $Export_{NO3}$, $Biomass_{rainfed}$ and *crop price index* cause a greater positive change on nitrates concentration, leading to the final increasing trend. Higher prices may translate into higher production intensity (Kirchner and Schmid, 2013; Renwick et al., 2013), what may lead farmers to make decisions about production without evaluating their environmental consequences (Boardman et al., 2003). 'NO_3 Increasing' trend is located in 'Semi intensive' and 'Coast' areas, where annual crops dominate (in 2009 cereals, industrial crops and vegetables comprise 39%, 12% and 3% of the total agricultural area), giving farmers more flexibility to change their crop patterns from year to year according to the market demand.

The direct influence of CAP's decoupling process is appreciated in 'NO_3 Decreasing' and 'NO_3 No trend' with the positive effect of coupling to production (*% Coupling*) to the nitrates content. In the 'Decreasing' group, the observed trend of nitrates concentration is correlated with the reduction of *% Coupling* applied during the decoupling process as well as to lower irrigated subsidies ($Subsidies_{irrigated}$). These results would be in line with the more extensive management practices applied under decoupled income support (Piorr et al., 2009; Cortignani and Severini, 2012; Overmars et al., 2013). In fact, the average biomass per agricultural area has decreased from 5.5 to 5 t ha^{-1} between 1999 and 2009, mainly because of the reduction of industrial crops (from 7.4 to 3.8 t ha^{-1}). Regarding the positive correlation between the concentration of nitrate in rivers and subsidies payments, Broussard et al. (2012) also found it in the United States of America.

Table 3.6. Panel data regressions for nitrates (*NO_3*) considering all the subbasins (Total), by agricultural type (Olives, Coast, Semi intensive and Mountain) and by existing time trend (Decreasing, Increasing, Minimum and No trend). Only significant variables (p<0.05) are included in the regression models.

Dependent variable NO_3	**Subbasins classification**								
Explanatory variables	**Total**	**Olives**	**Coast**	**Semi intensive**	**Mountain**	**Decreasing**	**Increasing**	**Minimum**	**No trend**
$Export_{NO3}$	0.20***	0.27***		0.25***			0.38***		0.23***
Precipitation			0.17*			-0.19*			
Slope		0.33***							
Population density		0.21***	0.64***					31.92***	0.07***
$Biomass_{rainfed}$	0.33***	0.36***					0.48***		0.19***
$Biomass_{irrigated}$	0.35***	0.29***	-0.76**		2.85***				0.18***
Shannon	0.08*	0.31***							0.09***
N_{cons}				0.75***					0.22***
% Drip				0.31**			-0.34**		
Ratio VZ		0.15*		-0.06***			-0.57***	4.18**	0.10**
Crop price index							0.29*		
$Subsidies_{rainfed}$			-0.91*						
$Subsidies_{irrigated}$					-0.09*	0.30***			
$L.Subsidies_{rainfed}$		0.24***							
$L.Subsidies_{irrigated}$	0.09**								
% Coupling						0.17*			0.04*
Change CAP reform	0.15**		0.62**	0.16*	-0.45***				
Intercept			-1.41***	0.36**	1.61***	-0.27**	-0.23*	-15.41***	
No. Subbasins	89	41	3	32	13	4	8	6	72
No. Observations	778	293	33	293	137	35	70	51	645
R^2	0.48	0.46	0.785	0.44	0.28	0.53	0.44	0.54	0.54
Model	DKSE with RE	PCSE with RE	Pooled OLS	DKSE with RE	Pooled OLS	Pooled OLS	Pooled OLS	PCSE with FE	PCSE with RE
	HET	HET		HET		HET		HET	HET
	MA			MA					
	CCC	CCC		CCC				CCC	CCC

* p<0.05, ** p<0.01, *** p<0.001. OLS: Ordinary Least Squares, PCSE: Panel Corrected Standard Errors, DKSE: Driscoll-Kraay Standard Errors, FE: Fixed effects, RE: Random effects, AR(1): Autocorrelated of type AR(1), MA: Autocorrelated with moving average, CCC: Contemporaneous cross-correlation, HET: Heteroskedasticity.

Regarding suspended solids, overall export of sediments from upstream (*$Export_{SS}$*, $p<0.001$) and agricultural intensification ($p<0.001$) are in general the most significant variables and with positive sign. *$Export_{SS}$* can explain the negative sign of terrain slope (*Slope*) in our panel data regressions, since sediments are dragged from upstream and accumulated downstream (Gómez, 2008). For the total of subbasins ('SS Total'), erosion rates are aggravated by greater biomass intensification, *$Subsidies_{irrigated}$*, *% Drip* and after the *Change CAP reform*. As a result, greater erosion rates are expected in more productive areas, with better water efficiency, larger subsidies to irrigated land and after 2006/2007 agricultural season.

In 'Olives', suspended solids concentration are positively correlated with biomass intensification (particularly for *$Biomass_{irrigated}$*), *Shannon*, *ratio VZ* and *$Subsidies_{irrigated}$*. In contrast, the more efficient is the irrigation system and the lower *$Subsidies_{rainfed}$*, the lower suspended solids concentrations are found. The olives orchards have grown by 48,990 ha thanks to modernization and new irrigated areas have expanded on soils with steeper slopes (Gómez-Limón and Riesgo, 2012). According to our biomass calculation, the olive trees biomass grew from 70 to 85% of the total biomass during the study period in *'Olives'*, thanks to new irrigated areas (from 63,200 ha in 1999 to 106,300 in 2009) and the intensification of biomass production from 4 to 5 t ha^{-1}. Nevertheless, no significant correlation has been found between soil erosion rates and olive yield (Vanwalleghem et al., 2011). Instead, soil management practices have proved to be unsustainable (Gomez et al., 2008; Junta de Andalucía, 2008; Xiloyannis et al., 2008). Since olive farmers can hardly perceive the economic costs of inappropriate soil management, they do not feel forced to adopt soil conservation practices (Ibáñez et al., 2014). The positive sign of *Shannon* in *'Olives'* might be then related to more frequent tillage practices or herbicide control of weeds under more variety of crops, which reduce surface cover and roughness (Vanwalleghem et al., 2011).

Subbasins that show decreasing trends of suspended solids ('SS Decreasing') are explained by improving water quality conditions upstream, larger *$Biomass_{irrigated}$*, more efficiency of irrigation system and lower *Subsidies*$_{rainfed}$. This trend can also be explained by the reduction of total agricultural area in this subbasin group at the same time that additional olive orchards and modernization of irrigation system has taken place. In contrast, 'SS Increasing' trend is supported by *change CAP reform* and higher prices index from the previous season (*L.crop price index*). The most significant change between 1999 and 2009 in 'SS Increasing' has been the expansion of 4,230 ha and 2,030 ha of olives and citrus, respectively, which might have led to more frequent tillage practices or herbicides control applications leaving the soil bare. The suspended solids that worsen the concentration up to a certain point in time ('SS Minimum') are also related to larger consumption of nitrogen (N_{cons}) (such as in 'Total' and 'Semi intensive) probably in line with the higher frequency of tillage and herbicides use in major agricultural producing areas.

Table 3.7. Panel data regressions for suspended solids (*SS*) all the subbasins (Total), by agricultural type (Olives, Coast, Semi intensive and Mountain) and by existing time trend (Decreasing, Increasing, Minimum and No trend). Only significant variables (p<0.05) are included in the regression models.

Dependent variable *SS*	**Subbasins classification**								
Explanatory variables	**Total**	**Olives**	**Coast**	**Semi intensive**	**Mountain**	**Decreasing**	**Increasing**	**Minimum**	**No trend**
$Export_{SS}$	0.24***	0.21***	1.22***	0.33***		0.70***		0.57***	0.21***
Precipitation						-0.14**			
Slope	-0.29***			-0.46***	-0.51***				
Population density			-0.26**	-0.11***					
$Biomass_{rainfed}$	0.17***	0.27***		0.21***	0.13*				0.21***
$Biomass_{irrigated}$	0.26***	0.44***				-4.19***			0.49***
Shannon		0.06*							
N_{cons}								0.40***	
% Drip	0.07***	-1.04***				-1.48***			
Ratio VZ		0.20***	-1.10*	-0.20***					
Crop price index									
L crop price index							0.16*		
$Subsidies_{rainfed}$		-0.10***				0.40**			
$Subsidies_{irrigated}$	0.12***	0.14***		0.20**					0.07***
% Coupling									
Change CAP reform	0.21***						0.69**	0.64***	0.17***
Intercept	-0.03*	1.13***			-0.59***	-2.55**			
No. Subbasins	89	41	3	32	13	5	3	14	71
No. Observations	803	330	30	291	133	39	28	134	638
R^2	0.59	0.51	0.79	0.62	0.49	0.867	0.89	0.59	0.53
Model	PCSE with RE	PCSE with FE	Pooled OLS	PCSE with RE	PCSE with RE	PCSE with FE	Panel OLS with FE	PCSE with RE	PCSE with RE
	HET	HET	HET	HET	HET	HET		HET	HET
					AR(1)				
	CCC	CCC				CCC			CCC

* p<0.05, ** p<0.01, *** p<0.001. OLS: Ordinary Least Squares, PCSE: Panel Corrected Standard Errors, FE: Fixed effects, RE: Random effects, AR(1): Autocorrelated of type AR(1), CCC: Contemporaneous cross-correlation, HET: Heteroskedasticity.

3.4. Conclusions

This study analyses diffuse pollution in the Guadalquivir river basin resulting mainly from the land use cover and agricultural practices, focusing on the effects of the 2006/2007 CAP reform. In general, in the GRB the water quality parameters have remained stable during the study period. But the majority of significant trends of both nitrates and suspended solids are worsening the water quality conditions. The observed relationships of the nitrates and suspended solids with natural environment, agricultural sector characteristics, urban areas and economic policy factors are likely to be region specific. However, improving the understanding of these correlations helps to further implement policies related to water and land management.

Discerning the effects of point and diffuse sources in the water quality status is difficult. In general, exports of both nitrates and suspended solids from upland subbasin and intensification of agricultural systems increase the concentration of both water quality indicators. In regions where agricultural abandonment and/or de-intensification have taken place (i.e. 'NO_3 Mountain' and 'SS Decreasing') the water quality conditions have improved. The decoupling of agricultural subsidies of the CAP reform and the reduction of the amount of subsidies to irrigated land underlies the observed reduction trends of nitrates concentration. In subbasins that show an increasing trend for nitrates, measures of irrigation modernization as well as the establishment of vulnerable zones to nitrates help to reduce nitrates concentration. However, the effect of nitrates load from upland areas, intensification of biomass and crop price index present a greater weight leading to the final positive trend in this subbasins group where annual crops dominate.

For suspended solids, there is no clear evidence that the decoupling process has had a negative or positive influence. Although some improvements in suspended solids concentration are observed in the basin, greater concentrations can be found in more productive areas, with better water efficiency, greater subsidies to irrigated land and after 2006/2007 agricultural season. The implications of intensification are particularly significant for erosion rates in *'Olives'* and under irrigated conditions.

While our modeling framework included some of the most important measures of income support and a relevant price index, our causation hypotheses can be disturbed by price volatility in the world's and European's agricultural markets occurred after 2007. An intra-annual assessment would determine if commodity prices over short periods affect the physicochemical status of surface water bodies. Low R^2 values suggest, particularly in *'Mountain'* nitrates model, that additional explanatory variables e.g. livestock load are required to explain a larger proportion of the variance of the water quality parameters. Despite the technical and political difficulties that appeared during the negotiation of the CAP post 2014, conditionality measures, greening components and rural development programs might be great opportunities to improve the water quality conditions encountered in the Guadalquivir river basin in the next 2014 CAP reform.

4. The water footprint of olives and olive oil in Spain

4.1. Introduction

In a context where water resources are unevenly distributed and, in regions where flood and drought risks are increasing, improved water management is urgently needed in Spain. In this country, about 85% of all water is used for food production (Garrido et al., 2010), and it is the largest producer and exporter of olive oil and table olives in the world. In the 2011/2012 agricultural season, 50% of the olive oil world production was produced in Spain, amounting to 1.6 million tons and 13,900 million euros (IOC, 2014; MAGRAMA, 2014). The European Union comprises the first olive oil consumer in the world with 58% of the total during this agricultural season (IOC, 2014). In addition, olive oil is a basic product of the Mediterranean diet and its moderate consumption contributes to a healthy diet (IOC, 2010).

The water footprint of a product is the volume of freshwater used to produce the product, measured over the full supply chain. It is a multi-dimensional indicator, showing volumes of water consumption and pollution (Hoekstra et al., 2009). The blue water footprint refers to consumption of blue water resources (surface and ground water) along the supply chain of a product. The green water footprint refers to consumption of green water resources (rainwater stored in the soil as soil moisture). The grey water footprint is defined as the volume of freshwater that is required to assimilate a load of pollutants, based on existing ambient water quality standards. Previous to this study, Garrido et al. (2010) calculated a total water footprint for crop production (blue and green) in Spain of 27,620 and 23,590 hm^3 for a humid (1997) and dry (2005) year type, respectively. The green and blue water footprint of olives represented 35% of the total water consumed for crop production in Spain for the period 1997-2006.

Authors have claimed that drought effects can be mitigated and water savings achieved through virtual water trade (food trade) in water stress regions (Allan, 1999; Chapagain et al., 2006; Hoekstra and Chapagain, 2008). The unequal spatial distribution of global water resources can be compensated by virtual water trade (Islam et al., 2007). Other authors indicate that virtual water trade is a misleading concept, which cannot be used to alleviate water scarcity (Ansink, 2010) and neither used alone as a criterion for selecting optimal trading strategies (Wichelns, 2010). Garrido et al. (2010) found that Spain is a net importer of virtual water embedded in crops, where Andalusia stands out as a net exporter owing mostly to olive oil production. Spain imports lower value crops (e.g. grain) and exports high value crops (e.g. vegetables and fruits) (Novo et al., 2009). Within their study, Garrido et al. (2010) also showed that virtual water imports and exports grew significantly during the period 1997-2006. Most of the exports originate in the southern and south-east regions, with the most water-stressed basins. Dietzenbacher and Velázquez (2007) also evaluated virtual water trade in Andalusia. Since this region is a net virtual water exporter under semi-arid climatic conditions, environmental concerns have been raised about the expansion of irrigated olive orchards.

The present study analyses geographically the explicit green, blue and grey water footprints of olives and one liter of olive oil, the apparent water productivity and the related virtual water exports of olive oil over the period 1997-2008 in Spain. The aim of this study is to provide an overview of the three color components during the supply chain of olive oil, but is not intended for decisions making of future olive oil sustainability, which would require studies at local scale considering social, economic and environmental indicators. The chapter is structured as follows. Section 4.2 describes the material and methods for the calculation of the water footprint, apparent water productivity and virtual water exports for olive oil in Spain. Section 4.3

reports the main results, and Section 4.4 summarizes the main conclusions of the study.

4.2. Materials and methods

4.2.1. The water footprint assessment of olive oil

The green, blue and grey water footprints of olives and 1 L of bottled olive oil are calculated following and refining the method described by Hoekstra et al. (2009). The water footprint is determined for a region (i.e. province and country) in absolute terms (volume) and relative terms (volume per unit of product). First, the water footprint of olive orchards is calculated as a whole including both oil and table varieties. Two types of olive production systems are analyzed: irrigated vs. rainfed. Then, the analysis focuses on the production chain of 1 liter of olive oil, indicating the consumptive volume along the value chain. Apparent water productivities and virtual water exports calculations are also provided based on Garrido et al. (2010).

The water footprint of olive oil includes both a supply chain and an operational part:

$$WF_{product} = WF_{supply\ chain} + WF_{operational}$$

[4.1]

Where $WF_{product}$: the water footprint of a product; $WF_{supply\ chain:}$ the water footprint of the supply chain; $WF_{operational}$: the operational water footprint (all of them in hm^3 or L $product^{-1}$).

The $WF_{supply\ chain}$ is defined as the amount of freshwater used to produce all the goods and services that form the product inputs at a specific business unit:

$$WF_{supply\ chain} = WF_{supply\ chain\ \ ingredients} + WF_{supply\ chain\ [other\ parts]}$$

[4.2]

The $WF_{supply\ chain\ [ingredients]}$ refers to the water footprint directly associated to ingredients (i.e. olives) and the $WF_{supply\ chain\ [other\ parts]}$ includes the water footprint of other components (i.e. bottle, cap and label).

The $WF_{operational}$ is defined as the amount of freshwater used at a specific business unit. The operational water footprint of olive oil includes the water associated to the production of olive oil. Life cycle assessment studies (Avraamides and Fatta, 2008) of olive oil production calculated that the blue water consumed during the olive oil processing stage only accounts for 1.4% of the overall water consumption. In addition, it can be assumed that all wastewater is treated with 100% treatment performance and effluent characteristics of the treated wastewater are within the legal limits. With these assumptions, the operational water footprint for olive oil production is considered to be negligible.

4.2.2. Supply chain water footprint related to the olive fruit

The supply chain water footprint of olives (in hm^3 or m^3 t^{-1}) in Spain has been calculated distinguishing the green ($WF_{supply\ chain\ [ingredients]\ green}$), blue ($WF_{supply\ chain\ [ingredients]\ blue}$) and grey water components ($WF_{supply\ chain\ [ingredients]\ grey}$):

$$WF_{supply\ chain\ \ ingredients} = WF_{supply\ chain\ \ ingredients\ green} + WF_{supply\ chain\ \ ingredients\ blue} + WF_{supply\ chain\ \ ingredients\ grey} \quad [4.3]$$

A. Green and blue water footprint of olives

CROPWAT model configuration

The green and blue water consumption of olives has been estimated as the actual evapotranspiration of olive orchards using the CROPWAT model (FAO, 2009). Two scenarios were distinguished for rainfed and irrigated conditions. Calculations were done by each province *i* and year *j*. For irrigation scenario we did not assume that plant water requirements were met, since this is not practical for olive agricultural practices (See section *CROPWAT scenarios*).

CROPWAT requires soil characteristics, climatic data and crop parameters. The olive orchards area on each soil texture type was obtained for each of the provinces using ArcGIS 9.3.1 software (ESRI, 2009). The olive orchard cropping pattern is outlined using the Corine Land Cover 2000 (CLC2000) (EEA, 2009) and the Inventory and Characterization of Irrigated Land in Andalusia 2002 (RGA, 2003) (Figure 4.1). The first layer presents a 1:100,000 scale (Bossard et al., 2000) and the latter is obtained at 1:50,000 scale. CLC2000 illustrates the rainfed and irrigated olive orchards distribution for all provinces, except for the distribution of irrigated olive groves in Andalusia, which is taken from the Inventory. Both layers provide a reliable distribution of this perennial crop and indicate the most probable locations where olive orchards are grown. Nevertheless, both the CLC2000 and the Inventory and Characterization of Irrigated Land in Andalusia 2002 present some limitations as they have not been updated since their creation. In addition, CLC2000 does not include the six provinces where olive groves have been developed after the year 2000 (Álava, Guipúzcoa, Lugo, Las Palmas, Santa Cruz de Tenerife and Valladolid), but these provinces comprised only 852 ha in 2008 out of 2,450,447 ha in Spain as a whole (MARM, 2010a).

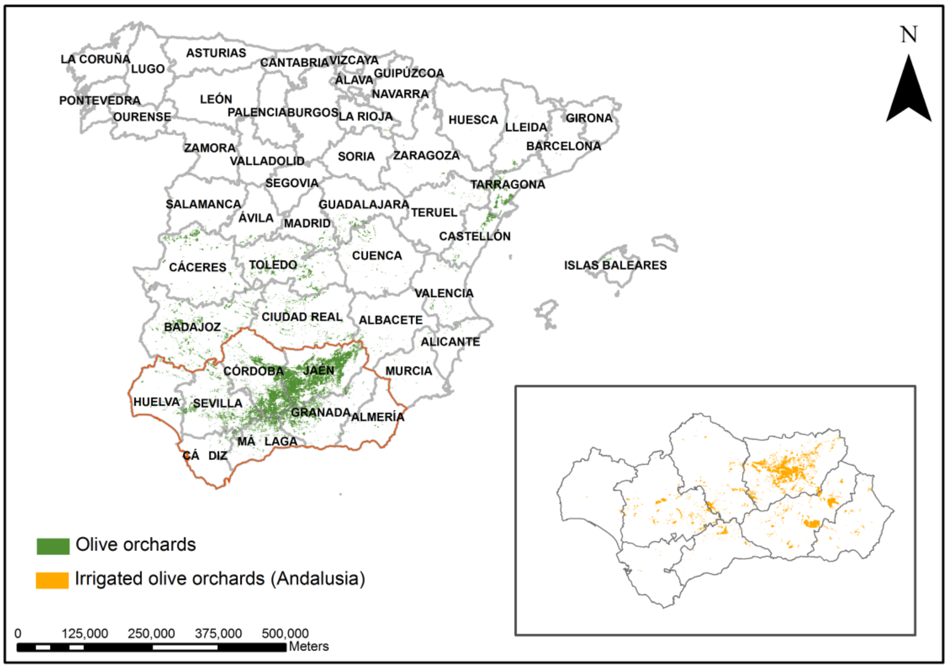

Figure 4.1. Olive orchards distribution (rainfed + irrigated systems) in Spain and irrigated olive orchards distribution in Andalusia. Source: Own elaboration based on EEA (2009) and Regional Government of Andalusia (2003).

Soil type data have been taken from European Soil Data Base version V2.0 (European Commision, 2003) at 1: 1,000,000 scale. Four textural classes *k* were identified: coarse, medium, medium-fine and fine. Reference values of physical soil characteristics depending on its texture are taken from Israelsen and Hansen (1965). The proportion of each textural class per province was calculated. The initial soil moisture content of each year is estimated using a ratio between the sum of the precipitation of November and December from the previous year to the total available water content.

Representative meteorological stations located in the major crop-producing regions are selected depending on data availability. Monthly reference evapotranspiration (ET_o) and precipitation for each of the provinces are obtained from the National Meteorological Agency (AEMET, 2010). These databases have been completed with the Integral Service Farmer Advice for the years 2007 and 2008 (MARM, 2010b). CROPWAT was set up to calculate the effective rainfall based on USDA soil conservation service method.

Required crop parameters have been obtained from the literature (Allen et al., 2006; Lorite et al., 2004; Orgaz et al., 2005), making a distinction between rainfed and irrigated olives. Constant tree densities and crown volume are assumed for rainfed (100 trees ha^{-1} and 9,000 m^3 ha^{-1}) and irrigated orchards (200 trees ha^{-1} and 9,000 m^3 ha^{-1}). Root depth is assumed to be 0.6 m, since most of the roots are located at this depth (Connel and Catlin, 1994). Once climate data, crop parameters and proportion of soil textural class per province were determined, CROPWAT calculations were performed per province *l*, year *j* and textural class *k*. The distinction of water consumption depending on the textural class *k* is a refinement of the method of Hoekstra et al. (2009).

CROPWAT scenarios

Rainfed production is simulated in the model by choosing to apply no irrigation for each province *l*, year *j* and soil textural class *k*. In the rainfed scenario (indexed with *irr = 0*), the green water evapotranspiration is equal to the actual evapotranspiration as simulated by the model and the blue water evapotranspiration is zero:

$$ET_{green\,ijk}\ irr = 0\ = ET_{a\,ij}\ irr = 0 \qquad [4.4]$$

$$ET_{blue\,ijk}\ irr = 0\ = 0 \qquad [4.5]$$

Where ET_{green} *(irr=0)*: Green water evapotranspiration (mm) in the rainfed scenario ET_{blue} *(irr=0)*: Blue water evapotranspiration (mm) in the rainfed scenario; $ET_{a\,ij}$ *(irr=0)*: Actual water evapotranspiration (mm) in the rainfed scenario.

For the irrigation scenario (*irr = 1*) the irrigation water estimated according to the water allowances in the Guadalquivir river basin, since this basin comprised approximately 88% of the irrigated olive area in Spain in 2004 (AQUAVIR, 2005; MARM, 2010b). For a normal climatic year, olive orchards have a net water allowance of 2,281 m^3 ha^{-1} (GRBA, 2007b). This water allowance considers transport and distribution losses. Estimated water allowances over the period 1997-2008 depend on the drought level in the basin (*ibid*) and are calculated based on the net water allowance of a normal climatic year when no drought occurs (Table 4.1). To establish the level of drought, the management system "General Regulation" of the Guadalquivir basin was analyzed since it included nearly 70% of irrigated water use for agriculture in the basin (MARM, 2008a). Each year of the period 1997-2008 was classified in relation to the volume of water stored in reservoirs and drought level, which indicates the reduction in agricultural water use applied.

Table 4.1. Estimated water allowances for the period 1997-2008 based on the level of drought.

Year	Level of drought	Water saving in agriculture (%)	Estimated water allowances (m^3 ha^{-1})
1997	No drought		2280
1998	No drought		2280
1999	Prealert	5	2170
2000	Alert	30	1600
2001	No drought		2280
2002	No drought		2280
2003	No drought		2280
2004	No drought		2280
2005	Prealert	5	2170
2006	Alert	30	1600
2007	Alert	30	1600
2008	Alert	30	1600

The CROPWAT irrigation option selected was "irrigation at fixed interval per stage" with "fixed application depth". It was assumed a field efficiency of 0.9 for drip (Strosser et al., 2007) since most olives areas in Spain are irrigated with the drip method. From 2003 to 2009 drip irrigated systems of olives have increased from 90 to 94% of the total irrigated systems (MARM, 2009b). Irrigation schedule required to be representative for the completely olive extension, so it was necessary to generalize the irrigation schedule for the entire area. Thus, irrigation schedule was determined distributing the net water allowance along the irrigation period. Though the frequency and application depth of a same irrigation volume will affect the final green water, since the crop does not meet the water requirements, irrigation losses owing to the designed program schedule will tend to be minimal.

In order to facilitate calculations with CROPWAT only two irrigation schedules were established based on the estimated net water allowances. The first one applies 2,200 m^3 ha^{-1} for normal years (years 1997-1998 and 2001-2004) and pre-alert situations (years 1999 and 2005). Irrigation depth is 3 mm one day out of four between March and October. This period provides an irrigation depth of 24 mm $month^{-1}$. The second irrigation schedule applies 1,600 m^3 ha^{-1} for alert level of drought (years 2000 and 2006-2008). The irrigation is limited to one out of four days with an irrigation depth of 2 mm between March and May and one out of three days between June and October. Over the first period 16 mm $month^{-1}$ were applied, whereas 20 mm $month^{-1}$ over the second one. The origin of blue water could only be taken into account for provinces of Andalusia based on the Inventories of Irrigated Land in Andalusia (RGA, 1999; 2003; 2008; Corominas, J 2010, pers. comm., 29 June).

Under irrigated conditions, the actual evapotranspiration is equal to the actual water use by crop over the growing period. The blue water evapotranspiration refers to the 'total net irrigation' ($Irr_{loss\ net}$) minus irrigation losses (Irr_{loss}). The former includes irrigation losses owing to type of irrigation system; the latter refers to water losses because of no adequate irrigation schedule. The green water evapotranspiration (ET_{green}, mm) is equal to the actual evapotranspiration (ET_a, mm) minus the blue water evapotranspiration (ET_{blue}, mm), as simulated in the irrigation scenario (*irr=1*) for each province *l,* year *j* and soil textural class *k*:

$$ET_{blue\ ijk}\ (irr = 1) = Irr_{los\ net} - Irr_{loss} \quad [4.6]$$

$$ET_{green\ ijk}\ (irr = 1) = ET_{a\ ijk}\ (irr = 1) - ET_{blue\ ijk}(irr = 1) \quad [4.7]$$

The green water footprint of the crop per unit has been estimated as the ratio of the green water consumption to the crop yield. The green water consumption is obtained by summing up separately the green water evapotranspiration over the growing period of rainfed and irrigated systems. The green water footprint in m^3 is calculated multiplying the final green water consumption over the growing period and the crop area. Similar calculations were applied to obtain the blue water footprint per unit and total. Calculations were made for each province *i,* year *j* and production system *l* for the green WF, and for each province *i* and year *j* for the blue WF:

$$WF_{green\ ijl}\ [m^3\ t^{-1}] = \frac{10 \times (ET_{green\ k} \times R_k)_{ijl}}{Y_{ijl}} \quad [4.8]$$

$$WF_{green\ ijl}\ [m^3] = 10 \times (ET_{green\ k} \times R_k)_{ijl} \times A_{ijl} \quad [4.9]$$

$$WF_{blue\ ij}\ [m^3\ t^{-1}] = \frac{10 \times (ET_{blue\ k} \times R_k)_{ij}}{Y_{ij}} \quad [4.10]$$

$$WF_{blue\ ij}\ [m^3] = 10 \times (ET_{blue\ k} \times R_k)_{ij} \times A_{ij} \quad [4.11]$$

Where WF_{green}: Green water footprint (m^3 t^{-1} or m^3)*; ∑ (*ET_{green} *x R)*: Green water evapotranspiration (ET_{green} in mm) according to the proportion *R* of each textural class *k; Y* : Crop yield (t ha^{-1})*; A*: Crop area (ha)*;* WF_{blue}: Blue water footprint (m^3 t^{-1} or m^3); *∑ (*ET_{blue} **R)*: Blue water evapotranspiration (ET_{blue} in mm) according to the proportion *R* of each textural class *k.* Area and yield data were obtained from the Agricultural

Statistics Yearbooks (MARM, 2010a), except for the area of irrigated olive orchards in Andalusia that has been interpolated using the Inventories of Irrigated Land in Andalusia of 1997, 2002 and 2008 (RGA, 1999; 2003; 2008).

B. Grey water footprint of olives

The grey water footprint of a primary crop is an indicator of the degree of freshwater pollution associated with the production of the crop (Hoekstra et al., 2009). As it is generally the case, the production of olives concerns more than one form of pollution. The grey water footprint has been estimated for nitrogen since it is a very dynamic element which can be the source of surface and ground water pollution caused by surface runoff and leaching (Fernández-Escobar, 2007). The grey water footprint for each province *i*, year *j* and production system *l* can be expressed as following:

$$WF_{grey\,ijl}\ m^3\,t^{-1} = \frac{WF_{grey\,ijl}\ million\,m^3 \times 10^6}{Pr_{ijl}} \qquad [4.12]$$

$$WF_{grey\,ijl}\ million\,m^3 = \frac{N_{surp} \times A_{ijl} \times 10^{-3}}{c_{max} - c_{nat}} \qquad [4.13]$$

Where WF_{grey}: grey water footprint (hm^3 or $m^3\ t^{-1}$); *Pr*: crop production (t); N_{surp}: nitrogen surplus (kg ha^{-1}); C_{max}: the maximum acceptable concentration (50 mg NO_3 L^{-1}); C_{nat}: natural concentration in the receiving water body (mg NO_3 L^{-1}); *A*: crop area (ha).

Modifications of the method of Hoekstra et al. (2009) were made since the grey water footprint is calculated based on nitrogen surplus instead of the chemical application rate per hectare times the leaching fraction. Nitrogen surplus i.e. the difference between nitrogen inputs and outputs in agriculture can be a good indicator of potential losses to the environment at global, local or farm scale (European Commision, 2002). Nitrogen balances of 2006 have been used to determine the nitrogen surplus in olive orchards for each province as calculated by the Ministry of the

Environment and Rural and Marine Affairs of Spain (MARM, 2008b). Nitrogen surplus is constant throughout the years for each province and does not differentiate between rainfed and irrigated olives. Thus, the nitrogen balance provides an approximate measure of nitrogen surplus for both olive production systems.

An ambient water quality standard of 50 mg NO_3 L^{-1} of water is used to calculate the water volume necessary to assimilate the load of pollutants following the nitrates and groundwater directives (European Commission, 1991; 2006). The natural concentration of pollutants in the receiving water body has been assumed negligible.

4.2.3. Supply chain water footprint related to other product components

The water footprint of the supply chain of 1 liter of bottled olive oil is not only made up of ingredients but also of other components that form the final product. Other main components of the product are presented in Table 4.2. For the calculation of the water footprint related to other components, raw material and process water requirements are taken into account separately.

Table 4.2. Water footprint of raw material and process water use (m^3 t^{-1}) of other product components.

Components	Raw material	Weight (g)[1]	Water footprint raw material[2]			Process water use[2]		
			Green	Blue	Grey	Green	Blue	Grey
Bottle – PET[3]	Oil	39	0	10	0	0	0	225
Cap – HDPE[4]	Oil	3	0	10	0	0	0	225
Label - PP[5]	Oil	0.3	0	10	0	0	0	225

[1] Source: weight estimated for 1 liter bottle from Ercin et al. (2009).
[2] Source: van der Leeden et al. (1990).
[3] PET: polyethylene terephthalate
[4] HDPE: high-density polyethylene
[5] PP: polypropylene

4.2.4. The water footprint of crop products

The water footprint of crop products (i.e. olive oil) is calculated by dividing the water footprint of the input product (i.e. olives) by the product fraction (Hoekstra et al., 2009; Garrido et al., 2010). The latter is defined as the quantity of the output product obtained per quantity of raw material. In the present study the product fraction calculation is based on the industrial olive oil yield (Ruíz, 2001), which is known as the olive oil obtained per kilogram of milled olives. We have assumed an olive yield content of 22% for normal climate year according to Pastor et al. (1999) with 50% olive moisture content. A product fraction of 19.6% is obtained.

As a result, the water footprint of 1 L olive oil for each province *i*, year *j* and production system *l* can be expressed as follows:

$$WF_{olive\ oil\ ijl} = \frac{WF_{olives\ ijl}}{p_f} \times d \quad + WF_{supply-\ chain\ \ other\ parts}$$

[4.14]

Where $WF_{olive\ oil}$: Water footprint of olive oil (L L^{-1}); WF_{olives}: Water footprint of olives (L kg^{-1}); p_f : Product fraction (%); *d*: density of olive oil (0.918 kg L^{-1}); $WF_{sup.\ chain\ [other\ parts]}$: Water footprint supply chain of other parts (L L^{-1}).

4.2.5. Apparent water productivity of olive oil

The concept of apparent water productivity (AWP, in € m^{-3}) is used to assess the economic efficiency of the water consumed per ton of olive oil produced. Market prices for each province are determined based on the production and price of the three types of virgin olive oil: extra, fine and normal virgin olive oil (MARM, 2010a). For province *i*, in year *j* and under production system *l* the AWP was calculated as follows:

$$AWP_{ijl} = \frac{P_m \times V_{m\ \ ij}}{WF_{olive\ oil\ ijl}}$$

[4.15]

Where $\sum (P_m \times V_m)$: market price (*P* in € t^{-1}) according to the proportion *V* of the type of virgin olive oil production *m*; *WF olive oil*: water footprint of olive oil (m^3 t^{-1}).

4.2.6. Virtual water exports of olive oil

The olive oil virtual water exports indicate the water embedded in exports (E_{ij}, t year^{-1}). The green (VWE_{green}, hm^3 year^{-1}) and blue (VWE_{blue}, hm^3 year^{-1}) for each province *i* in year *j* have been analyzed as follows:

$$VWE_{green\,ij} = WF_{green\,ij}\ m^3\ t^{-1}\ \times E_{ij} \times 10^{-6} \quad [4.16]$$

$$VWE_{blue\,ij} = WF_{blue\,ij}\ m^3\ t^{-1}\ \times E_{ij} \times 10^{-6} \quad [4.17]$$

Main olive oil producing provinces do not match with the major olive oil internationally exporting provinces, because of internal trade within Spain. Virtual water exports of olive oil are based on the share of the production of each province to the national olive oil production, in order to take into account where the olive oil production comes from.

4.3. Results

4.3.1. Water footprint of olives orchards

For the studied period, Spain shows the following average water footprint: 7,890 hm^3 green water footprint (rainfed), 1,400 hm^3 green water footprint (irrigated), 710 hm^3 blue water footprint and 1,070 hm^3 grey water footprint. The main factors influencing the water footprint in absolute terms (hm^3) are crop area, rainfall and irrigation volume. As shown in Figure 4.2, in the analyzed period there is a clear upward trend of total water footprint. This trend is due to the increase of olive orchards from 2,157,600 ha in 1997 to 2,450,500 ha in 2008, since the volume of precipitation at the end of the period is lower than one in 1997. It is noteworthy to mention than 70% of the olive orchards expansion during the period are irrigated. The green water footprint of rainfed olives is significantly larger than the irrigated one, because the former comprises from 7.4 to 3.5 times the irrigated area, at the beginning and end of the period of study. In the case of the grey water footprint, variations rely uniquely on the area expansion because the same value of nitrogen surplus has been used for each year. There seems to be a correlation between the total annual water footprint and yearly rainfall, but the effective rainfall is higher in rainfed orchards than in irrigated ones. The lowest annual rainfall in 2005 (with 430 mm) is clearly reflected in the decrease of the green water footprint both under rainfed and irrigated conditions. The blue water footprint dropped in 2000 and 2006-2008 owing to the estimated water allowance of 1,600 m^3 ha^{-1} for the mentioned years due to the drought situation prevailing in the Guadalquivir basin.

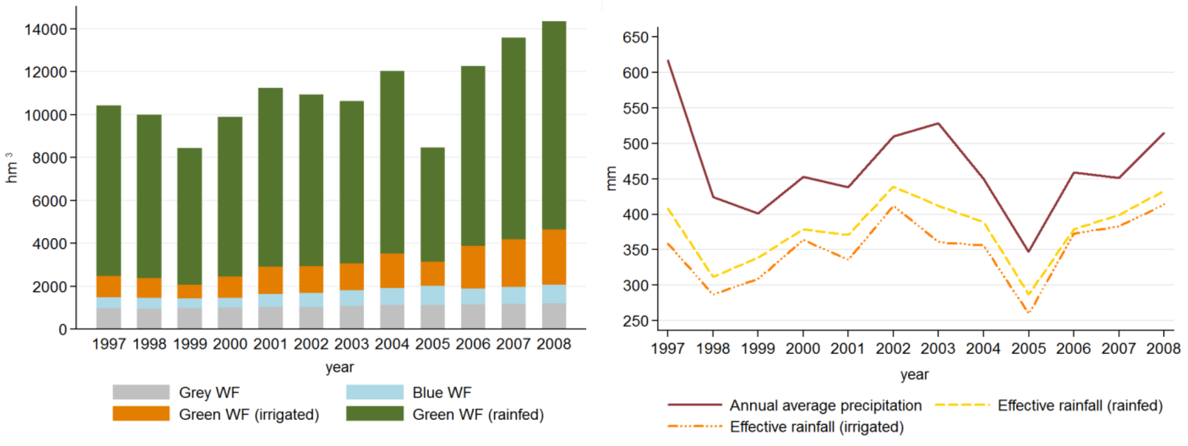

Figure 4.2. Total green, blue and grey water footprints of olive production in Spain in hm^3 (left) and annual average rainfall and effective rainfall in mm (right) for the period 1997-2008.

Within the study period, Andalusia comprises 87% of the national blue water footprint of olive production, with 761 hm^3 in 2008. However, only 13% of the national blue water footprint of olives was allocated to olive table production in that year. Sevilla is by far the most important table olives producing province, consuming 82 hm^3 of blue water, 64% of the blue water footprint of olive production within the province. Surface water irrigation for olive orchards decreased in Andalusia from 66 to 43% in relation to the national blue water footprint over the study period. In contrast, groundwater resources have been increasingly consumed from 19 to 43%, growing groundwater abstractions from 106 hm^3 (1997) to 378 hm^3 (2008). Jaén is the first blue water consumer in Andalusia, and also in Spain with 401 hm^3 in 2008, of which 99% belongs to olives for olive oil production. Between 1997 and 2008, surface water consumption moderately decreased and groundwater resources consumption more than doubled in the province (Figure 4.3). In fact, in 2008 most provinces increased groundwater consumption for olive production with the exception of Almería.

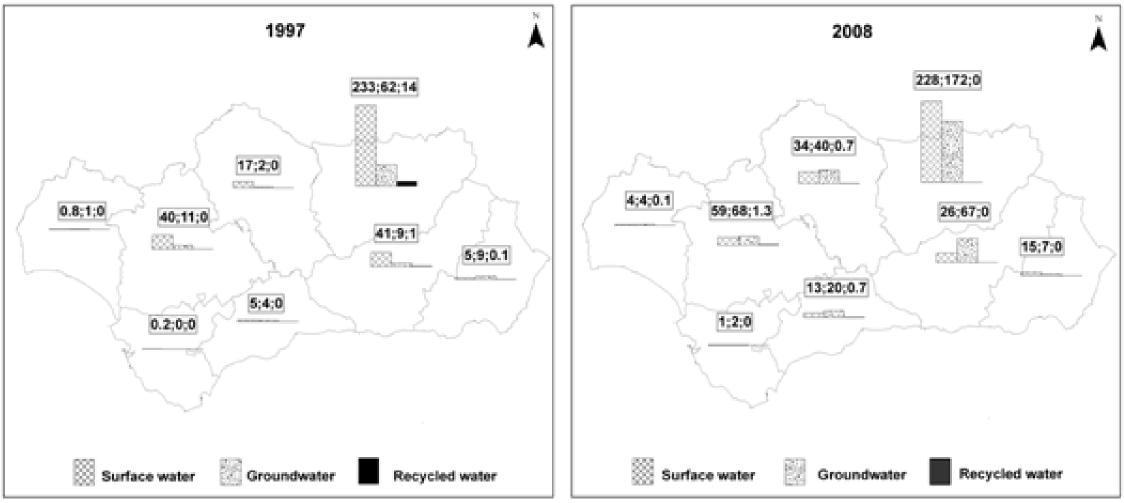

Figure 4.3. Origin of blue water footprint: surface, groundwater and recycled in hm^3 for 1997 (left) and 2008 (right) in Andalusia. Source: own elaboration based on the Inventory and Characterization of Irrigated Land in Andalusia of 1997, 2002 and 2008 (Regional Government of Andalusia, 1999, 2003, 2008).

The water footprint in m^3 t^{-1} is an indicator of the crop's blue and green water efficiency per unit of crop produced. In addition the grey water footprint in relative terms illustrate the estimated volume of water contaminated by nitrates per unit of crop produced, which can indicate the nitrate pollution potential. Higher crop water efficiency and less nitrates pollution potential are associated with lower footprints. For the studied period, Spain presents the following average water footprint per unit: 1,971 m^3 t^{-1} green water footprint (rainfed), 856 m^3 t^{-1} green water footprint (irrigated), 408 m^3 t^{-1} blue water footprint and 190 m^3 t^{-1} grey water footprint.

Figure 4.4 compares the total water footprint and the water footprint per unit of crop for the main olive producing provinces for an average rainfall year (2001). Only provinces that comprise ≥1% of the national olive production in 2001 are illustrated. In 2001, Jaén, Córdoba and Sevilla jointly represent 69% of the national olive production and 52 % of the national water footprint of olive production with 3,199; 1,457 and 874 hm^3 respectively. While their total water footprints in hm^3 are the largest, they are very efficient in terms of green and blue water use (m^3 t^{-1}). Based on the nitrogen balances, olive production in Jaén, Córdoba and Sevilla does not generate any grey water footprint. In absolute terms, Granada presents the largest grey water footprint with 273 hm^3. However, the provinces showing the highest nitrogen pollution per ton of crop produced are minor olive producers such as Lleida, Albacete and Toledo.

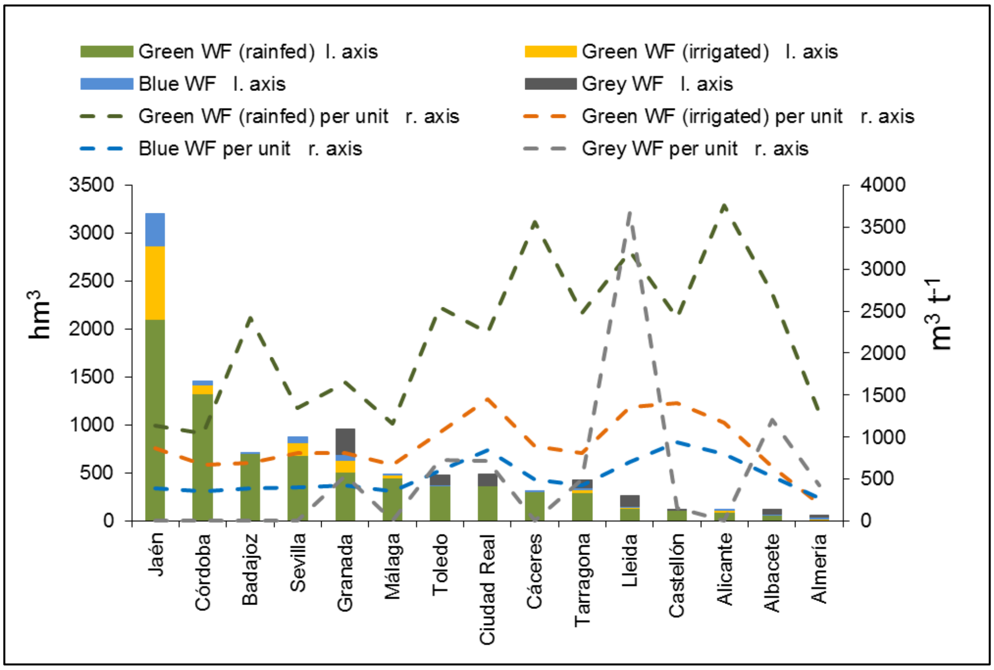

Figure 4.4. Green, blue and grey water footprints in hm^3 and $m^3\ t^{-1}$ of the main Spanish olive producing provinces in 2001.

4.3.2. Water footprint of olive oil

The water footprint of olive oil includes the sum of the water footprint of the ingredients and other components i.e. the supply chain water footprint. The water footprint related to other components for olive oil production does not represent more than 0.5% of the total supply chain for each year and province of study. In conclusion, most of the water used (consumed and polluted) to produce olive oil can be directly associated to olive production in the field. Table 4.3 presents the water footprint of olive oil in hm^3 during the period of study. In color terms, the components of the water footprint can be summarized as follows: 72% green water footprint from rainfed systems, 12% green water footprint from irrigated ones, 6% blue water footprint and 10% grey water footprint.

Table 4.3. National water footprint of olive oil in hm^3 during the period of study.

Year	**1997**	**1998**	**1999**	**2000**	**2001**	**2002**	**2003**	**2004**	**2005**	**2006**	**2007**	**2008**
Ingredients (olive fruit)												
Green WF (R)	7560	7207	5942	6903	7824	7535	7092	8016	5046	7836	8852	9119
Green WF (I)	909	851	553	905	1175	1140	1133	1395	992	1722	1929	2204
Blue WF (I)	470	463	404	413	574	596	652	712	788	651	703	770
Grey WF (R & I)	967	943	978	993	1010	1024	1078	1115	1130	1151	1159	1207
Other components (bottle, cap and label)												
Blue WF (R)	0.4	0.3	0.2	0.3	0.5	0.3	0.5	0.3	0.2	0.3	0.4	0.3
Blue WF (I)	0.1	0.1	0.1	0.1	0.1	0.1	0.2	0.1	0.1	0.2	0.2	0.2
Grey WF (R)	9.7	6.6	4.8	7.7	10.5	6.1	10.7	7	5.2	7.6	8.3	7.6
Grey WF (I)	2.1	1.9	1.4	2.2	2.9	2.3	3.9	3	2.7	4.1	4.7	4.6
Total	9918	9473	7883	9225	1059	1030	9970	1124	7964	1137	1265	1331

R: rainfed. I: irrigated

Spain has the following annual ranges of the water footprint per liter of olive oil produced: 8,250-13,470 L L^{-1} green water footprint (rainfed); 2,770-4,640 L L^{-1} green water footprint (irrigated); 1,410-2,760 L L^{-1} blue water footprint (irrigated); and 710-1,510 L L^{-1} grey water footprint (rainfed & irrigated). These ranges are weighted averages according to the share of each province to the national production. The blue water footprint of other components in rainfed olives has a negligible value of 0.4 L L^{-1}.

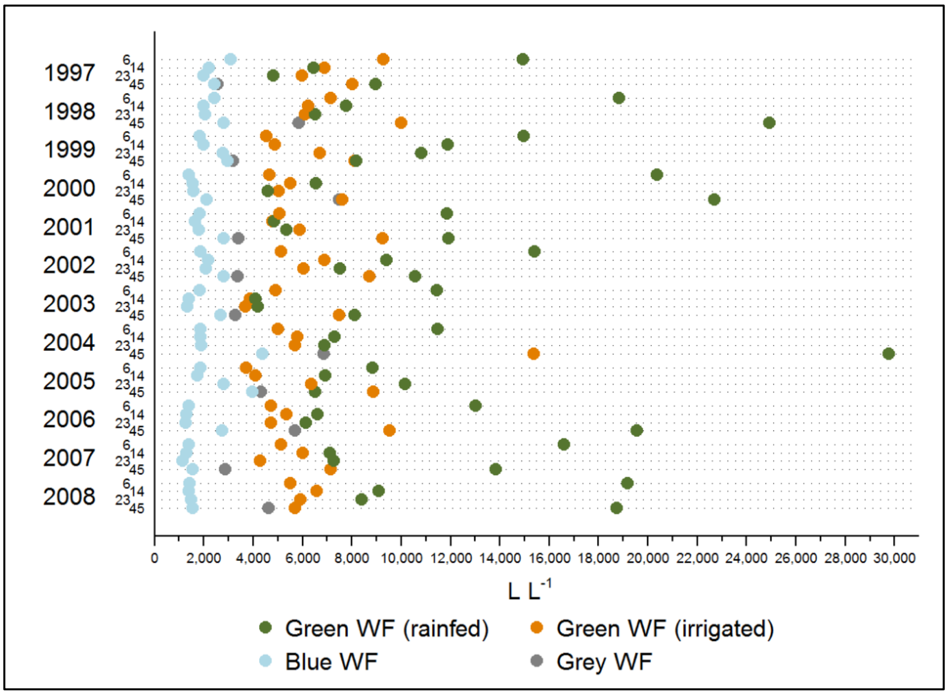

Figure 4.5. The water footprint of olive oil in L L^{-1} for four typical producing provinces. Provinces are coded as follows: 6 = Badajoz, 14 = Córdoba, 23 = Jaén and 45 = Toledo.

The water footprint in L L^{-1} is summarized for four typical olive oil producing provinces in Spain (Figure 4.5). In this figure, the blue and grey water footprints of other components are not included in their respective color components. The greater variation of the green water footprint in L L^{-1} of rainfed olives over the study period is remarkable, ranging from 4 100 L L^{-1} in Córdoba in 2003 up to 29 760 L L^{-1} in Toledo in 2004. This large variation of the water footprint in rainfed olive oil also occurs for the total of the provinces, as shown in Table 4.4. Among provinces with rainfed olive oil, Jaén and Córdoba are more efficient in terms of water consumption than Badajoz and Toledo. The green water footprint of olive oil produced from irrigated orchards exhibits less variation, with a minimum of 1,861 (Badajoz in 2005) and a maximum of 8,688 L L^{-1} (Toledo in 2004), probably because crop production is not so strongly affected by rainfall. From the selected provinces, Toledo is the only one that presents a grey water footprint ranging from 2,573 in 1997 to 7,484 L L^{-1}.

Table 4.4. Weighted average water footprint in L L^{-1}, according to the share of each province to the national production, and standard deviation (std.) of total water footprint (green + blue + grey components) for rainfed and irrigated olive oil.

	Total WF rainfed (L L^{-1})		**Total WF irrigated (L L^{-1})**	
Year	**Mean**	**std.**	**Mean**	**std.**
1997	8,253	36,467	6,730	7,331
1998	11,711	56,243	6,845	10,253
1999	13,468	49,429	7,338	6,198
2000	9,661	69,656	6,219	10,804
2001	7,881	74,503	6,234	5,903
2002	13,064	91,961	7,651	59,839
2003	7,129	52,331	4,751	20,973
2004	12,144	26,204	7,133	9,810
2005	10,937	29,474	6,862	11,193
2006	10,985	17,937	5,939	5,151
2007	11,273	30,293	5,742	5,548
2008	12,702	20,636	6,690	6,248

4.3.3. Apparent water productivity of olive

Two typical producing provinces, Jaén and Toledo, were compared to analyze the AWP (€ m^{-3}) of olive oil (Figure 4.6). The AWP is inversely related to the water footprint per unit of olive oil and follows a similar pattern over the period in both production systems, due to the variation of olive oil market prices. In rainfed systems, the AWP of olive oil presents the lowest values. The AWP ranges from 0.20 to 0.62 € m^{-3} in Jaén and from € 0.07 to 0.36 m^{-3} in Toledo. The AWP of irrigated systems has a relatively stable trend between 1997 and 2005 with values below 2.4 and 1.7 € m^{-3} in Jaén and Toledo, respectively. The peaks of AWPs in 2006 and 2007 are related to highest olive oil prices of 4,119 (2006) and 4,868 (2007) € t^{-1} in Jaén and 5,525 (2006) and 5,436 (2007) € t^{-1} in Toledo, as well as the reduced water allowances for irrigation during this years (see Table 4.1). Greater olive oil prices in Toledo are caused by its relatively larger production of virgin olive oil of premium quality.

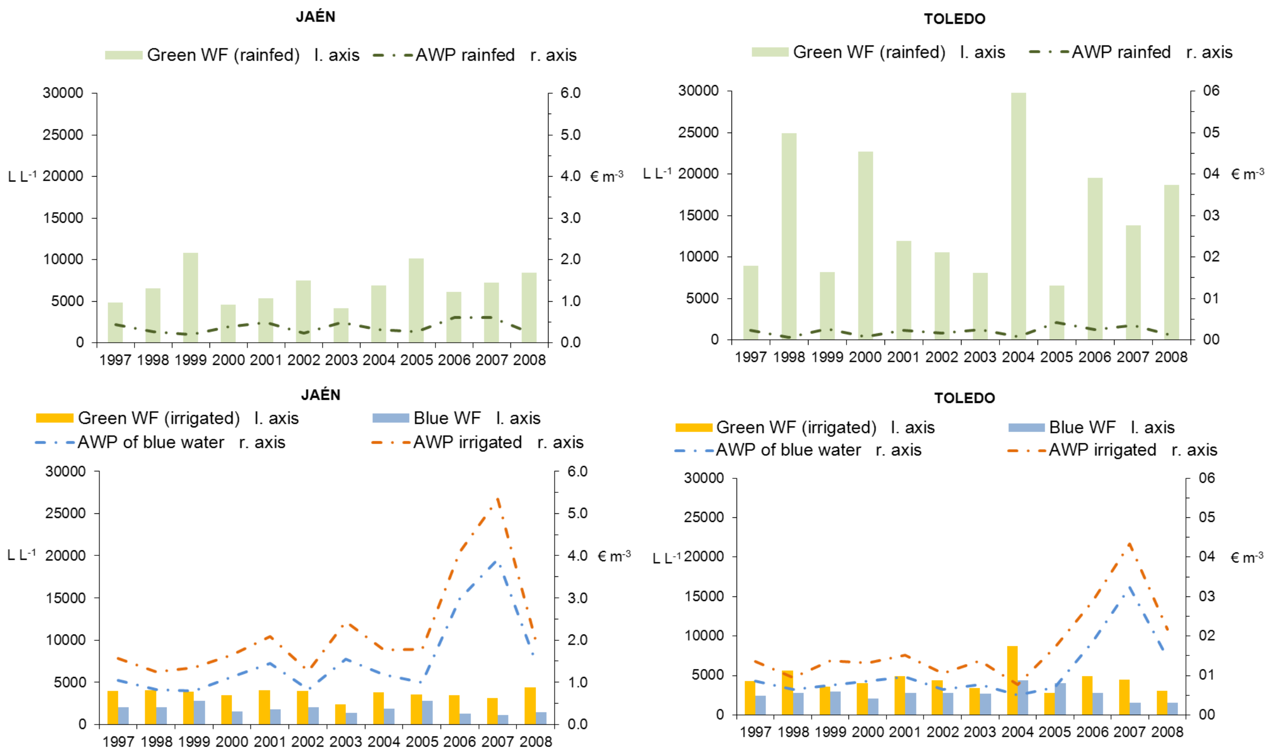

Figure 4.6. Olive oil water footprint and apparent water productivities for rainfed (top) and irrigated (bottom) production systems in Jaén (left) and Toledo (right) provinces over the period 1997-2008.

4.3.4. Virtual water exports of olive oil

According to the information from the Olive Oil Agency (OOA, 2010) exports comprise 55% of the total national olive oil production between 2005/2006 and 2007/2008 agricultural seasons. As shown in Figure 4.7 the olive oil exports and total virtual embedded tend to increase during the period 1997-2008. Differences between production and exports of olive oil are based on the final stocks of each agricultural season. For instance, in 2002 olive oil production was not significantly large but final stocks of the olive oil produced in 2001 were exported. Then the fall of olive oil production in 2002 is reflected in the decline of exports in the following year (Figure 4.7).

The green water is the main component in virtual water exports, amounting to 79% of the total virtual water exports during the study period. Differences among years are significant, green water being the most unstable component and closely dependent on precipitation. Note, that blue virtual water exports are much more stable. Rainfed olives therefore have an important role in virtual water exports, even if both the area of irrigated olive trees and the related blue water component have increased during the period of study.

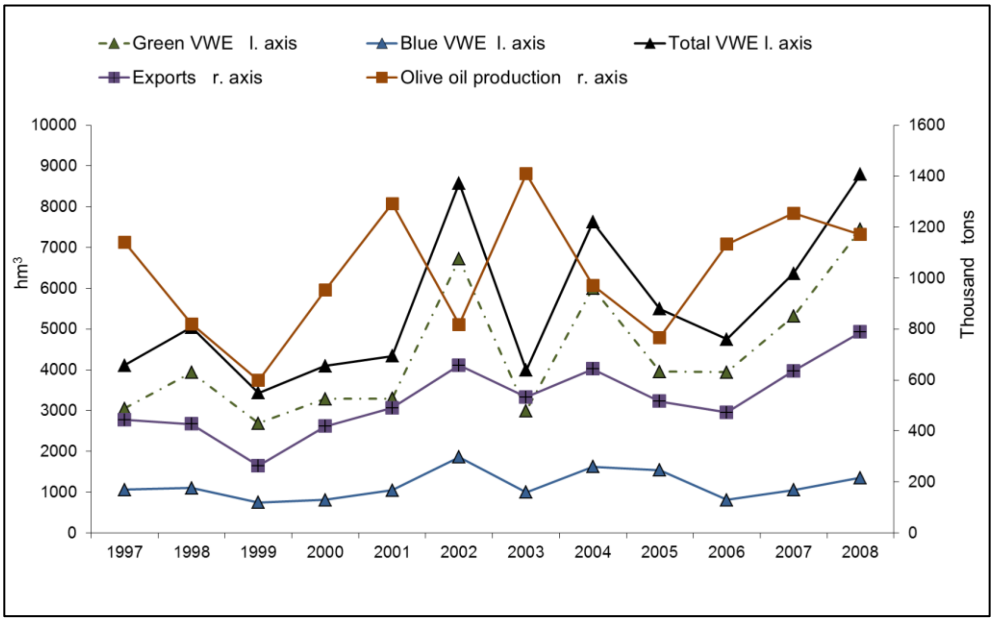

Figure 4.7. Green, blue and total virtual water exports (VWE in hm^3), exports (thousand tons) and production (thousand tons) for olive oil over the period 1997-2008.

4.4. Discussion

Over the studied period, the Spanish olive fruit production presents the following average percentage of water footprints: 72% green water footprint from rainfed systems, 12% green water footprint from irrigated ones, 6% blue water footprint and 10% grey water footprint. If we compare the WF of olives with figures at national level of the WF for crop production (Mekonnen and Hoekstra, 2011), the importance of their production is shown in terms of water appropriation and nitrates pollution level. The olive fruit production accounts for 20% and 5% of the green and blue water respectively, and 13% of the Spanish grey WF for crop production. For the studied period, Spain has an average national water footprint (without including the grey component) of 9,960 hm^3 which is in contrast to that estimated by Garrido et al. (2010) of approximately 2,800 hm^3. In the present study, the green water footprint is larger since soil is taken into account applying a soil water balance. In comparison to the blue water, the green component of the rainfed olives constitutes the largest proportion of the water footprint of olive orchards due to the greater harvested area of this production system. Moreover, according to our CROPWAT results effective rainfall is higher in rainfed orchards than in irrigated ones, since the irrigation application lowers the green water evapotranspired. When rainfall and irrigation occur in the same day, CROPWAT assumes that crops firstly evapotranspired the irrigation amount. Later, the part of the rainfall that is effectively used by the crop after rainfall losses (i.e. surface run off and deep percolation) is accounted for.

The water footprint per unit of fruit produced can illustrate the efficiency of water consumption in relation to crop production. In our study, the water footprint per unit of rainfed olive orchards (green) is usually higher (about 1,971 $m^3\ t^{-1}$) than the irrigated one (green plus blue) (around 1,264 $m^3\ t^{-1}$) due to lower crop yields. In rainfed olive trees, the rainfall and temperature patterns contribute to the fruit production, whereas irrigated olive orchard production depends mainly on temperature since water stress is usually avoided by the irrigation water supply (Lavee, 2007). Despite the fact that rainfed olive systems depends only on precipitation, low yields per hectare of rainfed olives seem to point to a transition towards more productive olive orchards. However, problems related to diversity losses and environmental pressures arise with more intensive agricultural systems of olive orchards (MARM, 2007; Scheidel and Krausmann, 2011) and rainfed olive production does not use scarce blue water resources.

Aldaya and Llamas (2008) estimated the green and blue water footprint per unit of production in the Guadiana river basin. In line with their study, in 2001 the green water footprint has a value of 600 m^3 t^{-1} and 210 m^3 t^{-1} for rainfed and irrigated systems respectively, and a value of 750 m^3 t^{-1} of blue water footprint in the Middle Guadiana basin, which contains Badajoz and Cáceres provinces. The present study shows under rainfed conditions significantly greater green water footprints (2,280 and 3,500 m^3 t^{-1} for Badajoz and Cáceres). Irrigated systems indicate higher green water footprints (700 and 900 m^3 t^{-1} for Badajoz and Cáceres) and lower blue water footprints (430 and 540 m^3 t^{-1} for Badajoz and Cáceres) in 2001, in spite of using the same crop yields in both studies. These differences in the results could be due to methodological improvements: the present study takes into account different soil textures and does not assume optimal irrigation conditions. In any case, we should also bear in mind that the scale of our study is larger than in the case of Aldaya and Llamas (2008), which could lead to greater dispersion of the results. For future studies, considerations also need to be taken on the implication of changing climate conditions on the response of olive yield and economic losses (Ponti et al., 2014).

The average water footprints ranges for 1 L of olive oil are: 8,250-13,470 L L^{-1} green water footprint (rainfed), 2,770-4,640 L L^{-1} green water footprint (irrigated), 1,410-2,760 L L^{-1} blue water footprint (irrigated) and 710-1,510 L L^{-1} grey water footprint. The lion's share of the water footprint of 1 L of bottled olive oil is in the supply chain, and in particular in the olive production process. In fact, the water footprint of the supply chain of other components (bottle, cap and label) comprises less than 0.5% of the product's water footprint, which is in line with previous studies (Ercin et al., 2009). The results of this study confirm the importance of a detailed water footprint supply chain assessment of ingredients in the case of agriculture based products. The variability of the water footprint of 1 L olive oil among provinces depends mainly on the type of production system and year. The value of 15,831 m^3 t^{-1} provided in Chapagain and Hoekstra (2004b) during the period 1997-2001 for virgin olive oil in Spain, which is equivalent to 14,533 L L^{-1}, is significantly larger than those obtained in this study, particularly in irrigated conditions. This is probably because they assumed that the crop water requirements are met in olives trees.

Based on the grey water footprint results, the main olive oil producing provinces do not seem to represent significant sources of nitrate pollution with the exception of Granada. For instance, within the Guadalquivir basin, Guadajoz and Jaén catchments show the lowest nitrogen surplus per hectare in the basin due to olive orchards land use (Berbel and Gutiérrez, 2004). Jaén, the first olive oil producing province in Spain, presents a negative nitrogen balance (MARM, 2008b), suggesting that applications (mostly as mineral and organic fertilization) do not compensate the losses. Nevertheless, nitrogen inputs of irrigated olives are nearly three times higher than rainfed ones (IDAE, 2007). Consequently, a nitrogen balance that differentiates between these two olive production systems would yield more accurate evaluations of the grey water footprint. Further research of grey water footprint also needs to focus both on spatial and temporal variation of pollutants since higher concentrations of nitrates in water bodies would be expected after fertilization application followed by rainfall gages (Rodríguez-Liziana et al., 2005) and in the dry season than in the wet one (Angelopoulos et al., 2009).

The olive oil production has increased its apparent water productivity during 1997-2008 incentivized by growing trade prices, and also did the amount of virtual water exports. Compared to the Spanish virtual water exports for crop production (Mekonnen and Hoekstra, 2011), our results show that olive oil accounts for 32% and 15% of the green and blue virtual water exports (26% considering jointly green and blue water). These figures only take into account one crop within the country, but are well above of the 13% global average of the water for crop production that ends as virtual water export (Hoekstra and Hung, 2005). The olive oil virtual water exports vary across years, and mostly depend on the green water, which denotes the importance of the green water in the virtual water trade, as reported in previous studies (Aldaya et al., 2010). In the Spanish olive oil production, 23% of the virtual water exports correspond to irrigation. Andalusia is the largest blue water consumer region for olive production and in 2008 groundwater resources reached a value of 42% of the national blue water consumption for olive production. Groundwater abstractions have grown from 98 hm^3 in 1997 to 378 hm^3 in 2008 in Andalusia. The increasing groundwater use is in a way related to the blue virtual water exports of olive oil. Even during the first globalization period (1849-1935), the growing incorporation of Spain into international markets as a major exporter of Mediterranean products substantially increased pressure on water resources and required the expansion of irrigation (Duarte et al., 2014). In an irrigated district of Córdoba, 18% of farmers consider olive trees as an alternative to current cropping patterns (García-Vila et al., 2008). As a result, further development of this crop in irrigated systems may be expected in the coming years. Our results suggest that virtual groundwater exports related to olive oil exports may add further pressure to the already stressed basin. Although, the situation seems to be under control given the deficit irrigation practices and the constraints imposed by the sharp increase in energy prices (Salmoral and Chico, 2012).

Globally, several studies have accounted for the WF of agricultural and food products, but they usually assumes that crop water requirements are fully met under irrigation (Casado et al., 2008; Mekonnen and Hoekstra, 2011; Vanham, 2013). However, in Spain crops such as wheat, vineyards and olive orchards do not usually meet their irrigation water requirements because of the high-energy cost that would lead particularly in groundwater sources and also because of the irrigations limits that the River Basin Authorities impose for these crops. In this study, the irrigation water for olive orchards is calculated based on the water allowances and modified according to existing level of drought in the Guadalquivir river basin. Even if there are great uncertainties on their compliance, potential errors remain low in comparison with assuming that crop water requirements are fully satisfied. Nevertheless, improvements on the blue water assessment of olives can be performed since the estimated water allowances of olives vary among systems of exploitation and also depending on regulated, un-regulated and groundwater sources. Moreover, the scale of our study does not enable to take into account farmers' decisions that consider the precipitation during irrigation management, assuming that rainfall is sufficient and reducing their irrigation schedules (García-Vila et al., 2008).

4.5. Conclusions

According to this study, the largest producing places show high water use efficiency per product (m^3 t^{-1}) and apparent water productivity (€ m^{-3}) as well as less nitrates pollution potential (grey water in (m^3 t^{-1}). However, the land use changes with the expansion of irrigated olive orchards have enhanced the pressure on available water resources. The olive production has increased its apparent water productivity during 1997-2008 incentivized by growing trade prices, but also did the amount of virtual water exports. Olive production, and subsequently water, is linked to trends of international markets. Although 77% of virtual water exports for olive oil are related to the green WF, this growth took place due to increasing groundwater abstractions in the Upper Guadalquivir basin and has caused concerns about the sustainability of olive irrigation in those areas. Our results suggest that virtual groundwater exports related to olive oil exports may add further pressure to the already stressed basin.

In our study, the water footprint of olive oil has been estimated taking into account variables such as soil type, production system and variation over the time of climate conditions and water allowances. Evaluations change significantly from year to year because most production is obtained in rainfed systems whose production depends on a greater extent on precipitation, than irrigated system does. It is not possible to provide a unique value in relative terms of water footprint for olives and consequently for olive oil in Spain since there are widely different production systems, productivity levels and irrigation management. All these aspects can be put into context with further local olive production studies. Moreover, there are other factors such as plantation density of trees, volume of crown and volume and timing of irrigation that could not be taken into account in the present analysis. Further studies at local scale considering these elements could make improvements in this area. In addition, further assessment of the economic, social and environmental aspects of olive fruit production and effects of climate change could provide additional information for decision-making towards sustainable land use and water resource planning.

5. Drivers influencing streamflow changes in the Upper Turia basin, Spain

5.1. Introduction

Since the 1950s, many basins across the Mediterranean region have experienced a significant streamflow reduction (Giakoumakis and Baloutsos, 1997; Cigizoglu et al., 2005; Lespinas et al., 2009; García-Ruiz et al., 2011). Streamflow decrease has been reported on annual basis, but also seasonally, along the winter, spring and summer (Ceballos-Barbancho et al. 2008; Morán-Tejeda et al. 2011; Lorenzo-Lacruz et al., 2012; Martinez-Fernandez et al. 2013). Climate change (CC), and specifically precipitation reduction and increasing temperature, is a critical factor underpinning streamflow reduction (Bangash et al. 2013; IPCC, 2014). But additional drivers also need to be considered when understanding hydrological changes within basins i.e. land use and land cover changes (LULCC) (Delgado et al., 2010; Gallart et al., 2011; Iroumé and Palacios, 2013), the introduction of exotic species (Little et al., 2009; Vose et al., 2011), damming, inter-basin water transfers and intensive water use especially for irrigation (Milliman et al., 2008; Aus der Beek et al., 2011).

There is ample evidence that changing climate conditions across many regions of Spain are exerting a major role on the hydrological response of many basins (CEDEX, 2011; Lorenzo-Lacruz et al. 2012; Martínez-Fernández et al., 2013). However, several studies have reported that up to 25% of the streamflow decrease during the last decades can be attributed to ongoing LULCC (Beguería et al., 2003; Gallart and Llorens, 2004; Willaarts, 2012). Most of the observed LULCC across Iberian basins include an expansion in forest cover across the middle sections and headwaters, resulting from the progressive agricultural abandonment in both mountain and semi-arid areas, and the intensification of the lowlands for agriculture and urban development (Poyatos et al., 2003; Pinto-Correia and Vos, 2004; Lasanta-Martínez et al., 2005; Plieninger and Schaar, 2008; Rescia et al., 2010; Martínez-Fernández et al., 2013). The drivers of these landscape trends are diverse and complex, including the progressive depopulation of rural areas, Spanish industrial development and urbanization trend, changes of agricultural and environmental policies and ongoing globalization of food markets (Verburg et al., 2006; Westhoek et al., 2006; García-Ruiz, 2010; Lasanta and Vicente-Serrano, 2012; Viaggi et al., 2013).

The extent to which the hydrological response of a basin is likely to change due to LULCC depends on the magnitude of the shift taking place, the basin's physical conditions (e.g. climate, catchment size, slope, soil characteristics) (Peel et al., 2010; Nainggolan et al., 2012) and the ecological characteristics, i.e. vegetation type as well as plants' physiological and structural characteristics (Donohue et al., 2007). In large basins (>1000 km^2) the macro-climate is the principle determinant of evaporation flux (*ibid*). However, the impact of LULCC on evapotranspiration increases as catchment area decreases, and the incorporation of vegetation effects is expected to enhance the prediction of the hydrological response capacity (Donohue et al., 2007; Peel et al., 2010). A growth in forest cover usually has a negative effect on the water yield (Sahin and Hall, 1996; Andréassian, 2004; Farley et al., 2005; Chappell and Tych, 2012; Brown et al., 2013), whereas a decrease in forest cover increases runoff and associated risks of floods and soil erosion (Anderson et al., 1976; Leyer et al., 2012; Vieira et al., 2014). Vegetation type and cover is also an important factor explaining streamflow changes. Bosch and Hewlett (1982) found that in a temperate climate, the water yield rises as vegetation cover reduces from evergreen coniferous forest to deciduous hardwood, shrubland and grass vegetation types.

The hydrological impact of changing vegetation within basins is also controlled by the prevailing climate conditions. In water-limited landscapes, i.e. semi-arid to arid regions, the augmentation of streamflow through vegetation management might present little potential for success (Newman et al., 2006). Instead, water yield changes ascribed to LULCC are greater in high-rainfall areas (Bosch and Hewlett, 1982; Zhang et al., 2001). Nevertheless, there are still chances to alter the hydrological response under water scarce regions. For example, in an arid environment like Texas, an increase in the water yield from 105 to 165 mm has been recorded for areas after brush removal (TSSWCB, 2003). Similarly, Cosandey et al. (2005) also reported an annual streamflow increase of 10% after the clear-cutting of Mediterranean forest during the years immediately following the cut.

CC projections in Spain predict that larger reductions in mean annual streamflow are likely to occur in those basins, which are currently more vulnerable to water stress (Estrela et al., 2012). In the face of future CC and the likely reduction of water availability to satisfy the increasing water demand, better understanding is needed on the effects of LULCC on streamflow response. Particularly, because innovative solutions are being called for optimizing the use of scarcer available water resources, in order to meet societal demands without compromising the provision of critical ecosystem services (Vose et al., 2011; Curmi et al., 2013). In addition, gaining insight on the links between land use and water management is of critical importance to support the alignment and further integration of water and land policy goals (Falkenmark and Rockström, 2006; Willaarts et al., 2012).

Along the second half of the twentieth century an overall streamflow reduction has been observed in the headwaters of the Júcar river demarcation (eastern Spain) (Estrela et al., 2012; Lorenzo-Lacruz et al., 2012; Martínez-Fernández et al., 2013). CC conditions have been reported as major drivers, but the role that ongoing LULCC might have exerted on the basin's water yield remains unclear. The main objective of this chapter is to examine the influence of the LULCC on the recorded trend of streamflow reduction over the period 1973-2008 in one of the headwaters of the Júcar river demarcation, the so-called Upper Turia basin. The chapter is structured as follows. Section 5.2 describes the methodology and data used in our study, giving emphasis to the observed climatic, streamflow and LULCC trends found in the Upper Turia basin.

Section 5.3 presents the results that identify the role of LULCC on the hydrologic response, Section 5.4 discusses the main results, and lastly Section 5.5 outlines the conclusions.

5.2. Materials and methods

5.2.1. Catchment description

A. Environmental characteristics

The Upper Turia basin (UTB) has a drainage area of 3936 km^2 and accounts for 9.2% of the Júcar river demarcation (Figure 5.1). Despite its small area, the UTB is one of the main sources of drinking water to the downstream city of Valencia and its nearly 792,000 inhabitants. The UTB itself is a sparsely populated area, with Teruel city as the largest urban settlement with 35,600 inhabitants in 2011 (INE, 2013). Elevation ranges from 540 to 2,020 m. Climate is typically Mediterranean, with an average annual precipitation of 388 mm (AEMET, 2013).

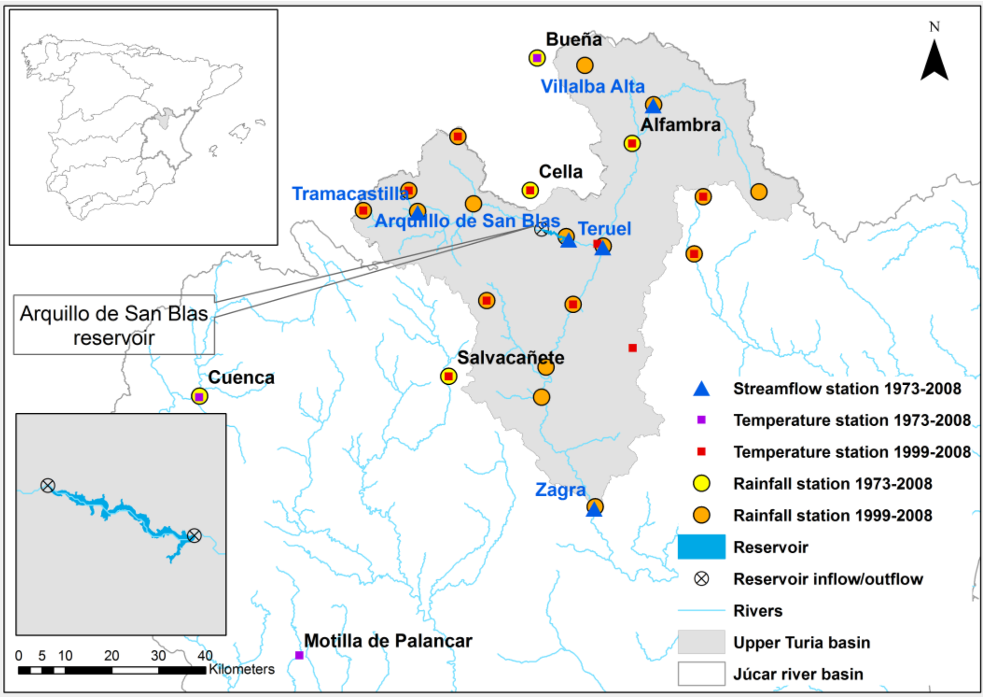

Figure 5.1. Location of the Upper Turia basin and distribution of the climate and streamflow monitoring stations.

The annual thermal oscillation presents continental characteristics, with cold and long winters not surpassing 4.5 ºC as average temperature, and short and mild summers with an average of 21.2 ºC (*ibid*). A reservoir, named Arquillo San Blas with 22 hm^3, is located on the central-north of the UTB and was put into operation in 1967. Most representative soils include Calcic Cambisol, Dystric Cambisol, Humic Cambisol, Calcaric Phaozem, Calcaric Regosol, Eutric Regosol, Leptic Podzol and Ranker soils (Trueba et al., 1999). Soil depth ranges from 450 to 1200 mm with an average of 630 mm (*ibid*).

B. Streamflow and climatic trends

The UTB is relatively well monitored (see Figure 5.1), allowing for the study of the streamflow and climatic trends over a period of four decades (1973-2008). Table 5.1 shows the annual time trend analysis for five streamflow gauging stations (two under natural regime and three under regulated flow), six precipitation stations, three temperature stations and one reservoir inflow[5]/outflow station. Under a natural flow regime, only one gauge station (Villalba Alta) has a relative significant negative trend. Two out of three regulated gauge stations (Arquillo de San Blas and Teruel), located downstream of the reservoir, present also a significant negative trend. Regarding the impacts of reservoir management on the downstream stream, Lorenzo-Lacruz et al. (2012) found that river regulation by dams does not affect the sign of the river flow trends when the regulatory capacity does not exceed the inflows. However, the sign and magnitude of change can be affected with important seasonal differences in highly regulated rivers, by retaining water during the wet season (winter and spring) in order to guarantee the water supply for irrigation and urban consumption during the dry season (Lorenzo-Lacruz et al., 2012; Belmar et al., 2013). In the UTB inflow and outflow of the reservoir also indicate a negative trend, but there is a slightly less negative magnitude at the outflow of the reservoir (-0.92 hm^3 $year^{-1}$) in comparison to the inflow (-1.09 hm^3 $year^{-1}$). This supports that reservoir management does not represent a significant driver of annual streamflow reduction downstream and has even weakened the negative trend downstream. All temperature stations reveal a

[5] MAGRAMA (2013) estimates the reservoir inflow as the difference between the storage variation capacity minus the outflow.

significant positive trend ranging between +0.04 and +0.1 ºC year^{-1}. Only one precipitation station (Salvacañete) presents a significant negative trend. These figures indicate that increasing mean temperature might be one driver responsible for the existing negative river flow trends. Precipitation does not contribute significantly, since Salvacañete station is not the closest precipitation to any of the streamflow stations with significant trend.

Table 5.1. Significance of the annual trends with Kendall rank correlation coefficient (τ) for precipitation, temperature, streamflow under natural regime, reservoir inflow, reservoir outflow and regulated streamflow during the period 1973-2008. Magnitude of change for significant trends (p<0.05) is shown with Sen's slope.

Station	Name	τ [(1)]	p-value τ [(1)]	Sen's Slope [95% confidence interval] [(2)]
precipitation	Cuenca	-0.072	0.551	
	Motilla de Palancar [(3)]	-0.067	0.610	
	Bueña	-0.116	0.334	
	Salvacañete [(4)]	-0.254	0.043	-5.34 [-10.45, +0.02] mm year^{-1}
	Alfambra [(5)]	0.050	0.709	
	Cella	0.019	0.881	
temperature	Cuenca	0.630	0.000	+0.05 [0.05, 0.06] ºC year^{-1}
	Motilla de Palancar [(6)]	0.639	0.000	+0.10 [0.06, 0.13] ºC year^{-1}
	Bueña [(7)]	0.455	0.000	+0.04 [0.02, 0.06] ºC year^{-1}
natural streamflow	Villalba Alta [(8)]	-0.231	0.061	-0.3 [-0.73, 0.03] hm^3 year-^{1}
	Tramacastilla	0.072	0.551	
reservoir	Inflow	-0.296	0.016	-1.09 [-2.04, -0.21] hm^3 year-^{1}
	Outflow	-0.241	0.040	-0.92 [-1.57, -0.14] hm^3 year-^{1}
regulated streamflow	Teruel	-0.304	0.011	-1.55 [-2.90, -0.20] hm^3 year-^{1}
	Arquillo de San Blas	-0.286	0.015	-0.88 [-1.89, -0.21] hm^3 year-^{1}
	Zagra [(9)]	-0.183	0.153	

Source: own elaboration based on AEMET (2013) and MAGRAMA (2013).
[1] Following the methodology proposed by Yue et al. (2002) and Lorenzo-Lacruz et al. (2012) using the Kendall's rank correlation and correcting the effect of autocorrelation on the series.
[2] Based on Newson (2006).
[3] It does not include the following years: 2000, 2001 and 2008.
[4] It does not include the following years: 1996 and 1997.
[5] It does not include the following years: 2003, 2004 and 2005.
[6] It does not include the following years: 2000, 2001, 2008 and 2009.
[7] It does not include the following years: 2000.
[8] It does not include the following years: 1979, 1980 and1983.
[9] It does not include the following years: 1974-1979 and 1982-1984.

C. Ongoing land use changes

According to the 2004 land use map (MARM, 2009), the basin is currently mostly covered by forests (occupying 45% of UTB), followed by rainfed annual crops (23%), pasture-shrubland transition (19%) and shrubland (10%). Irrigated crops occupy 1% of the basin area. Urban and water infrastructures account for less than 1% of the UTB. Forests are mostly located in the west and southern parts, and shrubland and rainfed crops in the north-east and east. Forest and shrubland dominate on higher areas with 1250 m (±225 std.) and 23% (±16 std.) slope, although along the river valley and at the basin outlet they are also found at lower altitudes. Cropland and pastureland are mostly located on flatter areas with 960 m (±155 std.) and 7% (±7 std.) slope.

To determine the LULCC that have taken place within the UTB we combined the information of the land use map of the years 1977 (MARM, 1985) and 2004 (MARM, 2009) to create a landscape transition matrix and determine the main LULCC processes. Both land use maps were selected because they describe the oldest and the newest land use statuses in the UTB during the period of analysis (1973-2008). The land use matrix and the resulting LULCC processes have been included in Annex C Figure C1.

During the studied period, nearly 50% of the UTB has experienced either a land use or a land cover change. 23% of the observed land changes leads to a reduction of the vegetation cover and include processes of *shrubland clearing* (12%), *forest degradation* (7%) and *conversion to agricultural land* (4%). *Shrubland clearing* has mainly occurred in the north-east of the UTB, along a wide range of slopes and generally high (20% ±15 std.), probably encouraged by grazing (Figure 5.2). *Forest degradation* is concentrated in the north-western, central and southern parts of the basin, and mainly involved the degradation of coniferous and its conversion to shrubland in more steep areas (25% ± 17 std.). Areas experiencing a *conversion to agricultural land* include mostly shrubland and forests that have been converted into rainfed annual crops. This land use trend has taken place in minor spots throughout the basin, but particularly in the south-west. An increase in vegetation cover has also been observed in other parts of the basin, affecting 18% of the total area. 11% of the increase in vegetation cover refers to the *reforestation/afforestation* trend that occurred along the north-west, central and south in areas with slopes close to 24% (±16 std.). Nearly 80% of the observed reforestation refers to the natural succession from pasture and shrubland areas. *Agricultural abandonment*, principally caused by the natural vegetation succession from annual crops to shrubland areas, affected 6% of the UTB and has mostly occurred in flatter areas, with slopes close to 9% (±8 std.) slope. Other less representative land cover processes include *forest transitions*, which affects less than 6% of the UTB and include the natural succession from coniferous forest into mixed forest.

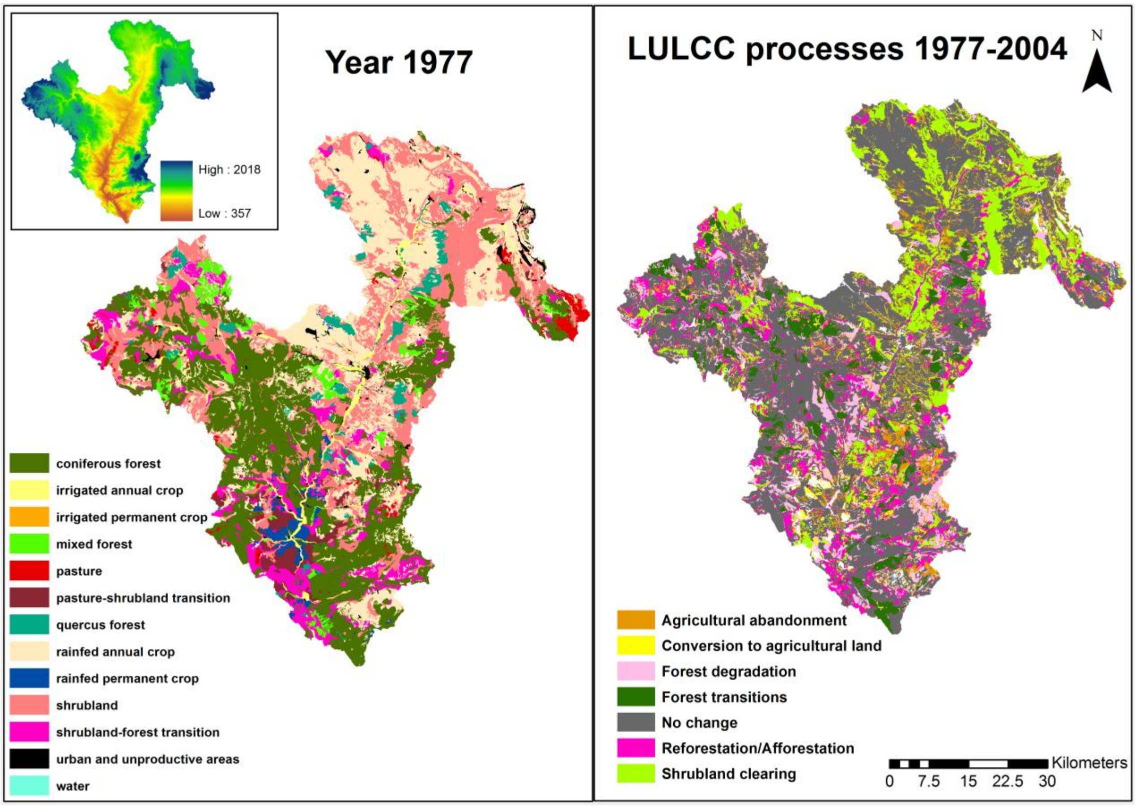

Figure 5.2. Existing land uses in 1977 (left) and land use land cover change (LULCC) processes that occurred between 1977 and 2004 (right) in the Upper Turia basin (UTB). Source: Own elaboration based on 1977 (MARM, 1985) and 2004 (MARM, 2009) land use maps. Only LULCC processes that comprise more than 1% of the total basin area are represented.

5.2.2. Drivers of observed changes in streamflow

Changes in the river flow of the UTB between 1973 and 2008 could be influenced by a combination of observed changes, including changing climate conditions, LULCC and river impoundment. Streamflow gauging stations showing a negative trend in Table 5.1 were selected to determine the drivers that are accountable for the existing trends. Firstly, regressions for the period 1973-2008 of annual streamflow (Q in hm^3) with annual precipitation (*P* in mm) and temperature (*T* in ºC) as explanatory variables are fitted, following the methodology proposed by Guardiola-Claramonte et al. (2011) and Lorenzo-Lacruz et al. (2012):

$$Q = aP + bT + c + \varepsilon \qquad [5.1]$$

Where *a* and *b* represent the coefficient estimators for *P* and *T* respectively, *c* is the intercept and *ε* is the error term. Secondly, an additional regression model is fitted to evaluate the effect of other drivers besides climate conditions to changes in streamflow. We include the variable time (*Y* in years) in equation [5.1], in order to include the effect of other drivers like LULCC in the prediction of changes in annual streamflow (Beguería et al., 2003; Morán-Tejeda et al., 2010):

$$Q = aP + bT + Y + c + \varepsilon \qquad [5.2]$$

The existence of a trend in the residuals is tested with Kendall's rank correlation coefficient (Kendall, 1938). Since the magnitude of the annual trend at the outflow and inflow of the reservoir shows that reservoir management does not represent a key driver for the existing negative trend downstream (See section 5.2.1), we assumed that *Y* mostly accounts for LULCC also under a regulated regime.

5.2.3. Quantifying the impact of LULCC: applying the ecohydrological model SWAT

A. Model description and design

Soil & Water Assessment Tool (SWAT) is a semi-distributed hydrological model performing on a daily time step (Arnold et al., 1998). A key strength of SWAT is its flexible framework that allows the simulation of both hydrology and plant growth, besides other processes such as sediment, nutrients and pesticide cycle (*ibid*). Moreover, the model is freely available and is readily applicable through the development of geographic information system. Worldwide applications of SWAT model are numerous and have been identified in several studies (White and Chaubey, 2005; Gassman et al., 2007). Land use change assessment studies using SWAT have been previously reported in Fohrer et al. (2001), Cao et al. (2008), Li et al. (2009) and

Ouyang et al. (2010), among others. However, they only were able to include a single land use layer and change the fractional areas of each hydrological response unit[6] (HRU), which could result in static model responses (Pai and Saraswat, 2011). In SWAT 2009, a land use change module was introduced (Arnold et al., 2010) to solve this issue and a computer-based geospatial tool was created to prepare the input files required to activate the module for each land use map layer (Pai and Saraswat, 2011).

The data required to run SWAT are detailed in Table 5.2. Surface runoff is calculated using a modified curve number method (Soil Conservation Service, 1972) and is estimated based on land use, hydrologic soil group, and previous soil moisture. For climate, SWAT uses the data from the station nearest to the centroid of each subbasin. Potential evapotranspiration (PET) was estimated by SWAT using Hargreaves' method that only relies on temperature (Hargreaves and Samani, 1985).

Table 5.2. Data used to run SWAT.

Basic data	**Description**	**Source**
Digital elevation model (DEM)	25m x 25m grid cell	IGN (2012)
Land use map for years 1977 and 2004	Scale 1:50,000	MARM (1985); MARM (2009)
Daily data of maximum temperature, minimum temperature and precipitation	17 monitoring stations for precipitation and 13 for temperature	AEMET (2013); JRBA (2013)
Soil	Representative soil unit for the whole study area	Estimated based on Trueba et al. (1999)
Surface of reservoir, volume parameters and daily release flow		MARM (2011b)
Additional data		
Crop area	Crops area at municipal scale distinguishing between rainfed and irrigates systems	MAGRAMA (2012)
Agricultural practices	For each crop in the study	See Annex D Table D1
Urban water consumption	It refers to year 2009	Júcar River Basin Authority

To run SWAT, the UTB was firstly divided into 28 subbasins taking into account the location of actual flow calibration stations and dominant land uses according to 2004 land use map. These subbasins are further divided into HRUs. The hydrography of the basin is delineated based on the digital elevation model (DEM). Each land use

[6] Areas with unique combination of land uses, soil types and slope. A basic computational unit assumed to be homogeneous in hydrologic response.

classification is linked to the "Land Cover/Plant Growth" SWAT 2009 database. The legend of the land use maps is further split for some crops and natural areas depending on the dominant species per subbasin (Table 5.3). The agricultural management practices are detailed in Annex D and Table D1.

Table 5.3. Land uses identified in the UTB and dominant vegetation species and crops composition.

Original land use classification	Split land use based on dominant species
Rainfed annual crops	barley, wheat
Irrigated annual crops	barley, wheat
Rainfed permanent crops	almonds, vineyards
Irrigated permanent crops	apple orchards
Pasture	
Pasture-shrubland transition	pasture, shrubland
Shrubland	
Shrubland-forest transition	shrubland, Juniperus forest, Quercus forest, Coniferous forest, Poplar forest
Coniferous forest	
Quercus forest	Quercus forest, Poplar forest
Mixed forest	Quercus-Coniferous forest, Coniferous forest, Juniperus forest
Water	
Urban and unproductive land	

Regarding soil parameters, SWAT requires detailed information on the physicochemical properties of each of the soil units of the different HRUs. Soil data of the UTB is very limited and the only information available (Trueba et al., 1999) includes a total of 14 soil profiles, with very similar physicochemical characteristics in terms of soil moisture content and soil conductivity. Therefore, we decided to apply a representative soil unit for the whole UTB (See Annex D Table D2). Urban water consumption is estimated for each subbasin according to the urban water demand for year 2009, based data provided by the Júcar River Basin Authority. We assume that 70% of water abstraction returns to the water system.

B. Model calibration, validation and evaluation

The parameters controlling plant growth include plant heat units (PHU), radiation use efficiency (RUE), leaf area index (LAI) and root depth. PHU refers to the heat requirements to meet a plant maturity. We calculate it based on the temperature within the study area, the base temperature of each plant species and the period that ranges from planting date until harvest, for annual crops, or from blooming date until seed, for natural vegetation (See Annex E Table E1). RUE, LAI and root depth were estimated based on SWAT default values and through literature review (See Annex E Table E2).

A multisite manual calibration for daily streamflow is performed in Villalba Alta, Teruel and Zagra (basin outlet) stations. Table 5.4 includes the stations used for the calibration and validation periods and the years considered. The parameters modified during calibration procedure and classified by the main hydrological component that they influence are included in Annex E Table E3.

Table 5.4. Calibration and validation periods for each monitoring station.

Baseline$_{2000-2008}$		
Stations	**Calibration period**	**Validation period**
Villalba Alta Teruel Zagra (basin outlet)	2000-2001, 2004-2005, 2007	2002-2003, 2006, 2008

Model efficiency for flow discharge is assessed with graphical techniques, and quantitative statistics efficiency criteria as described by Moriasi et al. (2007). Flow duration curves (FDC) are plotted comparing both measured and simulated daily records. The FDC plots the streamflow and the percentage of time that a specific flow is equal or likely to exceed it. In the FDC, the streamflow is transformed with logarithm as *log (streamflow+1)* in order to gain a more graphically comprehensive distribution of the flow. Efficiency statistics measures include the coefficient of determination (R^2), percent bias (PBIAS), the root mean square error (RMSE) and RMSE-observations standard deviation ratio (RSR). The RSR standardizes the RMSE dividing the RMSE by the observations standard deviation.

C. Land use scenarios

SWAT was used to simulate the hydrological changes of the basin under different land use scenarios. In the baseline scenario (*$Climate_{2000\text{-}2008}$ & $LULC_{2004}$*), we calibrated SWAT to simulate the hydrological response of the UTB for the time period 2000-2008, using the most updated land use information (year 2004) (MARM, 2009). To account for the impact of LULCC and isolate the effect of CC, an alternative scenario (*$Climate_{2000\text{-}2008}$ & $LULC_{1977}$*) was developed and run also with SWAT. This alternative scenario assumes the same climate conditions of 2000-2008 but with the land use situation corresponding to 1977 (MARM, 1985). Land use information was included in the simulation using the tool SWAT2009 LUC[7] (Pai and Saraswat, 2011) that generates the required files to run SWAT under both scenarios. Model outputs for the baseline scenario include the annual runoff coefficient (RC) (streamflow/precipitation). We also calculate the mean annual evapotranspiration rates (ET) of each land use (in mm).

Changes in streamflow between the two scenarios were assessed at daily, monthly and annual time steps. Mann-Whitney two-sample statistic test (Mann and Whitney, 1947) was applied to identify significant changes in daily streamflows. Likewise, the percentage of change of the daily flow (Q_{95}, probability of exceedance>95%), median (Q_{50}, probability of exceedance=50%) and high flow (Q_5, probability of exceedance<5%) were compared. To discern the impacts of the different types of LULCC, each one of the land use processes identified in Section 5.2.1 were summarized as spatial annual variations in ET.

5.3. Results

5.3.1. Drivers influencing observed streamflow trends

[7] The tool up today can only be run for non-split land uses. So we run the tool with a non-split HRU model. Then, the obtained HRU fraction was recalculated based on the percentage of split land use classification per subbasin for years 1977 and 2004.

Table 5.5 shows the results of the regression models fitted to assess the influence of CC and LULCC on the streamflow regime of the UTB. Under natural flow regime (Villalba Alta), *P* is the only significant variable that explains observed streamflow variability in Model 1. However, the negative trend of the residuals (τ) found in Model 1 also suggests that relevant variables are being omitted. With Model 2 under natural regime too, the significance and negative slope of *Y* highlights the relevance of the time variant changes because of a steady LULCC. Residuals in Model 2 do not exhibit any longer a trend and the adjusted R^2 increases from 0.32 to 0.38. Since 32 observations are available for Model 2 in Villalba Alta and only two explanatory variables are significant, over-parameterizing is not likely to occur.

P and *T* are significant climatic variables in the regression Model 1 for the reservoir inflow/outflow and monitoring stations (Arquillo de San Blas and Teruel) under regulated flow. In Model 1, residuals do not show a clear negative trend since p-values range from 0.14 to 0.33. However, once the variable *Y* is included (Model 2), *T* becomes no longer significant and the trend in the residuals change the sign to positive and becomes even less significant. In addition, the adjusted R^2 also increases with the inclusion of the variable *Y*. *T* is not significant in Model 2 since it is correlated with *Y* (ρ=0.56; p<0.001). Therefore, the significance of *Y* captures the effect of temperature but also of other factors changing over time like LULCC.

Table 5.5. Model 1 presents the regressions of streamflow (*Q* in hm^3) against precipitation (*P* in mm) and temperature (*T* in ºC). Model 2 also includes the explanatory variable year (*Y* in years). Adjusted coefficient of determination (Adj. R^2) and the significance of trends in residuals (p-value<0.05) with Kendall rank correlation coefficient (τ) are included as model efficiency measures.

Dependent variable: *Q* (streamflow)										
	Villalba Alta		**Reservoir inflow**		**Reservoir outflow**		**Arquillo San Blas**		**Teruel**	
Explanatory variables	**Model1**	**Model2**	**Model1**	**Model2**	**Model1**	**Model2**	**Model1**	**Model2**	**Model1**	**Model2**
P	0.08***	0.09***	0.14**	0.12*	0.13**	0.11**	0 .13**	0.12**	0.21***	0.18**
T			-12.91*		-12.11*		-12.78*		- 23.08**	
Y		-0.39*		-1.24**		- 1.24*		- 1.39*		- 2.03***
Intercept		757*	112*	2467**	107*	2452**	112*	2759**	221**	4053***
No.	32	32	35	35	36	36	36	36	36	36
Adj. R^2	0.32	0.38	0.21	0.27	0.19	0.27	0.20	0.30	0.38	0.39
τ	-0.32	-0.06	-0.12	0.09	-0.14	0.08	-0.18	0.10	-0.16	0.08
p-value τ	0.02	0.62	0.33	0.48	0.24	0.52	0.14	0.42	0.19	0.49

* p<0.05, ** p<0.01, *** p<0.001

5.3.2. Modeling the hydrological impacts of LULCC

A. Model outputs and efficiency performance

The calibrated model (baseline scenario) estimates an average annual ET in the UTB of 362 mm (±59 std.) between 2000 and 2008. The annual RC at the outlet of the basin has a value of 0.16 (± 0.03 std.), indicating that nearly 84% of the annual precipitation of the UTB is evapotranspired. Estimations of mean annual ET (mm) per land use are summarized in Figure 5.3. Annual ET rates per land use increase in the following order: annual vegetation<shrubland<woody vegetation<irrigated areas. Taking into account the annual ET and the representativeness of the different land uses in the baseline scenario, the greatest share of total annual water consumption in the UTB occurs in coniferous forest (28%), rainfed annual crops (20%) and pasture-shrubland transition (16%). The water appropriation by mixed forests (10%), shrubland (10%) and shrubland-forest transition (9%) is also significant.

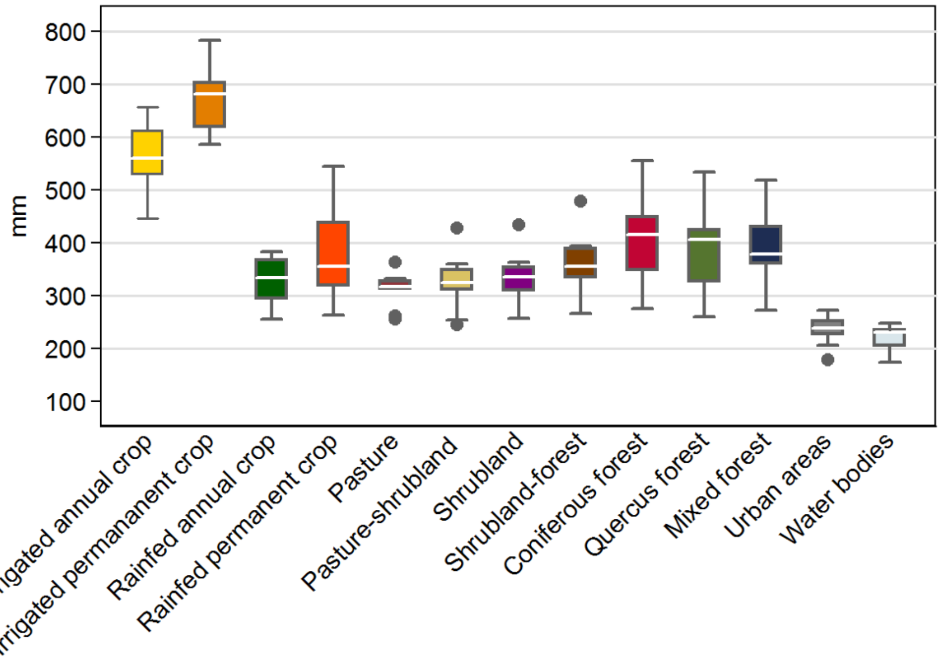

Figure 5.3. Box plot graph of annual ET in mm for each land use class for the baseline scenario (*Climate*$_{2000\text{-}2008}$ *& LULC*$_{2004}$). The white segment in the box represents the median, the lower (upper) box limits represent the first (third) quartile and whiskers the range of the data. Outliers are plotted as dots.

The efficiency criteria in the ecohydrological model for both calibration and validation periods are in general acceptable with R^2 above 0.59, with the exception of Villalba Alta station (Table 5.6). Nevertheless, in Villalba Alta the PBIAS shows absolute values no greater than 10% and the model tends to underestimate the streamflow. Teruel and Zagra (basin outlet) stations present the lowest values for RSR ranging from 1.13 to 1.41, indicating a better fit in these stations. Considering the whole period 2000-2008 both stations present a PBIAS below 10% and with negative sign, implying that the model fit has a tendency to slightly overestimate the flow. The FDC for the total period and each calibration station is gathered in Annex E Figure E1.

Table 5.6. Efficiency measures for the ecohydrological model *Climate*$_{2000\text{-}2008}$ *& LULC*$_{2004}$.

Gauge station	period	R^2	PBIAS	RMSE	RSR
Villalba Alta	Calibration	0.47	9.45	1.84	1.43
	Validation	0.29	5.11	2.86	1.66
	Total	0.36	7.17	2.36	1.58
Teruel	Calibration	0.72	-0.16	1.95	1.13
	Validation	0.59	-14.25	2.60	1.33
	Total	0.65	-6.09	2.28	1.24

	Calibration	0.64	-1.70	3.99	1.31
Zagra (basin outlet)	Validation	0.64	-4.32	4.96	1.41
	Total	0.64	-2.87	4.47	1.37

B. Hydrologic response under LULCC

According to the Mann-Whitney test, significant changes ($p<0.05$) are observed in the daily streamflow along the different monitoring stations when comparing the hydrological response of the baseline ($Climate_{2000-2008}$ & $LULC_{2004}$) and the alternative scenario ($Climate_{2000-2008}$ & $LULC_{1977}$) (See Annex F Table F1). In total terms, annual streamflow in the alternative scenario decreases relative to the baseline by 8.8 (±2.6 std.), 3.9 (±2.1 std.) and 2.8 hm^3 $year^{-1}$ (±1.5 std.) at the basin outlet, Teruel and reservoir inflow drainage areas, respectively. As Figure 5.4 shows, the largest reduction in monthly streamflow takes place between September and November in most stations, except for Villalba Alta. In Teruel a larger decrease in monthly streamflow occurs between January and February (7.5%) and between July and December (9.5%). At the basin outlet, greater streamflow reduction exists in September (9%), October (13%) and November (14%). Besides changes at annual and monthly steps, daily streamflow also presents lower values in the alternative scenario in comparison to the baseline. Reduction of the daily streamflow is particularly relevant in periods of low flow (Q_{95}) at the reservoir inflow, Teruel and the basin outlet (Zagra). The largest change for Q_{95} takes place at the basin outlet being reduced up to 42%, followed by the reservoir inflow area with 35.6%. In contrast, changes on high flow period (Q_5) show a minor magnitude of change (Table 5.7).

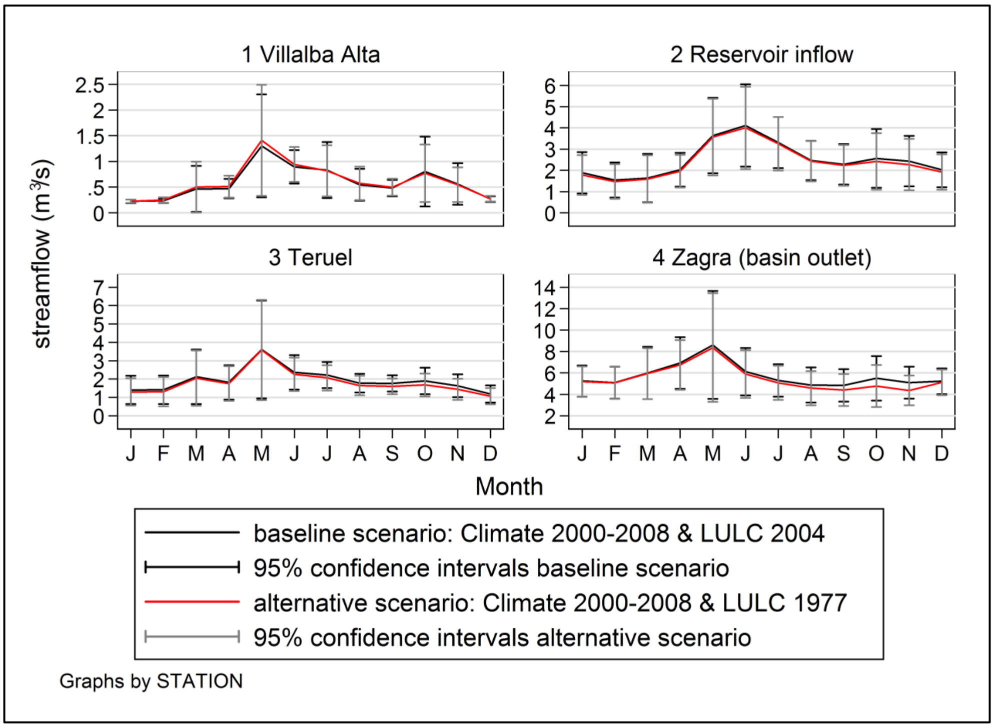

Figure 5.4. Monthly flow (m^3s^{-1}) for the baseline scenario (*Climate*$_{2000\text{-}2008}$ *& LULC*$_{2004}$) (black line) and the alternative scenario (*Climate*$_{2000\text{-}2008}$ *& LULC*$_{1977}$) (red line). 95% confidence intervals are also shown.

Villalba Alta represents an exception showing a significant increment ($p<0.05$) of streamflow in the alternative scenario compared to the baseline one (See Annex F Table F1). Most of the monthly streamflow increase takes place along the high flow period between March and May with a net increase of 8% (Figure 5.4), but increasing ET is also found during these months from 2.8 to 4.4%. Therefore, the greater values in Villalba Alta streamflow during peak flows are caused by the larger amount of precipitation during these months together with a larger annual curve number (from 65 in the baseline to 68 in the alternative scenario); whereas for the total basin the curve number remains constant (62). But, the lower model performance in Villalba Alta (Table 5.6) warrants some cautious interpreting of the conclusions.

Table 5.7. Average change in % for low (Q_{95}), median (Q_{50}) and high (Q_5) daily flow between the baseline scenario (*Climate*$_{2000\text{-}2008}$ *& LULC*$_{2004}$) and the alternative scenario (*Climate*$_{2000\text{-}2008}$ *& LULC*$_{1977}$).

Subbasin	ΔQ_{95} (%)	ΔQ_{50} (%)	ΔQ_5 (%)
Villalba Alta	0.0	2.0	3.6
Reservoir inflow	-35.6	-11.9	2.2

Teruel	-17.5	-13.0	1.5
Zagra (basin outlet)	-42.4	-10.9	1.4

To understand the changes in streamflow, we need to consider the underlying LULCC between 1977 and 2004 land use maps. Streamflow is larger under the baseline scenario with the 2004 land use map, because the overall ET decreased by 0.8% (±0.8 std.) at the reservoir inflow and by 1.6% (±2.3 std.) in Villalba Alta in comparison to the alternative scenario with the 2004 land use distribution. In twenty-seven out of twenty-eight subbasins, annual ET is lower under the baseline scenario, with the greatest differences occurring in the north-east of UTB, where ET values show a decrease between 20 and 26 mm year^{-1} with the actual land use distribution (Figure 5.5). In absolute terms, the land use trends have led to an overall ET drop of approximately 30 hm^3 (7.5 mm for the total basin) between 1977 and 2004.

This overall reduction, however, masks different hydrological responses and is the result of the prevailing trend of *shrubland clearing* and *forest degradation.* Each of these land use trends has reduced the ET by nearly 16 hm^3 for the total UTB's area. Shrubland clearing diminishes ET particularly above Teruel station's drainage area by 14 hm^3, whereas *forest degradation* decreases ET below this station by 10 hm^3. The largest ET rates for *shrubland clearing* and *forest degradation* take place at the reservoir inflow's drainage area with -64 mm and -88 mm, respectively. Other land use trends, although less representative in the context of UTB, have led to an overall increase of ET. For instance, *reforestation* processes that occurred within the UTB caused a total ET augmentation of 13 hm^3.

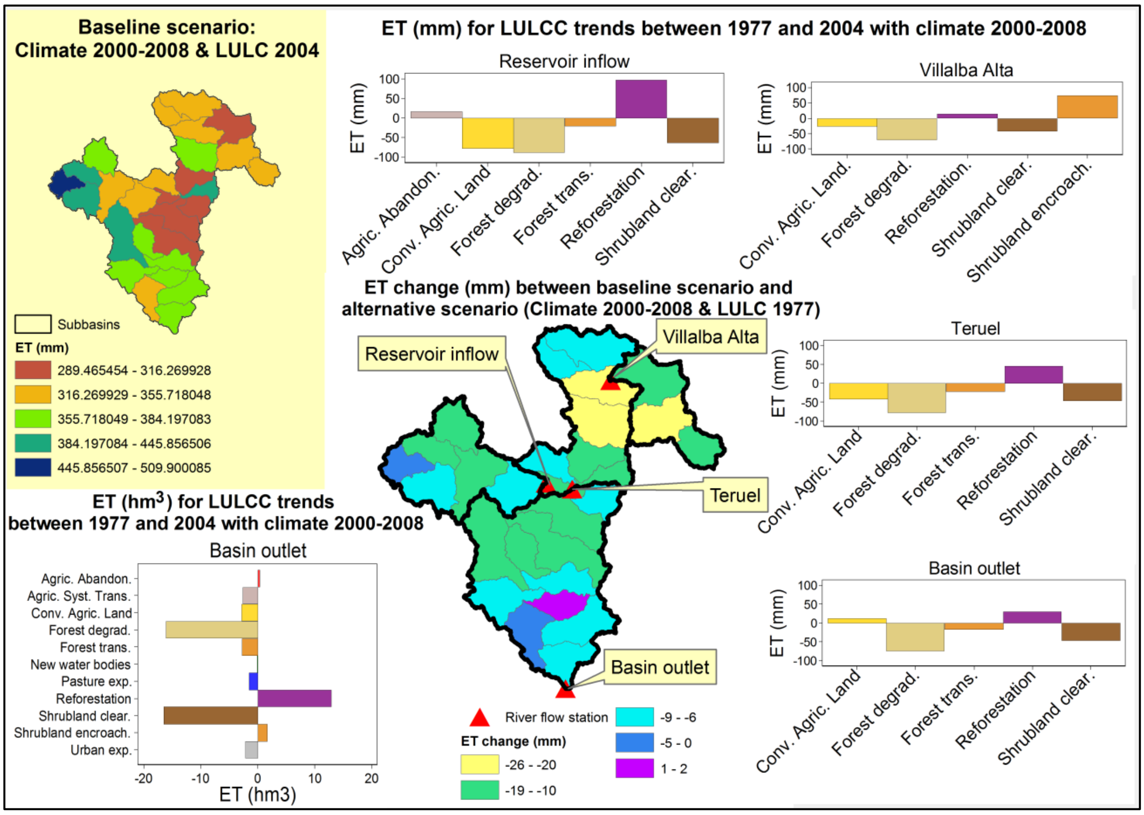

Figure 5.5. Map with yellow background: ET in mm for each subbasin considering the baseline scenario during 2000-2008 period. Map with white background: ET change in mm under the two land use scenarios (Climate$_{2000\text{-}2008}$ & LULC$_{1977}$ and Climate$_{2000\text{-}2008}$ & LULC$_{2004}$). The bar graphs on the top and right side show the ET (mm) for each land use trend and for the drainage area of each flow station. Only land use trends comprising more than 5% water appropriation for each gauge station are shown. Bar graph on the bottom left represent the ET (hm^3) associated to each land use trend in the entire basin.

5.4. Discussion

At large scales climate acts as a major control on the water balance partitioning (Thompson et al., 2011), but at lower scales the hydrological effects of LULCC can be as important as those caused by CC (Cui et al., 2012), and the impact of LULCC can be overshadowed by climate variability and vice versa. Our regression models of annual streamflow reveal the significance of considering a time variant variable like LULCC in addition to changing climate conditions, when understanding the changes on the hydrological regime of the UTB. Moreover, modeling the hydrological performance with SWAT allowed us to isolate and account for the hydrological impacts directly linked to LULCC, and discerning the ET rates by land use.

From a land and water planning perspective, gaining understanding on the water consumption (ET) of different land uses per unit of area (mm) is a valuable information in the context of semi-arid basins like the UTB, susceptible to the effects of CC. Mostly because it rises the issue of how to allocate water more efficiently within a water scarce basin and provides insights on land use options, which could be more suitable to adapt in a changing climate context. It thus provides an opportunity for strengthening the interdisciplinarity between hydrological and spatial planning. Nevertheless, and despite the relatively high water consumption of certain land uses, other factors i.e. ecosystem service benefits need to be also considered (Perring et al., 2012) when moving forward towards a more integrated water-land planning.

According to our modeling scenarios, the LULCC that occurred between 1977 and 2004 under actual climate conditions would have led to an overall decrease of the basin's ET i.e., an increase of annual streamflow. This result implies that ongoing LULCC have not exacerbated the streamflow reduction observed in UTB over the last four decades (1973-2008). The streamflow changes observed between both scenarios are significant despite being relatively small. Since only LULCC are modified in the alternative scenario, then any streamflow change is significant as changes in the streamflow are ascribed entirely to the existing LULCC in the UTB. The land use scenarios show that streamflow changes during peaks event are less relevant than alterations during low flow periods. Our results are in line with other studies (Brown et al., 2005; Farley et al., 2005; Zhang et al., 2012; Brown et al., 2013) that detected a larger influence on low flows compared to high flows, probably due to the larger impact of ET changes during water scarce periods. Factors such as the ET rate of the basin and the vegetation distribution have a high influence on the low flow regime (Smakhtin, 2001; Carrillo et al., 2011).

The overall positive impact of LULCC in the water balance of the UTB by increasing water availability and streamflow is greatly influenced by the predominant land use trends i.e. *shrubland clearing* and *forest degradation*, which have led to an overall decrease of ET. This circumstance requires a careful interpretation when understanding the likely impacts of LULCC in the water balance of semi-arid basins. Similarly to Nainggolan et al. (2012) in the Guadalentín basin (southeastern Spain), in the UTB various land use trends are taking place at the same time, due to changing socio-economic conditions (e.g. depopulation, urbanization) and biophysical characteristics (e.g. level of agricultural productivity). In fact, the diversity of LULCC occurring simultaneously in the UTB and the prevalence of some trends explains why the overall impacts of LULCC in UTB are contrary to those found in other studies conducted in the headwaters of the Pyrenees in the north of Spain like those of Delgado et al. (2010) and Gallart et al. (2011). Other studies have documented changes in vegetation cover as a result of changing climate conditions e.g. the forest die-off found after dry and heat events in Mediterranean Western Australia (Matusick et al., 2013) and Southwestern USA (Clifford et al., 2013), among others. In the context of the UTB, the influence of an increasing mean temperature in the land use trends observed (and particularly in those related to a decrease of vegetation cover) is likely but not certain, and further research is needed on this front. As many studies have demonstrated (García-Ruiz and Lana-Renault, 2011; Lasanta and Vicente-Serrano, 2012), most of the land use trends observed in Mediterranean landscapes like the UTB are very much related with socio-economic drivers i.e. abandonment of marginal agriculture and further intensification of the lowlands.

Similarly to previous studies (Cui et al., 2012; Zhang and Wei, 2012), an offsetting effect in streamflow generation is caused between LULCC and CC in the UTB. During the period 1973-2008, the streamflow at the Teruel monitoring station has been negatively influenced, on one hand, by a growing temperature; and on the other hand, it was positively compensated with lower ET demand thanks to LULCC trends. Between 1977 and 2004 land use maps, the effect of LULCC augment 3.9 hm^3 $year^{-1}$ (± 2.1 std.) in Teruel station with the existing 2000-2008 climate conditions. We can compare this value with the overall (i.e. with both CC and LULCC effects) negative trend of -1.55 hm^3 $year^{-1}$ at Teruel station during 1973-2008 (Table 5.1). The latter presents a negative slope because of a negative change caused by CC plus a positive change related to LULCC. We can affirm that LULCC represent a significant positive influence for streamflow variation; however, the increasing temperature rates are responsible for a negative greater effect on the streamflow response, leading as a result to a final negative trend. However, other studies have shown the greater role of LULCC on streamflow than CC. For instance, Guardiola-Claramonte et al. (2011) found that an increase of 0.5 ºC resulted in a depletion of only 10% of annual water yield, whereas the observed decrease including vegetation land cover change led to a depletion of 50% of annual water yield. Local and global future water resources must be estimated, not only in the light of climate-forcing projections but also in anticipation of hydrological consequences of LULCC (Gallart and Llorens, 2003; Piao et al., 2007; Delgado et al., 2010).

5.5. Conclusions

Likewise, to other basins of the world, the UTB presents significant decreasing trends in streamflow. SWAT was used to assess the implications of LULCC on the hydrological response, and has been proved an efficient tool for hydrological modeling and for the plant's ET simulation. In the absence of the changing climatic conditions, LULCC would have had a positive effect on the streamflow availability in the UTB during the period of study (1973-2008). We recommend River Basin Authorities considering land use management as a ‘’stream flow reduction activity’’, as South Africa (Jewitt, 2006) and Australia (Greenwood, 2013) already do, and to explicitly include the consequences of LULCC on water resources for future hydrological planning processes. We have attempted to isolate the effects of LULCC on the hydrological response. But there are still limitations in relation to the uncertainty and scarcity of available data (Romanowicz and Booij, 2011), related for example to the inadequate maintenance of climate and streamflow monitoring stations, as well as to the accuracy of land use maps. Secondly, the calculation of the reference evapotranspiration with Penman-Monteith method, instead of Hargreaves, would have allowed considering also the effects of changes in relative humidity, wind speed and solar radiation climatic variables. In this case, SWAT would have considered the impact of changes on vapor pressure deficit on the leaf conductance of the land cover for the estimation of the actual evapotranspiration. However, daily data were not available for all these climatic variables. Thirdly, we did not consider changes in biophysical conditions that do not necessarily change the type of land use (Lasanta and Vicente-Serrano, 2012) i.e. forest disturbance and forest management, mortality and succession of vegetation, which are considered as a key challenge for ecohydrology and forest science disciplines (Newman et al., 2006). Finally, a finer scale of analysis and modeling would have allowed distinguishing water strategies under water stress between species (Lázaro-Nogal et al., 2013). Nevertheless, our findings are valuable for gaining understanding on the nexus between water planning and land management in the UTB and by enlargement for the Júcar river demarcation. In addition, we provide a useful approach for the determination of the weight of LULCC on the hydrological response in other regions. For future studies, the magnitude of change for ET and streamflow will be analyzed considering both LULCC and under different climatic scenarios, in order to determine the effect of past and predicted climate conditions. It would be interesting to perform a comparative hydrology that learns from individual local studies and embeds basins along gradients (i.e. climate and land use distribution) and across spatial scales from small to large (Wagener et al., 2010).

6. Conclusions

6.1. General major findings

This thesis has been mainly motivated by the challenges that agricultural production systems, agricultural policy and land use and land cover changes (LULCC) pose on water resources management, with a special focus on a catchment analysis. LULCC on human induced (e.g. agricultural areas) and natural (e.g. forests) landscapes can lead to environmental effects such as modifications of erosion risks, hydrological response and water quality conditions. Agriculture shapes existing landscapes and is the main responsible of total water withdrawals, a motor of the economy in diverse regions and the provider of food for a growing demand. LULCC trends (like agricultural expansion) can change over time depending mainly on the productivity of the region, agricultural policy and external forces such as the globalization of food market and climate change.

In Spain during the last two decades, there have been significant changes of LULCC with a rapid expansion of new irrigated areas or the conversion of rainfed systems into irrigated one. One of the most relevant changes has occurred with the expansion and intensification of olive orchards in the Guadalquivir river basin, which have been promoted even under hilly slopes. This trend has been enhanced by the existing agricultural policy as well as national and international demand of olive oil. The basin embraces one the most intensified world region of olives production, but some environmental concerns have appeared regarding diffuse pollution, erosion risks as well as pressure on available water resources.

Green water is directly related to land occupation, but it is better framed in the context of the hydrological cycle. This water footprint component comprises the largest proportion of precipitation and changes in the green water flow (i.e. LULCC) can have an influence on the blue water component. As a result, green water management ties spatial and water planning. In the south-eastern Spain (Upper Turia basin), climate conditions are changing with an observed increase of temperature rates. Climate change might exert a major role on the reduction of water yield, but ongoing LULCC have also a significant effect on observed river flow trends in Spain. These overall key messages arise from three specific research focuses. The following sections outline the insights derived from each research component of the thesis.

The implications of changes of the Common Agricultural Policy and agricultural development on surface water quality

The Common Agricultural Policy (CAP) is a main driver for trends in agricultural systems and rural landscapes. Nevertheless, an analysis of the implications of past CAP reforms on water quality parameters has not been performed with a clear land and catchment perspective. In this thesis, Chapter 3 has examined the effects of the 2003 CAP reform on nitrates and suspended solids in the Guadalquivir basin's surface water, as well as their relation to variables that explain natural environment, agricultural sector, urban sector, CAP programs and economic factors. In the basin, about 20% of monitoring stations show significant trends, linear or quadratic, for both nitrates and suspended solids during the period 1999-2009. More significant trends of nitrates are augmenting than decreasing, and most significant quadratic terms of both indicators exhibit U-shaped patterns. Despite the acute suspended solids concentrations in the basin, only 5.6% of monitoring stations reduced the suspended solids concentrations during the study period. Surface water quality concerns have arisen, particularly for the high suspended solids concentrations found along the river tracts.

Among its main objectives, the 2003 CAP reform has attempted to protect the environment with the decoupling process and cross-compliance measures. Theoretically, with a common budget for agricultural subsidies, a large coupled component would incentive more production. The panel data analysis shows that the decoupling process and lower subsidies to irrigated agriculture decrease the nitrates concentration in Guadalquivir's subbasins that present a negative nitrates trend. For suspended solids, there is no clear evidence that the decoupling process has influenced negatively or positively. Nevertheless, greater values of subsidies still linked to production, particularly in irrigated regions, lead to increasing erosion rates.

The panel data regressions reveal that the most important drivers that are worsening both water quality indicators are agricultural biomass intensification and nitrates and suspended solids exports from upland. For nitrates the effect of point sources is also significant. Measures of irrigation modernization and establishment of vulnerable zones to nitrates reduce the concentration of nitrates in subbasins showing an increasing trend. However, the effect of nitrates exports from upland areas, intensification of biomass and crop prices present a greater weight leading to the observed increasing trend. The role of market prices stimulates agricultural production and intensification, in areas where annual crops dominate. This is because they give farmers more flexibility to change their crop patterns from year to year according to the market demand. Although agricultural production has grown in the basin and water efficiency in the agricultural sector has improved, the issue of high erosion rates has not yet been properly faced.

The water footprint and virtual water trade: a useful tool for water resource assessment

The water footprint (WF) is an indicator that provides a magnitude of human water appropriation for a certain product, process or specific geographical area. In a broad sense, this appropriation could be understood as the consumption of water resources that generates value for society. In Spain, the olive oil system has been expanded and intensified during the last decades, but the assessment of the water appropriation that this crop production demands have not yet been carried out. In Chapter 4, the WF of the Spanish olive oil production is calculated to emphasize the importance of this crop production in terms of water appropriation and nitrates pollution assimilation. Over the period 1997-2008 the Spanish olive oil production presents the following average percentage of the total WF: 72% green for rainfed systems (7,410 hm^3), 12% green for irrigated ones (1,250 hm^3), 6% blue (635 hm^3) and 10% grey (1075 hm^3). More than 99.5% of the WF for 1 L of bottled olive oil takes places during the production of the fruit in the tree, whereas only less than 0.5% of the WF is related to the plastic bottle, cap and label. The results confirm the importance of a detailed WF supply chain assessment of ingredients in the case of agriculture based products. The green component stands out as the largest portion for both rainfed and irrigated systems. This circumstance makes olive oil production largely dependent on the precipitation pattern. WF evaluations that omit the grey component would lead to incomplete conclusions, as they may contribute to increased efficiency in water consumption but fail to consider the environmental quality aspects.

Irrigated olive orchards generate more added value to olive production and require less amount of water to produce 1L of olive oil, although their production relies on scarce blue water resources. Between 1997 and 2008 olive orchard area has more than doubled in the major production region (Andalusia) leading to what could be called an 'olive oil bubble'. The largest producing places (Jaén and Córdoba provinces) show high water use efficiency per product (m^3 t^{-1}) and apparent water productivity (€ m^{-3}) as well as less nitrates pollution potential (grey water in (m^3 t^{-1}), but involve great pressure on the available water resources.

The olive production has increased its apparent water productivity during 1997-2008 incentivized by growing trade prices and CAP coupled payments, but also did the amount of virtual water exports. The country is exporting increasingly more olive oil and a certain degree of specialization has occurred, improving the productivity of the water consumed. Although 77% of virtual water exports for olive oil are related to the green WF, the LULCC generated by the expansion of irrigated olive orchards took place mainly at the expense of increasing groundwater abstractions in the Upper Guadalquivir basin. This has caused concerns about the sustainability of olive irrigation in those areas. Olive production, and subsequently water, is linked to trends of international markets, as well as to aids from agricultural policy. Our results suggest that virtual groundwater exports related to olive oil exports may add further pressure to the already stressed Guadalquivir river basin. It seems that the role of the Guadalquivir River Basin Authority of monitoring and controlling water uses has failed to limit the growth of water diversions caused by the 'olive oil bubble'.

The LULCC as a driver of streamflow response

In the Upper Turia basin, headwaters of the Júcar river demarcation, a negative trend in the streamflow between 1973 and 2008 period has been observed. Changing climate conditions might exert a major role on water yield trends, but it remains unclear the role that ongoing LULCC might have on observed river flow trends. Chapter 5 analyzes the effects of LULCC on the observed streamflow reduction in the UTB. Regarding climate variables, increasing temperature rates is the responsible of the existing hydrological response, since precipitation records do not show a clear significant trend. Additional drivers, like LULCC, also play a critical factor underpinning streamflow behavior, but they can be overshadowed by climate variability and vice versa. The overall impact of LULCC in the water balance is highly influenced by the predominance of the LULCC trends, which take place at the same time and are highly controlled by socio-economic drivers and biophysical characteristics. In the Upper Turia basin, the LULCC cause positives changes on the streamflow, since the land use processes shrubland clearing and forest degradation lower evapotranspiration demand, outweighing the influence of other land use trends (i.e. reforestation). A larger effect occurs in low flow periods, probably related to the greater impact that evapotranspiration changes pose during water scarce periods. However, during 1973-2008 an offsetting impact direction in streamflow response is caused between climate change and LULCC in the Upper Turia basin. The streamflow has been negatively influenced by increasing temperature rates, whereas LULCC trends have caused a positive effect on the streamflow. In the end, the growing temperature has a greater influence on the streamflow response, leading as result to a final negative streamflow trend.

6.2. Original contributions and practical application of the research

Addressing environmental policies at basin scale

This thesis highlights the role of agriculture policy on water quality status in order to improve the environmental conditions in basins where agricultural activity dominates and is one of the major pressures. The research focuses on one area of water resources assessment that still faces challenges for its mitigation, namely, the diffuse pollution. Since the beginning of the 1990s, observations of surface water quality parameters are available across Spain (although data availability depends on the indicator and basin). This thesis attempts to valorize existing available data regarding nitrates and suspended solids in the Guadalquivir river basin. From a policy perspective, it considers the main drivers that might be exerting an influence on both water quality indicators are considered, including the 2003 CAP reform. To conduct my research, I had to gather large and complex data sets and process explanatory variables (describing the natural environment, agricultural sector, CAP programs, urban sector and economic factors).

Scale limitations arise in studies that combine variables usually available in administrative units (e.g. crops area, crops yield and prices), but the assessment requires to be related to a naturally delineated area i.e. the basin. The management of spatial data with Geographic Information System permitted estimating variables such as agricultural biomass, percentage of coupled subsidy, agricultural subsidies or crop price index at the level of subbasin within the Guadalquivir. This research enhances the characterization of a basin in order to achieve a more detailed analysis, advantageous in water and land management and as well as in decision-making process.

A further step in water resource management: the water footprint and virtual water trade

The thesis has put into context the water resources assessment for the production of olive oil in Spain. Actual water resource management can be complemented with water footprint studies, particularly for crops that require the largest amount of water (both green and blue) or can generate concerns regarding diffuse pollution (grey water). Moreover, the virtual water exports help to identify pressures on existing local water resources, showing the need to balance the market forces with the available local resources.

The blue water assessment have been improved, since it is based on the water allowances and modified according to existing level of drought in the Guadalquivir river basin, which can restrict irrigation amount. Even if there are great uncertainties on their compliance, potential errors remain low in comparison with the assumption of complete fulfillment of crop water requirements. Moreover, improvements of the method of Hoekstra et al. (2011) were made for the grey WF since it is calculated based on nitrogen surplus (nitrogen inputs - nitrogen outputs) instead of the chemical application rate per hectare times the leaching fraction. The application of nitrogen surplus leads to more realistic WF grey, since the assessment relies on the nitrogen amount that is lost by surface runoff or leaches to groundwater bodies.

The water label has become a new way of certificating the efficient use of water resources for all water users (European Commision, 2011). However, our results lead us to conclude that the implementation of water label would not be accurate for food products if only a single value is given regarding to water consumed/polluted. This is because of its high variability due mainly to existing climate conditions, soil type and agricultural system production. The water label would require also assessing the possible impacts that the production of a certain product can have on the local site e.g. high risk of erosion in slope olive orchards, diffuse pollution concerns because of high nitrogen inputs or existing pressure on the available water resources of the region. Moreover, although farm practices are performed on the field, users downstream can suffer their environmental impacts. Therefore, a recommendation from this analysis is that WF assessments should be communicated to the general public with a view to making them more aware of the environmental consequences and water needs of consumption goods.

Interdisciplinarity between hydrological and spatial planning

In the face of climate projections and the likely reduction of water availability to satisfy societal water demand, a better understanding of the effects of LULCC (i.e. green water flow) on streamflow response is needed for water and land managers. This thesis provides a useful methodology and approach for the determination of the weight of LULCC on the hydrological response. This approach can be extended to other river basins where significant changes on streamflow are occurring. Modeling the hydrological performance with SWAT has the advantage of accounting for the hydrological impacts of LULCC and discerning ET rates by land use. The use of ecohydrological models that can make this distinction allows for estimating the differences of evapotranspiration demand per unit area, which bring about the question of how to allocate more efficiently water within a basin. This provides insights on land use options, which could be more suitable to adapt in a changing climate context.

The thesis provides insight for strengthening the interdisciplinarity between hydrological and spatial planning for the management of the Júcar river demarcation. Spanish River Basin Authorities lack enough information about the implications that LULCC can have on existing and projected water resource planning, since studies regarding changes on streamflow due to LULCC at the adequate scale are scarce. A strong emphasis should be given to the dissemination of the research results to the Júcar river demarcation and other Spanish River Basin Authorities. A strong recommendation to River Basin Authorities is to consider land use management as an activity that is likely to change the availability of water in a water course, and to include the consequences of LULCC on water resources for future hydrological planning processes. A more efficient water resources planning in Spain requires taking into account in future hydrological plans projections not only drivers of changes such as climate change conditions, population growth, agriculture development or energy demand, but also the implications of expected LULCC.

6.3. Limitations and recommendations for further research

In the Guadalquivir basin for panel data analysis, some of the most important measures regarding environmental characteristics, agricultural structure, income support and price index are included. However, our causation hypotheses on nitrates and suspended solids response can be disturbed by factors not fully included in the model with the effects of price volatility, livestock load, fertilizer use and urban wastewater spills. Environmental implications could also be more adequately assessed if there were available information regarding the level of enforcement of the good agricultural and environmental condition (GAEC) after 2006/2007 or detailed information about the actual effects of rural development programs. Lastly, more detailed evaluation of the role of irrigation strategies (including fertirrigation), which certainly play a role in reducing agricultural non-point pollution, was not included. After the CAP post 2014, conditionality measures, greening components and rural development programs might be great opportunities to improve the water quality conditions encountered within the Guadalquivir river basin. For future research, studies need to focus on crop pattern changes as well as management practices after the CAP 2014 with cross-compliance and greening measures i.e. crops diversification, vegetation filter, minimum tillage and reforestation in order to mainly mitigate erosion rates in the basin. Additional studies need also to consider procedures of water abstraction control under GAEC requirements such as water abstraction permits, water meters and reports of water use on farmers' response, in order to assess jointly the environmental sustainability of possible future scenarios of agricultural policies and the impacts on the ecological status of local water resources.

The WF is an indicator of water appropriation but further local crop production studies, completed with a wider range of social, economic and environmental indicators are required for an appropriate sustainability assessment and informed decision-making. The WF of 1 L olive oil depends on the existing climatic conditions, soil characteristics, production systems and irrigation strategies. Irrigation farmers' decisions depend on several factors e.g. energy cost, particularly for woody crops that have traditionally been grown under rainfed conditions (i.e. olives and vineyards). Although in this thesis the main olive oil producing provinces in Spain do not seem to represent significant sources of nitrate pollution, in practice nitrogen inputs of irrigated olives are nearly three times higher than rainfed ones. Consequently, a nitrogen balance that differentiates between these two olive production systems would also yield more accurate evaluations of the grey WF. Moreover, further research of grey WF also needs to focus both on spatial and temporal variation across the basins and within the agricultural season. All these climatic, agronomical and management aspects heavily influence the calculated value of the WF, and should be carefully taken into account for an accurate evaluation of the water use performance of crops at local scale.

The example of olive orchards expansion in Spain can be perceived as a warning for those emerging countries, which are mainly net blue water exporters of agricultural products and are gradually integrating into international trade. For future studies, it would be desirable to adopt a global and long-term approach to virtual water exports and imports, in order to understand changes in sustainability problems associated with impacts on local water resources.

There are limitations in the effects of LULCC on the hydrological response in relation to the uncertainty and scarcity of available data regarding streamflow and land use distribution. For instance, the calculation of the reference evapotranspiration with Penman-Monteith method, instead of Hargreaves, would have not rely exclusively on temperature and would have allowed considering the impact of changes on vapor pressure deficit on leaf conductance for the estimation of the actual evapotranspiration. However, complete series of daily data were not available for all these climatic variables within the study site. Moreover, changes in biophysical conditions that do not necessarily change the type of land use i.e. forest disturbance and forest management, mortality and succession of vegetation are not included in SWAT ecohydrological model. A finer scale of analysis would have also permitted distinguishing vegetation response under water stress between species. Nevertheless, the findings regarding the implications of LULCC on streamflow response in the Upper Turia basin are valuable for the management of the Júcar river demarcation. For future studies, the magnitude of change for evapotranspiration and streamflow will be further analyzed considering LULCC under different climatic scenarios, in order to determine the effect of LULCC under significant climate changing conditions. It would also be challenging to perform a comparative hydrology that embeds basins along gradients (i.e. climate and land use distribution) and across spatial scales from small to large.

While acknowledging such limitations, this thesis has contributed to a better understanding on the effects of agricultural development and policies and LULCC on water quality conditions, hydrological response and human water appropriation at basin scale and eventually, promote and efficient, equitable and sustainable allocation of water resources in Mediterranean environments.

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Annexes

Annex A. Calculation of crop area and aboveground biomass

The crop area and aboveground biomass in dry weight are calculated for each subbasin *i* and agricultural season *t*, distinguishing between rainfed and irrigated areas, separately. Four rainfed crop groups (annual crops, olives orchards, vineyards and other woody crops) are visualized using the land use maps for years 1999, 2003 and 2007 (RGA, 2010; MARM, 2009). A crop group ratio is calculated as the proportion of the area of each crop group per municipality and subbasin. Crops areas are available at municipal scale (MAGRAMA, 2012a). To obtain crop areas for rainfed conditions, rainfed crop ratios are firstly assigned to its corresponding year, crop group, municipality and subbasin. Then the rainfed crop ratios are multiplied by the municipal crop area and sum up by subbasin identification. The irrigated land was allocated using the Irrigated Inventories of Andalusia for years 1996 and 2002 (RGA 1999; 2003) and data of the Guadalquivir Hydrological Plan that refer to 2010, provided by the Guadalquivir River Basin Authority. The irrigated olive orchards extension within Andalusia was estimated using the Irrigated Inventories of Andalusia for years 1996, 2002 and 2008 (Corominas, 2010, pers. comm., 29 June) because it afforded more accurate information than municipal data. Crop groups were not differentiated for irrigated areas, assuming them to be distributed homogeneously within the irrigated land. As a result, for irrigated land the procedure was the same, but instead we had a single ratio for all crops. Since the calculation could only be applied for years with exiting land use maps, the ratio for both rainfed and irrigated crops was interpolated for years in between.

The agricultural production (P^a, in t) comprises the economic or agricultural parts (grain, fiber, fruit or tuber) during crop production and is expressed as:

$$P_{it}^{a} = \sum_{j=1}^{129} \left(1 - M_j^{prod}\right) \times Y_{itj} \times A_{itj}$$

[A.1]

Where for each *j* crop *(j=1,..., 129)* M_j^{prod} refers to the crop moisture given as a unity fraction, and Y_j and A_j represent the crop yield (t ha^{-1}) and crop area (ha), respectively. Crop yields were obtained from the Agricultural Statistics Yearbooks (MAGRAMA, 2012b), which have a provincial reference.

The residual production (P^r, in t) refers to the crop residues that remain in the field after the crop is harvested and its calculation differs between annual and woody crops. The harvest index (*HI*) was applied for the calculation of residual production of annual crops. The *HI* is defined as the ratio of the agricultural part in dry weight to the total aboveground dry matter production at the time of harvest.

$$P_{it}^{r} = \sum_{k=1}^{95} \frac{1}{HI_k} \left(1 - HI_k\right) \times \left(1 - M_k^{prod}\right) \times Y_{itk} \times A_{itk}$$

[A.2]

Where for each *k* annual crop (*k=1,…, 95*) M_k^{prod} refers to the crop moisture given as a unity fraction.

For woody plants the total aboveground biomass calculation is carried out using the Residue to Product Ratio (*RPR*). The *RPR* indicates the amount of residue that becomes available for each ton of product. The *RPR* relates the residual biomass produced by pruning to the agricultural production.

$$P_{it}^{r} = \sum_{m=1}^{30} RPR_m \times \left(1 - M_m^{res}\right) \times Y_{itm} \times A_{itm}$$

[A.3]

Where M_j^{res} denotes the moisture of residual for each *m* woody crop (*m=1,…,30*) given as a unity fraction.

The total aboveground biomass was then determined summing the agricultural production and residual production for each subbasin *i* and agricultural season *t*:

$$B_{it} = \frac{P_{it}^{a} + P_{it}^{r}}{A_{sub}}$$

[A.4]

Table A1. Crops under study: gathering parameters required to calculate the aboveground biomass for annuals and permanent crops.

Crop	Crop group	Harvest Index	Product moisture (%)	Residue to Product Ratio	Residue moisture (%)	References
Wheat (*Triticum aestivum* L.), durum wheat (*Triticum durum* Desf.)	Winter cereals	0.30	14			Urbano, 2002; Moragues et al. 2006
Barley (*Hordeum* L.)		0.45	14			Urbano, 2002; Francia et al., 2006
Oat (*Avena sativa* L.)		0.35	14			
Rye (*Secale cereale* L.)		0.52	14			Urbano, 2002; Hansen et al., 2004
Triticale (×*Tritico secale* Wittm.)		0.35	14			Urbano, 2002; Giunta and Motzo, 2004
Other cereals		0.41	14			Average of wheat and rye
Rice (*Oryza sativa*L.)	Spring cereals	0.50	18			Casanova et al., 2000
Maize (Zea mays L.)		0.57	18			Pordesimo et al., 2004
Sorghum (*Sorghum bicolor* L.)		0.53	14			Hammer and Broad, 2003
Dry common bean (*Phaseolus vulgaris* L.)	Leguminous grain	0.45	14			Ferreira and Vieira, 2000
Lentil (*Lends culinaris* Medikus)		0.48	12			Ayaz et al. 2004; Hernández & Fuertes 2011
Chickpea (*Cicer arietinum* L.)		0.45	12			
Common vetch (*Vicia sativa* L.)		0.36	12			Fırıncıoğlu et al., 2010; Hernández and Fuertes, 2011
Bitter vetch (*Vicia ervilia* (L.)Willd.)		0.32	12			Hernández and Fuertes, 2011; Larbi et al., 2011
Dry broad bean (*Vicia faba* L.)	Proteaginous	0.39	13			Annicchiarico, 2008
Dry pie (*Pisum sativum* L.)		0.39	13			Annicchiarico 2008; Hernández & Fuertes 2011
Yellow bush lupine (*Lupinus arboreus* Sims.)		0.54	13			
Potato (*Colocasia esculenta* L. Schott)	Tubers	0.70	75			Victorio et al., 1986; Hernández and Fuertes, 2011
Sweet potato (*Ipomoea batatas* L.Lam)		0.70	75			Assumed identical to potato
Cotton (*Gossypium hirsutum* L.)	Cotton	0.29	12			Hernández and Fuertes, 2011; Ünlü et al., 2011
Linseed (*Linum usitatissimum* L. Griesb.)	Oleaginous	0.44	12			Hocking and Pinkerton, 1991
Flax (*Linum usitatissimum* L. Griesb.)	Flax and hemp	0.60	27			Dimmock et al., 2005
Hemp (*Cannabis sativa* Lam.)		0.84	27			Meijer et al., 1995
Sunflower (*Helianthus annuus* L.)	Industrial	0.40	10			Bange et al., 1997
Soya (*Glycine max* (L.) Merrill.)		0.38	15			Guerrero, 1999; Jin et al., 2010
Rape (*Brassica napus* L.)		0.25	9			Faraji et al., 2009
Sugar cane (*Saccharum officcinarum* L.)		0.85	70			Larcher, 2003; Hernández and Fuertes, 2011
Sugar beet (Beta vulgaris L.)		0.64	89			Sahin et al., 2004; Hernández and Fuertes, 2011
Cartamo (*Carthamus tinctorius* L.)		0.22	8			Guerrero, 1999; Koutroubas et al., 2009
Bell pepper (*Capsicum annuum* L.) for paprika		0.70	90			Assumed identical to pepper
Tobacco (*Nicotiana tabacum* L.)		1	85			Guerrero, 1999; Hernández and Fuertes, 2011
Anise (*Pimpinella anisum* L.), saffron (*Crocus sativus* L.)		1	14			Hernández and Fuertes, 2011
Other industrial crops		0.70	20			

Lavender (*Lavandula angustifolia* Miller.)		1	14			
French clover (*Trifolium incarnatum* L.), rose (*Rosa* spp), other flowers, ornamental plants	Flowers and ornamental plants	0.70				
Winter cereals, maize (Zea mays L.), sorghum (*Sorghum bicolor* L.), English ryegrass (*Lolium perenne* L.), other gramineous crops, alfalfa (*Medicago sativa* L.), clover (*Cuscuta epithymum* (L.) L.), common sainfoin (*Onobrychis viciifolia* Scop .), Italian sainfoin (*Hedysarum coronarium* L.), common vetch (*Vicia angustifolia* L.), celeriac (*Apium rapaceum* L., (Miller) Gaudin), grasslands, cabbage (*Brassica oleracea* L. (Alef.) capitata)	Fodder	1	65			Horrocks and Vallentine, 1999
Spargel (*Asparagus officinalis* L.)		0.14	92.8			Hernández and Fuertes, 2011
Spinach (*Spinacia oleracea* L.)		0.67	90.7			Biemond et al., 1996
Celery (*Apium graveolens* L.), chard (*Cnicus benedictus* L.)		0.67	90.7			Assumed identical to spinach
Caardon (*Cynara cardunculus* L.)		0.43	56.8			Piscioneri et al., 2000
Cucumber (Cucumis sativus L.)		0.84	96			Luomala et al., 2008
Tomato (*Lycopersicon esculentum* L.)		0.78	79.6			Ho, 1996
Strawberry (*Trifolium fragiferum* L.)		0.52	84			Reekie et al., 2007
Artichoke (*Cynara scolymus*)		0.27	85.3			Rincón et al., 2007
Cauliflower (*Brassica oleracea* var. botrytis L.(L.)		0.45	90			
Garlic (*Allium sativum* L.)		0.80	40			
Onion (*Allium cepa* L.)		0.75	89.6			
Spring onion (*Allium fistulosum* L.)		0.75	89.6			
Wild leek (*Allium ampeloprasum* L.)		0.68	88.8			
Wild carrot (*Daucus carota* L.)		0.83	88			
Radish (*Raphanus sativus* var. Radicula L. (Pers))	Vegetables	0.85	95.8			
Turnip (*Apium graveolens* var. rapaceum L., (Miller) Gaudin)		0.95	83			
Green bean (*Phaseolus vulgaris* L.)		0.45	90.1			Nendel et al., 2009
*Cabbage (*Brassica oleracea *L. (Alef.))*		0.75	92.8			
Lettuce (*Lactuca sativa* var. capitata L)., endive (*Cichorium endivia* L.)		0.80	96			
Watermelon (*Citrullus lanatus* (Thunb.) Matsumura & Nakai)		0.80	91.7			
Sweet melon (*Cucumis melo* L.)		0.80	93			
Pumpkin (*Cucurbita maxima* Duchesne), courgette (*Cucurbita moschata* DuchezPoir), aubergine (*Solanum melongena* L.), green pepper (*Capsicum annuum* L.)		0.70	90			
Pie (*Pisum sativum* L.)		0.39	79.9			Annicchiarico, 2008
Broad bean (*Vicia faba* L.)		0.39	90			
Mushroom (*Agaricus bisporus* (Lange))		0.59	91			Hernández and Fuertes, 2011
Orange tree (*Citrus sinensis* (L.) Osbeck), tangerine tree (*Citrus reticulata* Blanco), lemon tree (*Citrus limon* (L.) Burm. f.), grapefruit (*Citrus paradisi* (Macf.)), lime (*Citrus aurantifolia* (Christm.) Swingle) and other citrus	Citrus trees		89.1	0.10	40	Di Blasi et al., 1997; BEDCA, 2007

Pear (*Pyrus communis* L.),	Fruit trees	90	0.20	0	BEDCA, 2007; Esteban et al., 2008
Cherry (*Prunus laurocerasus* L.),		81	0.20	0	Esteban et al., 2008; Hernández and Fuertes, 2011
Peach (*Prunus persica* L. Batsch),		90	0.20	0	
Apple *(Malus domestica Borkh.),*		84	0.20	0	
Plum (*Vitis labrusca* L.),		82.4	0.20	0	BEDCA, 2007; Esteban et al., 2008)
Almond (*Prunus amygdalus* Batsch),		5.9	1.90	40	Di Blasi et al., 1997; BEDCA, 2007
Walnut (*Juglans regia* L.),		5.9	1.90	40	
Other fruits with shell		5.9	1.90	40	Assumed identical to almond
Other fruit trees			0.20	0	Assumed identical to apple and
Vineyard (*Vitis vinifera* L.)	Vineyard	80.7	0.20	0	BEDCA, 2007; Esteban et al., 2008
Olive tree (*Olea europaea* L.)	Olive tree	48	0.50	0	Esteban et al., 2008; Hernández and Fuertes, 2011

Annex B. Agricultural subsidies calculation

The agricultural subsidies (€ ha^{-1}) are calculated for rainfed and irrigated land in terms of total subsidies and divided by the total agricultural area under rainfed or irrigated conditions, respectively. The calculation differs between the Direct Payment Scheme (DPS) (1999-2006) and the Single Payment Scheme (SPS) (2007-2009). In order to determine the agricultural area that received subsidies, we assumed that all crop areas, whose crops were entitled to, obtained subsidies. According to the Royal Decree 1893/1999 (BOE, 1999) herbaceous irrigated areas were restricted to those presented in the agricultural season 1998/1999. As a result, production subsidies of new irrigated herbaceous areas after this season were considered as rainfed systems. After November 2001 olive trees planted after the 1997/1998 season were not entitled to receive the subsidies (OJEC, 1998). After the agricultural season 2001/2002, we assumed that the olive orchard area could not exceed that area of 1998.

With the DPS all crops were subsidized in terms of production (€ t^{-1}) or cultivated area (€ ha^{-1}) (Table B1), which, when multiplied by the crop area (ha) or crop production (t), result in the total subsidies received (€). The herbaceous subsidies based on production were estimated according to reference regional crop yields (BOE, 1997; 1999; 2002). Production crop subsidies for cotton and olives trees were estimated based on province crop yields (MAGRAMA, 2012b). Sugar beet farmers received aid based on a minimum price paid by the sugar industry, which in turn received a subsidy quota.

The SPS includes both single and coupled payments. The total subsidies for the single payment (€) was calculated by multiplying the decoupled proportion by the reference amount, which corresponds with the mean subsidies received during a period of reference (Table B2). Although the single payment is no longer coupled to production, it is still linked to agricultural land based on historical agricultural production. The coupled payment (€) was estimated by multiplying the subsidies in terms of production

(€ t^{-1}) or cultivated area (€ ha^{-1}) after the agricultural reform by the entitled crop area (in ha) or crop production (in t), and by the ratio of coupling.

Table B1. Agricultural subsidies per unit of production (€ t^{-1}) or cultivated area (€ ha^{-1}) before the 2006/2007 agricultural reform.

<table>
<tr><th></th><th></th><th colspan="9">agricultural season</th></tr>
<tr><th>Crops classification</th><th>units</th><th>1997/98</th><th>1998/99</th><th>1999/00</th><th>2000/01</th><th>2001/02</th><th>2002/03</th><th>2003/04</th><th>2004/05</th><th>2005/06</th></tr>
<tr><td rowspan="2">cereals without maize[1]</td><td>€ t^{-1}</td><td colspan="3">54</td><td>59</td><td colspan="5">63</td></tr>
<tr><td>€ ha^{-1} [2]</td><td colspan="2">359</td><td colspan="5">345</td><td>313</td><td>291</td></tr>
<tr><td>maize[1]</td><td>€ t^{-1}</td><td colspan="3">54</td><td>58.7</td><td colspan="5">63</td></tr>
<tr><td>rice[1]</td><td>€ ha^{-1}</td><td>111</td><td>223</td><td colspan="5">334</td><td colspan="2">1124</td></tr>
<tr><td rowspan="2">protein crops[1]</td><td>€ t^{-1}</td><td colspan="3">79</td><td colspan="4">73</td><td colspan="2">63</td></tr>
<tr><td>€ ha^{-1}</td><td colspan="7">-</td><td colspan="2">56</td></tr>
<tr><td>Leguminous grain[1]</td><td>€ ha^{-1}</td><td colspan="9">181</td></tr>
<tr><td>oil crops[1]</td><td>€ t^{-1}</td><td colspan="3">94</td><td>82</td><td>72</td><td colspan="4">63</td></tr>
<tr><td>flax (oil seed)[1]</td><td>€ t^{-1}</td><td colspan="3">105</td><td>88.3</td><td>76</td><td colspan="4">63</td></tr>
<tr><td rowspan="2">flax (textile)[1]</td><td>€ t^{-1}</td><td colspan="4">-</td><td>76</td><td colspan="4">63</td></tr>
<tr><td>€ ha^{-1}</td><td colspan="3">615</td><td>691</td><td colspan="5">-</td></tr>
<tr><td rowspan="2">hemp[1]</td><td>€ t^{-1}</td><td colspan="4">-</td><td>76</td><td colspan="4">63</td></tr>
<tr><td>€ ha^{-1}</td><td>717</td><td colspan="2">663</td><td>646</td><td colspan="5">-</td></tr>
<tr><td>cotton</td><td>€ t^{-1}</td><td>622</td><td>770</td><td>865</td><td>742</td><td>878</td><td>850</td><td>812</td><td>884</td><td>817</td></tr>
<tr><td>tobacco</td><td>€ ha^{-1}</td><td>2565</td><td>2564</td><td>2767</td><td>2768</td><td>2785</td><td>2782</td><td>2761</td><td>2766</td><td>2772</td></tr>
<tr><td>nut and carob tree[2]</td><td>€ ha^{-1}</td><td colspan="7">-</td><td colspan="2">242 [5]</td></tr>
<tr><td>olive fruit</td><td>€ t^{-1}</td><td>-</td><td colspan="7">150</td><td rowspan="2">Single payment[6] Coupled payment[7]: 57, 75 t ha^{-1}</td></tr>
<tr><td>olive oil[3]</td><td>€ t^{-1}</td><td>1422</td><td colspan="7">1323</td></tr>
<tr><td>sugar beet[4]</td><td>€ t^{-1}</td><td colspan="4">46</td><td colspan="5">48</td></tr>
</table>

[1] They comprise the herbaceous crops
[2] The supplement subsidy to durum wheat
[3] It was assumed a 20% production of olive oil per kg of oil fruit.
[4] Sugar beet farmers received aid based on a minimum price paid by the sugar industry, which in turn received a subsidy quota. We estimated sugar production assuming 10% impurities, 16.5° average polarization and 13% sugar yield (BOE, 2006a).
[5] Hazelnut tree receives additionally 105 € ha^{-1} after the agricultural season 2006/2007
[6] For the calculation of single payment the final subsidies received per ton of olive oil, including the penalizations, were applied for the reference period. 2000: 1408 € t^{-1}, 2001: € t^{-1}, 2002: 689 € t^{-1}, 2003: 1117 € t^{-1}
[7] 75 € ha^{-1}: municipalities whose olive area is greater than 80% of the overall arable land. 50 € ha^{-1}: olive groves located in traditional olive growing regions

Sources: (MAPA, 1999; 2000; 2001; 2002; 2003; 2004; 2005; 2006; 2007).

Olive orchards were required to meet some characteristics in order to receive the coupled subsidy (BOE, 2005). Firstly, they should not produce more than 57 t ha^{-1} of olive oil. Land holdings with a production larger than 57 t ha^{-1} were identified based on the holdings size at municipal level for year 2009 (INE, 2011), olive crop yield and production of olive oil per kg of olive. Olive orchards with more than 57 t ha^{-1} of olive oil production were not included. Additionally, in the Andalusian Community two categories were established (BOJA, 2006): 1) land holdings in municipalities whose olive area is greater than 80% of the overall arable land, 2) olive groves located in traditional olive growing regions. The former were estimated based on the calculated crops' areas at subbasin level, the latter being available in BOJA (2006).

The average percentage of coupled subsidy (*% Coupling*) is calculated with the product of the percentage of coupled subsidy (*Coup*, in %) of each *k* crop entitled to received subsidies *(k=1,..., 32)* and the crop area (*A* in ha), divided by the agricultural area entitled to receive subsidies (A_{CAP} in ha) and multiplied by a ratio that considers the proportion of crop area entitled to receive subsidies per subbasin area (*ratio CAP*):

$$\% \, Coupling_{it} = \frac{\sum_{K=1}^{32} Coup_{itk} \times A_{itk}}{A_{CAPit}} \times ratio \; CAP_{it}$$

[B.1]

Table B2. Decoupled payments (%), reference period and subsidies per unit of production (€ t^{-1}) or cultivated area (€ ha^{-1}) used for the calculation for the single payment system after the agricultural reform 2006/2007.

<table>
<tr><th rowspan="2">Crops classification</th><th rowspan="2">% decoupled payments</th><th rowspan="2">Reference period</th><th rowspan="2">units</th><th colspan="3">Subsidies (€ t^{-1} or € ha^{-1})</th></tr>
<tr><th>2006/07</th><th>2007/08</th><th>2008/09</th></tr>
<tr><td rowspan="2">cereals without maize[(1)]</td><td rowspan="2">75</td><td rowspan="9">2000/2001, 2001/2002, 2002/2003</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>€ ha^{-1}</td><td colspan="3">285[(5)]</td></tr>
<tr><td>maize[(1)]</td><td>75</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>rice[(1)]</td><td>58[(2)]</td><td>€ ha^{-1}</td><td colspan="3">Single payment: 648
Specific payment: 476</td></tr>
<tr><td rowspan="2">protein crops[(1)]</td><td rowspan="2">75</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>€ ha^{-1}</td><td colspan="3">56</td></tr>
<tr><td rowspan="2">grain leguminous</td><td rowspan="2">100</td><td rowspan="2">€ ha^{-1}</td><td colspan="3">181[(6)]</td></tr>
<tr><td colspan="3">175, 176, 150 [(7)]</td></tr>
<tr><td>oil crops[(1)]</td><td>75</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>flax (oil seed)[(1)]</td><td>75</td><td rowspan="3">2001/2002, 2002/2003</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>flax (textile)[(1)]</td><td>75</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>hemp[(1)]</td><td>75</td><td>€ t^{-1}</td><td colspan="3">63</td></tr>
<tr><td>cotton</td><td>51[(2)]</td><td rowspan="2">2000/2001, 2001/2002, 2002/2003</td><td>€ t^{-1}</td><td colspan="3">Single payment: 1358 € t^{-1}
Specific payment: 1039 € t^{-1}</td></tr>
<tr><td>tobacco</td><td>40</td><td>€ ha^{-1}</td><td colspan="3">2773 € t^{-1}</td></tr>
</table>

nut and carob tree		- [4]	€ ha^{-1}	242
olive fruit	93.6	1999/2000, 2000/2001, 2001/2002, 2002/2003	€ t^{-1}	Single payment[8] Coupled payment[9]: 57, 75 t ha^{-1}
olive oil	93.6		€ t^{-1}	
sugar beet	60[3]	2004/2005, 2005/2006	€ t^{-1}	- [10]

[1] They comprise the herbaceous crop group.

[2] Estimated. The proportion of coupled payments for rice growers was estimated by dividing the specific payment by the direct one before the reform took place. For cotton, the mean subsidy per ha for a maximum production of 249,000 t for Spain was estimated as indicated in MAPA (2007). The coupled proportion was obtained by dividing the mean subsidy per ha by its 1039 € ha^{-1} single payment.

[3] 64.2% after 2009 year.

[4] Single payment does not exist. A subsidy in terms of cultivated area is given for this sector. Hazelnut tree receives 347 € ha^{-1}

[5] The supplement subsidy to durum wheat.

[6] For human consumption. It includes chickpeas and lentils.

[7] For animal consumption. It includes common vetch and bitter vetch. For the calculation of the single payment: 175 € ha^{-1} in 2001, 176 € ha^{-1} in 2002 and 150 € ha^{-1} in 2003.

[8] For the calculation of the single payment the final subsidies received per ton of olive oil were used, including the penalizations, during the reference period. 2000: 1408 € t^{-1}, 2001: € t^{-1}, 2002: 689 € t^{-1}, 2003: 1117 € t^{-1}

[9] 75 € ha^{-1}: municipalities whose olive area is greater than 80% of the overall arable land. 50 € ha^{-1}: olive groves located in traditional olive growing regions.

[10] Sugar beet farmers received aid based on a minimum price paid by the sugar industry, which in turn received a subsidy quota.

Sources:(MAPA, 1999; 2000; 2001; 2002; 2003; 2004; 2005; 2006; 2007; OJEU, 2003; BOJA, 2006).

Annex C. Land use change processes

Land use dynamic between two land use maps was analyzed with a transition matrix using ArcGIS 9.3.1 (ESRI, 2009). This matrix refers to changes occurred from 1977 land use map (MARM, 1985) to the 2004 map (MARM, 2009). The Figure 1 gathers the legend classification and summarizes how the main land use and land cover change (LULCC) processes are quantified. The study proposes broad LULCC processes that comprise for example conversions to agricultural areas, pasture land extensification, agricultural abandonment or establishment of new forested areas, among others.

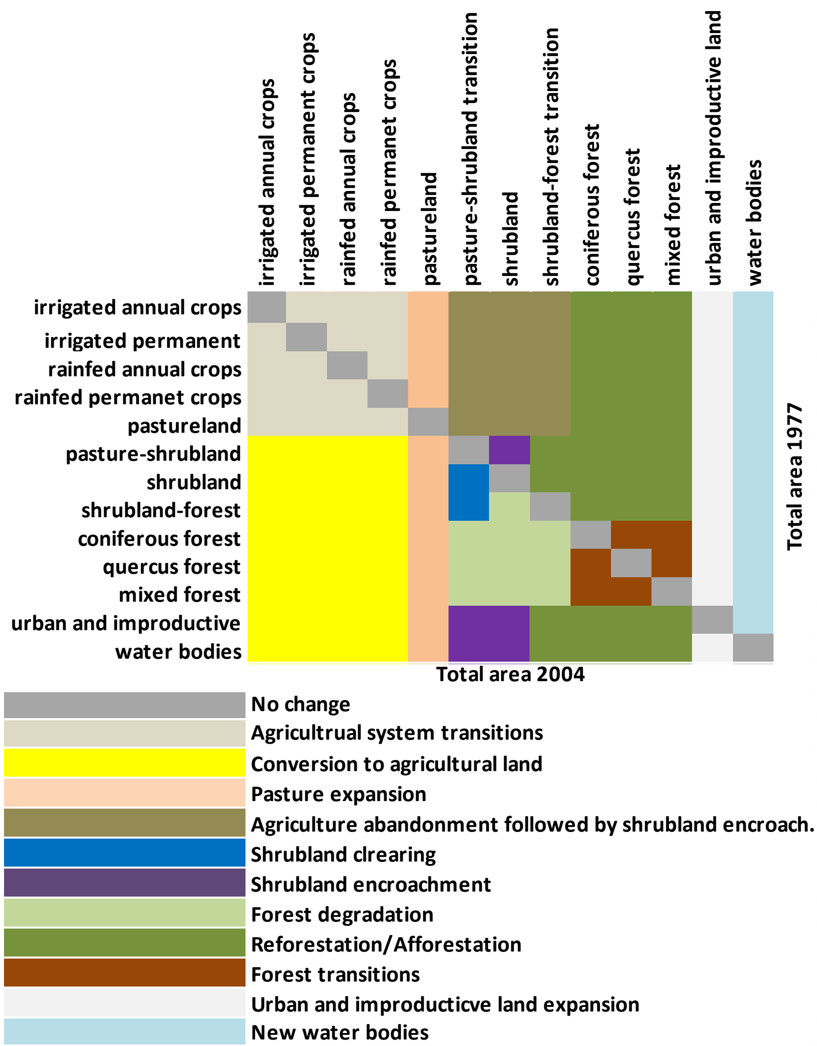

Figure C1. Main land use changes processes between 1977 and 2004 according to the summarized legend of the land use maps.

Annex D. SWAT model design: agricultural management practices

Table D1. Agricultural management practices.

Crop	Month	Day	Operation	Description
Barley[1]	1	15	Autofertilization	Nitrogen stress threshold: 1 Max. N per application: 50 kg ha^{-1} Max. N per year: 200 kg ha^{-1}
	4	15	Autoirrigation	Water stress threshold: 0.9 Irrigation efficiency: 0.8 Amount of irrigation per application: 5 mm
	6	15	Harvest & kill	
	11	1	Tillage	Chisel Plow 21 feet
	11	10	Planting	
Wheat[1]	1	15	Autofertilization	Nitrogen stress threshold: 1 Max. N per application: 50 kg ha^{-1} Max. N per year: 200 kg ha^{-1}
	4	15	Autoirrigation	Water stress threshold: 0.9 Irrigation efficiency: 0.8 Amount of irrigation per application: 5 mm
	7	15	Harvest & kill	
	11	1	Tillage	Chisel Plow 21 feet
	11	15	Planting	
Corn	4	15	Tillage	Field Cultivator Ge15ft
	4	24	Autoirrigation	Water stress threshold: 1 Irrigation efficiency: 0.9 Amount of irrigation per application: 10 mm
	4	25	Planting	
	4	27	Autofertilization	Nitrogen stress threshold: 1 Max. N per application: 50 kg ha^{-1} Max. N per year: 300 kg ha^{-1}
	10	27	Tillage	Disk Plow Ge23ft
	11	3	Tillage	Semi-chisel
	11	28	Tillage	Semi-chisel
	12	5	Harvest & kill	
Almonds	3	1	Planting (blooming)	
	3	1	Tillage	Moldboard Plow 2-way 4-6b
	3	2	Autofertilization	Nitrogen stress threshold: 1 Max. N per application: 50 kg ha^{-1} Max. N per year: 100 kg ha^{-1}
	4	1	Tillage	Tandem Disk Plow 14-18ft
	5	1	Tillage	Row Conditioner 1 Row
	6	1	Tillage	Row Conditioner 1 Row
	7	1	Tillage	Row Conditioner 1 Row
	10	1	Harvest	
	11	1	Tillage	Deep Ripper - Subsoiler
Apple	3	1	Tillage	Moldboard Plow 2-way 4-6b
	3	1	Planting (blooming)	
	3	2	Autofertilization	Nitrogen stress threshold: 1

				Max. N per application: 50 kg ha^{-1} Max. N per year: 300 kg ha^{-1}
	4	1	Tillage	Tandem Disk Plw 14-18ft
	4	15	Autoirrigation	Water stress threshold: 0.9 Irrigation efficiency: 0.8 Amount of irrigation per application: 5 mm
	5	1	Tillage	Row Conditioner 1 Row
	6	1	Tillage	Row Conditioner 1 Row
	7	1	Tillage	Row Conditioner 1 Row
	11	1	Tillage	Row Conditioner 1 Row
Vineyards	1	2	Autofertilization	Nitrogen stress threshold: 1 Max. N per application: 50 kg ha^{-1} Max. N per year: 100 kg ha^{-1}
	3	1	Tillage	Moldboard Plow 2-way 4-6b
	3	1	Planting (blooming)	
	3	20	Tillage	Tandem Disk Plow 14-18ft
	4	20	Tillage	Row Conditioner 1 Row
	5	10	Tillage	Row Conditioner 1 Row
	6	10	Tillage	Row Conditioner 1 Row
	7	10	Tillage	Row Conditioner 1 Row
	10	1	Harvest	
	10	15	Tillage	Deep Ripper - Subsoiler
Pasture	3	1	Planting (blooming)	
	3	15	Grazing	Grazing days: 250 days 7.4 kg ha^{-1} day^{-1} of sheep manure Dry weight of biomass consumed: 2.7 kg ha^{-1} day^{-1}

[1] For barley and wheat the irrigation requirements were not met as it is applicable with these crop species in Spain (Skhiri and Dechmi, 2012). Under rainfed conditions autoirrigation operation is not carried out.

Source: Hidalgo and Cabezudo (1994); MAPA (2001); AEMET (2012); FUNDEM (2006); Fertiberia (2007); Fernández et al. (2008); Pastor (2008); IFAPA (2012); Gobierno de Aragón (2013).

Table D2. Physicochemical characteristics for the soil unit in the Upper Turia basin.

	Soil layer 1	**Soil layer 2**	**Soil layer 3**
Depth (mm)	120	350	700
Bulk density (g cm^{-3})	1.5	1.2	1.2
Available water capacity (mm H_2O mm^{-1} soil)	0.12	0.07	0.09
Saturated hydraulic conductivity (mm h^{-1})	28	7	6
Organic carbon (% soil weight)	5.3	3.5	3.5
Clay (% soil weight)	18.6	24.1	27.6
Silt (% soil weight)	25.5	20.4	15.7
Sand (% soil weight)	55.5	55.5	56.7
Rock fragment (% total weight)	15	35	18

Estimated based on Trueba et al. (1999).

Annex E. SWAT model calibration

Table E1. Data required for the calculation of the plant heat units (PHU).

Crop	Base temperature[1] (ºC)	Planting /Blooming date	Harvest date	PHU
Barley	0	10 November	15 June	1730
Wheat	0	15 November	15 July	2320
Corn	8	25 April	5 December	1170
Almond tree	10	1 February	1 August	1380
Apple tree	7	1 April	1 October	1910
Vineyards	10	1 June	15 September	1170
Pasture	12	1 May	1 August	1080
Shrubland	12	1 May	1 August	1180
Pine forest	0	1 March	1 September	2880
Quercus forest	10	1 April	1 October	1380
Mixed forest	10	1 April	1 October	1380
Juniperus forest	10	1 April	1 October	1190
Poplar	10	1 April	1November	1470

[1] Default values in SWAT database.

Source: Hidalgo and Cabezudo (1994); MAPA (2001); FUNDEM (2006); Fernández et al. (2008); AEMET (2012); Gobierno de Aragón (2013).

Table E2. Parameters set up for plant characteristics.

Land use	Root depth (default SWAT & set up) (m)	LAI[1] (default SWAT)	LAI (set up)	RUE[2] (default SWAT) g/MJ	RUE (set up) g/MJ	references
Corn	2	6	3.8	3.9	2.8	Viña (2004); Aguilar et al. (2007)
Rainfed barley	1.3	4	2.3	3	1.8	Acreche et al. (2009)
Irrigated barley	1.3	4	3.7	3	2.5	Fergusson (1994); Berjón and Cachorro (2004); Mollah and Paul (2008); Acreche et al. (2009)
Rainfed wheat	1	4	2	3	1.6	Acreche et al. (2009); Nielsen et al. (2012)
Irrigated wheat	1	4	3	3	1.9	Acreche et al. (2009); Nielsen et al. (2012)
Almond	1.3	1.2	2	1.6	1.6	Andreu et al. (1997)
Vineyard	1.3	2	2	3	1.6	Baeza et al. (1999)
Pasture	0.8	4	1.8	3.5	3	Breuer et al. (2003)
Shrubland	1	2	2	3.4	3.5	Breuer et al. (2003)
Quercus	1.6	5	4.8	1.5	1.7	Sala and Tenhunen (1996); Bellot and Escarre (1998); Bellot et al. (1999)
Pinus	1.8	5	4.6	1.5	1.7	López-Serrano et al. (2000); Breuer et al. 2003
Juniperus	1.4	5	3	1.5	1.5	Gholz (1980)
Poplar	2	5	5.3	3	3	Gielen et al. (2001)

[1]LAI: leaf area index

[2]RUE: radiation use efficiency

Table E3. Parameters modified in the calibration process.

Parameter	Units	Hydrologic partitioning
Surface runoff lag coefficient		Surface flow
Curve number		
Maximum canopy storage	mm	
Available water capacity for each soil layer	mm H_2O/mm soil	
Soil evaporation compensation factor		
Effective hydraulic conductivity in main channel alluvium	mm/h	Surface/base flow
Saturated hydraulic conductivity for each soil layer	mm/h	
Threshold depth of water in the shallow aquifer for return flow to occur	mm	Base flow
Groundwater revap coefficient		
Threshold depth of water in the shallow aquifer for revap or percolation	mm	
Base flow recession constant[1]	days	
Groundwater delay time	days	
Lateral flow travel time	days	Lateral flow
Deep aquifer percolation fraction		Seepage lost to deep aquifer

[1]calculated with the Base flow Filter Program (Arnold et al., 1995; Arnold and Allen, 1999) available at http://swat.tamu.edu/software/

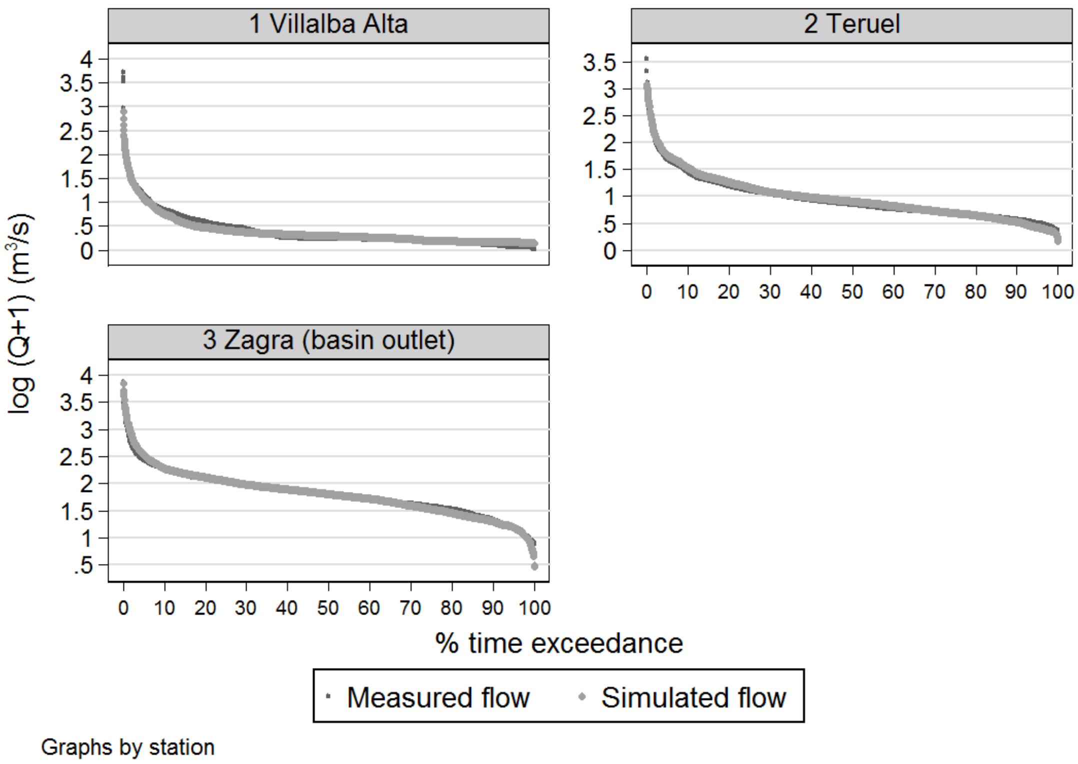

Figure E1. Flow duration curve for the total period 2000-2008 in each calibration station.

Annex F. Significant changes between baseline and alternative scenarios

Table F1. Man-Whitney test result for the identification of significant different distribution ($p<0.05$) for daily streamflow (Q) between the baseline scenario ($Climate_{2000\text{-}2008}$ & $LULC_{2004}$) and the alternative scenario ($Climate_{2000\text{-}2008}$ & $LULC_{1977}$). Only months with significant changes ($p<0.05$) are shown.

Gauge Station	Time period of analysis	z	p-value	Probability {Q alternative scenario < Q baseline scenario}
Villalba Alta	Period 2000-2008	-2.185	0.029	0.49
Reservoir inflow	Period 2000-2008	2.572	0.010	0.52
Teruel	Period 2000-2008	6.583	0.000	0.55
	January	2.081	0.038	0.55
	February	2.960	0.003	0.58
	July	2.121	0.034	0.55
	August	2.842	0.005	0.57
	September	2.718	0.007	0.56
	October	3.505	0.001	0.59
	November	3.848	0.000	0.60
	December	3.200	0.001	0.58
Zagra (basin outlet)	Period 2000-2008	4.222	0.000	0.53
	September	2.647	0.008	0.56
	October	2.948	0.003	0.57
	November	3.400	0.001	0.59

Annexes references

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