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Flows, footprints and valus: visions and decisions on groundwater in Spain
Flujos, huellas y valores: visiones y decisiones sobre el agua subterránea en España

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Tesis Doctoral

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Abstract

In many places, groundwater is a key resource. This is due to two essential aspects: its temporal and spatial availability. As these characteristics are also crucial for river base flows and wetlands, both direct benefits from groundwater and indirect values, linked to ecosystems services, are jeopardized by intensive use. In the context where the Water Framework Directive (WFD) requires a full integration of the environment into decision-making, this thesis aims at learning from the situation in Spain to explore options for better groundwater governance, both at the scale of the country and by detailing three case studies. Meanwhile, three more specific objectives are included: the formulation of criteria for groundwater allocation, the evaluation of the WFD implementation for groundwater, and the assessment of the relevance of the water footprint approach.

The starting point and guideline of the thesis is to focus on the role of groundwater to sustain surface water flows and ecosystems. The concept of ‘capture’ is introduced to help identify where, when, how much and how long flows are altered in the watershed. The acceptability of these changes for society is key in decision-making. This standpoint contrasts with the description of aquifers as groundwater stocks replenished by recharge, a common vision that fostered principles that need to be reassessed. In particular, ‘groundwater sustainability’ is analysed and the definition of groundwater as a Common-Pool Resource, the main economic modelling approach, or terms such as ‘groundwater mining’ or ‘non-renewable resources’ are questioned. The analysis of the criteria in the WFD and its implementation in Spain also shows that a traditional approach is reproduced for groundwater availability and groundwater body status assessments.

The WF is introduced as a flexible and multi-scalar indicator for computing direct and indirect freshwater appropriation. After identifying the origin and developments of the concept, its links to impact-oriented indicators, related to the Life Cycle Analysis, are discussed. The concept of an ‘Integral Water Footprint’ is then proposed to account for the whole water consumption in supply chains.

As regards the case studies, in the Western Mancha Aquifer, the WF and the dynamic of illegal groundwater use are characterized, emphasizing the regulatory framework and the implications of a recent public plan to reallocate groundwater. In the Guadalquivir River basin, in addition to the green and blue WF accounting for the whole basin, the current and future captures are distinguished. The La Loma de Úbeda Aquifer illustrates the issue of the integration of

groundwater resources in the watershed in the context of the development of irrigated olive groves. Finally, in Campo de Dalías, while a first aim is to account for the WF of vegetables grown in greenhouses, the general situation is also analysed, with a characterization of seawater intrusion and future supply options, particularly seawater desalination and its financing.

In the light of the different impacts from groundwater pumping for other users and the environment in the case studies, a main conclusion is that governance proposals for groundwater depend on the context, yet with groundwater integrated within the whole river basin, as a general rule. The relevance of policy instruments, like quotas, water markets, and water pricing, is also discussed, particularly faced with the situation where illegal use or over-allocation of rights is the rule. The role of drivers and subsidies is also identified. More generally, the transition to a better state for groundwater resources depends on the introduction of actors and values that are usually disregarded. This thesis illustrates the need for knowledge once the way a problem looked at has changed; first, regarding groundwater availability and local management and regulation; second, from the perspective of supply chains, with consumers and other entities as drivers for change.

Resumen

En muchas partes del mundo, el agua subterránea es un recurso fundamental gracias a su disponibilidad temporal y espacial, aspecto crucial también para el sostenimiento de ríos y humedales. Sin embargo, muchos beneficios directos e indirectos del agua subterránea, como los ligados a los servicios de los ecosistemas dependientes, se ven amenazados por su uso intensivo. En el contexto de la Directiva Marco del Agua (DMA), que requiere la integración del medioambiente en la toma de decisiones, esta tesis pretende identificar opciones para una mejor gobernanza del agua subterránea, en base al ejemplo de España, a escala del país y de tres casos locales. Se consideran también tres objetivos más específicos: la formulación de criterios para la repartición del recurso entre los usuarios, la evaluación de la aplicación de la DMA en relación con el estado cuantitativo de las masas de agua y la evaluación de la relevancia del enfoque de la huella hídrica.

El punto de partida de la tesis es la contribución fundamental del agua subterránea a los flujos superficiales y a los ecosistemas dependientes en la cuenca hidrográfica. El concepto de ‘captura’ es introducido para ayudar a identificar dónde, cuándo y cuánto estos flujos se ven alterados como consecuencia de los bombeos. La toma de decisiones se ha de basar en gran parte en la aceptación de estos impactos por la sociedad. Esta visión contrasta con la descripción común de los acuíferos como una reserva de agua, cuyo nivel de recurso se define por la recarga, y se discute una serie de principios asociados, como el agua subterránea como ‘recurso común’ (Common-Pool Resource), la modelación económica clásica, la definición de términos como ‘mina de agua’ o ‘recurso renovable’ o, más generalmente, el concepto de ‘sostenibilidad’ aplicado al agua subterránea. El análisis de los criterios de la DMA, y su aplicación en España, muestra también que se basa en un enfoque tradicional para la evaluación de la disponibilidad de recursos y del estado de las masas de agua.

Del punto de vista del uso, la huella hídrica es introducida como un indicador flexible y multi-escala de la apropiación directa e indirecta de agua. Además del origen y del desarrollo del concepto, se describe su relación con indicadores de impactos, ligados al Análisis del Ciclo de Vida. El concepto de una ‘Huella Hídrica Integral’ es propuesto para tener en cuenta la totalidad del consumo de agua en la cadena de valor.

En relación con los casos de estudio, en el acuífero de la Mancha Occidental se caracteriza la huella hídrica y la dinámica del uso ilegal, poniendo el énfasis sobre el marco regulatorio y las implicaciones de un reciente plan público para la reasignación del agua subterránea. En la

cuenca del Guadalquivir, además de la contabilidad de las huellas hídricas verde y azul, se identifican las 'capturas' actual y futura del uso del agua subterránea. El acuífero de La Loma de Úbeda ilustra la cuestión de la integración de los recursos de agua subterránea en la cuenca hidrográfica, en el contexto del desarrollo del olivar de regadío. Finalmente, en el Campo de Dalías, primero, se contabiliza la huella hídrica ligada a las hortalizas cultivadas en invernaderos y, segundo, se analiza la situación general, con la caracterización de la intrusión marina y de las opciones de abastecimiento futuro, particularmente la desalinización de agua de mar y su financiamiento.

En base a los impactos variables del uso del agua subterránea para los usuarios de fuera del acuífero y para el medio ambiente, una conclusión principal es que las propuestas de gobernanza del agua subterránea dependen del contexto, si bien deben incluirse como norma general dentro de la cuenca hidrográfica. La relevancia de instrumentos como los mercados de agua y los precios se discute también en este marco, particularmente en una situación en la que es frecuente el uso ilegal o la atribución excesiva de derechos. Se resalta también el papel de los incentivos y de los subsidios. Más generalmente, la transición hacia un mejor estado de los recursos de agua subterránea depende de la introducción en las decisiones de actores y valores que se encuentran muchas veces desconsiderados. Así, esta tesis ilustra la necesidad de conocimiento y de cambio de visión, primero, en relación con la disponibilidad del recurso y la gestión local y, segundo, en la perspectiva de las cadenas de valor, con la integración de los consumidores y otras entidades como incentivos para el cambio.

Resumen extendido

El agua subterránea es un recurso esencial para el desarrollo de muchas zonas del mundo, ya que, además de su buena calidad, está disponible tanto espacial como temporalmente, mientras que los recursos superficiales requieren a menudo obras importantes para su movilización. Esta disponibilidad espacial y temporal es también un beneficio desde el punto de vista ambiental, ya que ríos y extensas zonas húmedas se nutren de flujos de origen subterráneo. Sin embargo, el uso intensivo del agua subterránea ha generado una pérdida de servicios directos e indirectos (ligados a los ecosistemas asociados). La magnitud de estos cambios, muchos de ellos no considerados hasta hace poco, está aún por caracterizar. Además, la inclusión de valores ambientales implica la formulación de nuevas maneras de tomar decisiones y nuevos esquemas de gobernanza.

El objetivo principal de esta tesis es aprender del caso de España con el fin de formular propuestas para una mejor gobernanza del agua subterránea en un contexto general, donde la aplicación de la Directiva Marco del Agua (DMA) europea requiere la integración del medioambiente. La tesis trata de caracterizar el agua subterránea en España, es decir, su importancia, la huella humana y los valores y las visiones que incentivan su uso y fomentan las decisiones sobre su asignación. Se parte de una visión general a escala del país y de estudios más específicos de tres casos, en contextos que varían desde el punto de vista físico y del uso: el acuífero de La Mancha Occidental, la cuenca del Guadalquivir en su totalidad y el acuífero de La Loma de Úbeda como ejemplo específico y, finalmente, el acuífero del Campo de Dalías.

Se consideran también tres objetivos más específicos: 1) la formulación de un marco para la toma de decisiones sobre la repartición del agua subterránea integrada a escala de cuenca; 2) la evaluación del marco de la DMA para estimar el estado cuantitativo de las masas de agua subterránea y de su aplicación en los planes hidrológicos en España; 3) la relevancia del método de la huella hídrica para evaluar el uso del agua subterránea y mejorar la toma de decisiones.

Un cambio de perspectiva: de la formulación de criterios a la evaluación de la DMA

Es común presentar un acuífero como una reserva de agua en el subsuelo. Esta visión pone el énfasis sobre los flujos de entrada al acuífero, ya que constituyen la reserva. Sin embargo, esta tesis tiene como punto de partida la contribución fundamental del agua subterránea a los flujos superficiales y a los ecosistemas dependientes en la cuenca hidrográfica. Así, se pone de evidencia que los cambios en los niveles generados por las extracciones inducen una menor

salida de agua del acuífero (*decreased outflows*) y/o una mayor entrada (*increased inflows*), lo que afecta a los flujos superficiales. La suma de ambos efectos se define como 'captura'. La movilización de la captura es progresiva a medida que se propaga la perturbación inducida por el bombeo y las extracciones igualan la captura al equilibrio. Dependiendo de la localización de los bombeos, la contribución de las distintas fuentes de captura y el tiempo de movilización cambian. Si las extracciones no pueden ser compensadas por la captura, se observa un consumo continuo de la reserva.

En muchos casos los problemas asociados al consumo de la reserva adquieren importancia solamente después de que los impactos sobre los flujos superficiales son notables. Se debe tener en cuenta esta jerarquía de los impactos a la hora de tomar decisiones. Cuando las extracciones paran o se reducen, la captura y, por lo tanto, los impactos en términos de flujo no disminuyen hasta que los niveles del acuífero se recuperan en las áreas de captura. Por lo tanto, los bombeos tienen un efecto retrasado. Como contribución de esta tesis, se propone el término de 'captura futura' para referirse a este fenómeno y se muestra que las extracciones son iguales a la suma de la captura actual y de la 'captura futura'.

En relación con la toma de decisiones, la descripción de las distintas fuentes de captura, es decir, dónde, cuándo y cuánto se movilizan, es un elemento básico. Se trata de establecer límites concertados de conservación de los flujos superficiales y hacia los ecosistemas dependientes, con una visión integrada a escala de cuenca. La necesidad de este tipo de criterios muestra la perspectiva limitada que supone basarse solamente en la determinación de la recarga para definir los recursos disponibles de un acuífero, incluso si se resta un flujo ambiental, ya que oculta numerosos usuarios potencialmente afectados por los bombeos y el medioambiente.

El criterio habitual para la sostenibilidad de extracciones por debajo del valor de la recarga se basa en una visión equivocada de la dinámica del agua subterránea, que implica una identificación errónea de las condiciones para alcanzar el estado de equilibrio. Muchos autores han cuestionado también el hecho de que esta definición no incluya un balance de los valores ambientales, económicos y culturales. De hecho, en su aceptación común, la sostenibilidad es un objetivo general para la sociedad, que sugiere que la armonización de múltiples valores en un estado ideal universalmente aceptado es alcanzable. Sin embargo, este enfoque ha sido descrito como un 'concepto nirvana', ya que no contempla los numerosos *trade-offs* y costes implicados EN la mayoría de las decisiones. En base a estas constataciones, se clarifica el significado de los conceptos 'extracciones sostenibles' (*sustainable yield*) y 'seguras' (*safe yield*). Por otra parte, se destaca que en caso de múltiples orígenes de captura, no se pueden obtener los recursos disponibles localmente substrayendo a la recarga los flujos 'ambientales', ya que algún flujo 'ambiental' se vería afectado necesariamente.

La definición del agua subterránea como ‘Recurso Común’ (*Common-Pool Resource*, CPR) reproduce también una visión inicial discutible, ya que identifica a los acuíferos como reserva y a los usuarios directos como únicos usuarios con interés en la conservación del recurso. Así, la proposición de la acción colectiva de los usuarios directos tiene un punto de partida equivocado. El CPR debería definirse a escala de cuenca. Términos como ‘recursos renovables’ o ‘agua fósil’ están ligados también a una visión limitada del recurso y la relevancia de los modelos económicos tradicionales del agua subterránea y de sus conclusiones, como en el caso del muy debatido ‘efecto Gisser-Sánchez’, puede cuestionarse de igual modo. En relación a estos modelos, se muestra que la propia formulación del problema es cuestionable, ya que considera una tasa de descuento que implica un valor más alto para las extracciones actuales en comparación con las del futuro.

En cuanto a la aplicación de la DMA, a pesar de su objetivo principal de integrar la dimensión ambiental en la gestión de los recursos hídricos, se concluye, en base al análisis de los planes hidrológicos, que la evaluación práctica del estado de las masas de agua subterránea y de la disponibilidad de recursos reproduce un enfoque convencional. Esta situación tiene su origen en la presencia, en el propio texto de la DMA, de criterios basados en la evolución de los niveles de agua subterránea y la comparación de las extracciones con la ‘recarga’ menos unos flujos ambientales. Estos criterios prevalecen sobre los requisitos de estudiar en detalle las interacciones entre el agua subterránea y las masas superficiales y los ecosistemas, enunciados también en la DMA. Además, a pesar del objetivo de la Instrucción de Planificación Hidrográfica, el documento de referencia para la aplicación técnica de la DMA en las cuencas compartidas, de establecer métodos comunes, se observa una variedad de métodos y criterios entre unas demarcaciones hidrográficas y otras. Esto tiene implicaciones directas sobre la posibilidad de comparar la situación en diferentes cuencas. En relación con los métodos empleados, se destaca que los flujos ambientales son estimados muchas veces de manera uniforme y aproximada como un 20% de la ‘recarga’, que la definición de los ‘recursos disponibles’ no se integra a escala de cuenca y que es habitual que se computen los mismos flujos varias veces como recursos en diferentes masas, particularmente cuando se contabilizan las transferencias laterales entre las mismas. En respuesta a esta situación, esta tesis propone el termino de ‘recursos accesibles’ para evaluar el estado de una masa localmente.

No obstante, se ha de reconocer una serie de beneficios y avances gracias a la DMA, como la publicación sin precedentes de una gran cantidad de datos en relación con el conocimiento del estado y uso de las aguas subterráneas en España o la formulación de medidas para mejorar el estado del recurso. Según los datos presentados en los planes hidrológicos, el uso total de agua subterránea es de unos 6700 hm^3/año, lo que representa el 20 % del uso de agua en España. Se destinan tres cuartos del uso (5000 hm^3/año) al regadío, seguido por el abastecimiento urbano

(1400 hm^3/año) y las industrias con abastecimiento propio (300 hm^3/año). 296 (40 %) de las 712 masas de agua subterránea (sin considerar las islas Canarias) se clasifican en 'mal estado', 166 por su estado cuantitativo y 229 por su estado químico. Está previsto que la mayoría alcanzará el 'buen estado' en 2027 si bien se han establecido objetivos menos rigurosos para 39 masas.

La huella hídrica: origen, principales debates y avances de esta tesis

La huella hídrica mide la apropiación humana de recursos hídricos y su contaminación. En esta tesis, se consideran solamente las huellas hídricas azul y verde. La huella hídrica se computa como consumos de agua unitarios que se agregan en una parte o en la totalidad de una cadena de subministro (perspectiva del consumo) o para una zona geográfica delimitada (perspectiva de la producción). Se trata de un indicador flexible y multi-escala de los consumos directos e indirectos de agua. El concepto de huella hídrica fue propuesto en el marco de los debates sobre los flujos de agua virtual entre países y la idea de considerar los impactos ambientales en el país de origen. Un enfoque en términos de productos y de cadena de oferta fueron introducidos en siguientes desarrollos del concepto.

El concepto de 'apropiación' asociado a la huella hídrica se puede considerar según dos perspectivas. Primero, una 'huella' asocia el consumo de recursos directo e indirecto con una entidad de la economía. Este enfoque se relaciona con otras perspectivas de contabilidad de los flujos físicos, dentro de la economía ecológica por ejemplo. Segundo, la idea de 'apropiación humana' implica considerar un indicador adecuado de la presión humana. En este sentido, la huella hídrica integra el agua que no se puede usar de nuevo en la cuenca después de un uso, a diferencia de la consideración tradicional de las extracciones o uso del total. Así, la huella hídrica enlaza con otros métodos de contabilidad del agua previamente desarrollados, por ejemplo para el regadío. La consideración del agua verde permite introducir también varios debates relacionados con un mejor aprovechamiento de las precipitaciones en la agricultura o el análisis de las presiones en términos de uso del suelo. Sin embargo, las decisiones basadas en el agua verde son complejas y no se pueden formular solamente a partir de la contabilidad. La idea de mejorar la productividad del agua verde para reducir la presión sobre los recursos globales de agua azul es simplista y no integra la complejidad del sistema económico.

En los últimos años ha surgido un debate entre una huella hídrica volumétrica y un indicador de impactos, relacionado con el Análisis de Ciclo de Vida. El punto de partida es que el consumo de un volumen de agua no puede interpretarse directamente. Sin embargo, la formulación de un indicador de impacto es compleja y conlleva una dificultad adicional a la hora de interpretarlo y discutir los potenciales incertidumbres y errores. También se pierde la dimensión física de un volumen realmente consumido y se destaca que las opciones de reducción de los dos tipos de indicadores serán parecidas y complejas. Teóricamente, un estudio de huella hídrica tradicional implica también una evaluación de la sostenibilidad ambiental. Sin embargo, no hay consenso actualmente sobre su contenido exacto, y raramente se presenta en los estudios de huella hídrica. De igual modo, la 'sostenibilidad económica' y la 'sostenibilidad social', que forman parte supuestamente de un análisis de huella hídrica, se hacen de forma muy poco frecuente.

Como contribución de esta tesis, se formula el concepto de 'Huella Hídrica Integral' para referirse a la totalidad de los recursos hídricos consumidos en el ciclo de vida de un producto. Mientras la huella hídrica tradicional se limita a considerar el consumo de agua 'más arriba' en la cadena de subministro, la 'Huella Hídrica Integral' computa también el consumo de agua 'más abajo'. Esto permite identificar nuevas opciones de reducción, particularmente en relación con los desperdicios y las pérdidas de materia, ya que estos tienen también una huella. Este ejemplo es uno de los avances que la huella hídrica ha permitido identificar desde la perspectiva del consumo, junto con el tema de la equidad de la repartición de los impactos ambientales entre países o los impactos de ciertas pautas de consumo o de dieta. Desde el punto de vista metodológico, la tesis introduce también una serie de avances sobre la contabilidad de la huella hídrica. Se cuestiona de entrada la hipótesis de base de la mayoría de los estudios en cuanto a una satisfacción total de las necesidades de regadío. De hecho, ya que el regadío tiene lugar en zonas donde la disponibilidad de agua es limitada, usualmente las demandas no están satisfechas, incluso para el agua subterránea, ya que factores como el coste de bombeo o prácticas como el regadío deficitario limitan el uso. Por lo tanto, el cálculo de la huella hídrica en los casos de estudio se basa sobre estimaciones de uso real.

Caso de la Mancha Occidental

Un objetivo específico del caso de estudio del acuífero de la Mancha Occidental es caracterizar la huella hídrica del uso ilegal. La huella hídrica alcanzó 330 hm^3/año en 2009, con el viñedo como mayor componente (60 %), por delante de los cereales y las hortalizas (20 % y 15 % respectivamente). Este valor triplica el umbral necesario para mantener un flujo continuo hacía las Tablas de Daimiel, un Parque Nacional de alto valor ecológico, reconocido por la UNESCO. Sobrepasa también el objetivo de 220 hm^3/año establecido en el Plan Hídrologico del Guadiana.

Se pone de evidencia que los cultivos con un valor económico más alto y que generan más empleo presenten también una actividad ilegal mayor. El uso ilegal de agua generó más de la mitad de los ingresos y del empleo en el año 2008. Este dato refleja los obstáculos económicos y, por lo tanto, políticos, a la hora de actuar directamente contra el uso ilegal. La situación deriva particularmente de la repartición inicial de los derechos, vista como injusta, principalmente por los regantes de viñedo. El regadío de este cultivo estaba prohibido hasta el año 1995 y muchos no disponían de derecho. El Plan Especial del Alto Guadiana, activo en el periodo 2007-2013, cambió potencialmente esta situación de partida reasignando derechos comprados a los regantes de cereales a usos medioambientales y a la regularización del uso ilegal. Sin embargo, se resalta que el cambio de cultivo supone una posible intensificación del uso, ya que la demanda de regadío de los cereales era más variable de un año a otro. Los efectos positivos dependen también de un sistema regulatorio más efectivo. Finalmente, se cuestiona la posibilidad real de compatibilizar conservación ambiental y actividad económica en el actual orden de magnitud, como supuestamente han intentado las políticas hasta ahora. Particularmente, los bombeos deberían ser reducidos drásticamente durante los periodos de varios años secos, que ocurren con regularidad en la zona debido al clima semi-árido.

Caso del Guadalquivir

El caso del Guadalquivir es el único donde se considera la escala de una cuenca hidrográfica en su totalidad, integrando el agua verde y el agua azul total (origen superficial y subterráneo) en el periodo 1997-2008. La huella hídrica verde de la cuenca es de 7300 hm^3/año de media y la huella hídrica azul es 2800 hm^3/año. Como principales innovaciones de este estudio, se destaca 1) la consideración de la huella hídrica asociada a los embalses (315 hm^3/año), 2) la huella hídrica de las aguas subterráneas, estimando la componente que genera el consumo actual de flujos superficiales (730 hm^3/año) y la 'captura futura' (100 hm^3/año) y 3) la integración de las huellas hídricas verde y azul dentro de una contabilidad a nivel del ciclo hidrológico. La diferencia de productividad entre los distintos cultivos implica que se pueda conseguir un valor de la producción más alto usando menos agua. Sin embargo, se debe considerar el cambio de las condiciones de uso y la aceptabilidad de una reducción del mismo para algunos usuarios afectados. En relación con la huella hídrica verde, se puede cuestionar el enfoque como 'apropiación' de los recursos hídricos, ya que este concepto no es de aplicación directa cuando se trata de repartir el consumo de agua entre usos 'naturales' y humanos.

En los últimos veinte años se ha vivido un aumento sostenido del uso en el agua subterránea, principalmente asociado al regadío del olivar. Sin embargo, la integración de este consumo a escala de cuenca no ha sido considerada suficientemente. Por ejemplo, en el acuífero de La Loma de Úbeda, un recurso disponible de 40 hm^3 ha sido identificado. Este valor no integra el hecho de que este 'recurso' fluye naturalmente hacía los cauces superficiales. Sin embargo, ha sido la base para la regularización de parte de los usuarios no registrados. Con una huella hídrica total del acuífero que alcanzó 80 hm^3/año en el año 2010, en un contexto de clientelismo para la asignación de los derechos, no está claro si se tomarán acciones para reducir el uso. Cabe resaltar también el papel de las subvenciones en el desarrollo del olivar de regadío, y el hecho de que el aumento de la producción suponga la bajada de los precios, lo que disminuye la productividad del olivar tradicional de secano, desencadenando un 'circulo vicioso' hacía más regadío.

Caso del Campo de Dalías

La huella hídrica del Campo de Dalías es de 145 hm^3, con el uso principal (103 hm^3/año) vinculado a 16 500 ha de cultivo de hortalizas bajo plástico. La actividad se desarrolla durante un único ciclo largo, o dos ciclos del mismo cultivo o de dos cultivos, lo que genera un desafío a la hora de calcular la huella hídrica en comparación con los estudios 'tradicionales'. Además, se usa el agua para otras operaciones adicionales al regadío, como la desinfección del suelo. Esto resulta en una huella hídrica de las distintas hortalizas 'a puerta de finca' de 35 L/kg (pepino) a 90 L/kg (judías), lo que corresponde a una 'Huella Hídrica Integral' de entre 60 y 160 L/kg debido a las pérdidas en la cadena de subministro. El agua virtual exportada representa 66 hm^3/año y corresponde a 700 millones de euros. Factores como la producción invernal y el acceso al mercado europeo implican una productividad del agua alta (10 €/m^3), es decir un valor de más de 1000 millones de euros para toda la producción.

La geología del acuífero del Campo de Dalías es muy compleja. Sin embargo, se pueden distinguir dos sistemas acuíferos principales superpuestos. El principal origen del agua es el sistema acuífero inferior que tiene mayor calidad. Las extracciones han generado intrusión de agua del mar, lo que ha llevado al abandono de pozos donde el fenómeno es más agudo. El aislamiento parcial del mar de la mayor parte del sistema, por razones geológicas, ha facilitado que los bombeos pudieran seguir sin verse afectados por una intrusión indirecta y de magnitud limitada. Sin embargo, como análisis especifico de esta tesis se plantea la hipótesis de que esta particular configuración implique una intrusión marina continua en el futuro, incluso si se reducen los bombeos, lo que pone en peligro el uso del acuífero a largo plazo.

La principal propuesta de las autoridades para reducir los bombeos es el uso de agua residual y desalada. Sin embargo, los costes de producción crecientes y las condiciones de mercado generan incertidumbre para los regantes. Aunque el agua represente menos del 4 % de sus costes totales, los costes de la desalación serán principalmente asumidos por la demanda urbana. Distintos escenarios de recuperación de costes permiten discutir esta situación. Se pone en evidencia que el sistema de abastecimiento debería considerar las distintas fuentes de agua de manera conjunta, con todos los usuarios participando de manera transparente en su financiación. Por esta ausencia de transparencia y debate y, más directamente, por su coste, parece poco probable que la desalación sea usada en su capacidad total y remedie el deterioro del acuífero.

Consideraciones finales

Los casos de estudio muestran situaciones donde las extracciones de agua subterránea afectan a otros usuarios en la cuenca y al medioambiente de manera variable. En el caso del Guadalquivir, la disponibilidad del recurso se debe formular en coordinación con todos los usuarios de la cuenca. En la Mancha Occidental, hay que integrar los impactos ecológicos aceptables sobre las

Tablas de Daimiel. En el Campo de Dalías, los principales afectados son los usuarios del acuífero. Por lo tanto, las propuestas de gobernanza del agua subterránea deben ser moduladas dependiendo del contexto físico y de los impactos a distintas escalas. La acción colectiva por los usuarios directos de un acuífero no es una institución que se pueda proponer de manera general como única solución. Esta propuesta, como las derivadas de los modelos económicos tradicionales, proviene de una visión limitada del recurso donde el sistema de CPR es limitado al acuífero y a sus usuarios directos, en lugar de considerar toda la cuenca hidrográfica.

A nivel general, el uso ilegal o la asignación excesiva de derechos aparecen como las dos principales causas de uso excesivo de un acuífero. Cambiar esta situación es particularmente difícil debido al valor generado para la economía local. Además, todos los instrumentos propuestos para la regulación del uso de agua subterránea, como cuotas, mercados de agua o instrumentos de precio, deben tener en cuenta esta situación de partida, ya que su aplicación genera un cambio en las condiciones de uso que excluye a ciertos usuarios.

Una visión en términos de 'captura futura' y la existencia de una recarga efectiva permite resaltar que muchas veces existe una posibilidad de recuperar las reservas, como lo ilustran los casos de estudio. Esta debe ser la situación de base con la cual comparar el potencial éxito de las políticas. Sin embargo, los impactos retrasados puestos en evidencia por la 'captura futura' deben integrarse plenamente. También se resalta el papel de los incentivos, particularmente las subvenciones de la Política Agraria Común.

La transición hacía un mejor estado de los recursos de agua subterránea está condicionada a la inclusión de actores y valores que están tradicionalmente fuera de la toma de decisión. Por lo tanto, un mejor conocimiento o, mejor dicho, una manera distinta de ver un sistema socio-ecológico es determinante a la hora de tomar decisiones justas y transparentes e impulsar un cambio en la gestión. Esta idea está ligada con uno de los principales enfoques de esta tesis: describir los *trade-offs* asociados al uso del agua subterránea para que sean integrados en la toma de decisiones. La difusión del conocimiento debería ser promovida también para los actores de las cadenas de valor. La huella hídrica contribuye a este objetivo, ya que revelar los impactos reales del consumo puede contribuir a cambios en el mismo, a acciones para la diminución directa de la huella hídrica o a una demanda de regulaciones a nivel más general.

Este trabajo, por lo tanto, introduce una variedad de opciones para mejorar la toma de decisiones respecto a las aguas subterráneas – en general e ilustrado para España y tres casos de estudio –, mejorando la integración de valores divergentes y de los diferentes impactos generados por su uso. Particularmente, subraya los aspectos ligados a la decisión y a los incentivos a escala local y a la cadena productor-consumidor en la economía mediante la aplicación de la huella hídrica.

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Abbreviations and Acronyms

AEMET	Agencia Estatal de Meteología
CH	Confederación Hidrográfica
CPR	Common-Pool Resource
CWC	crop water consumption
DEM	Direct Employment
DEV	Direct Economic Value
DGAACM	Dirección General de Mejora de Explotaciones Agrarias y Centro de Estudios del Agua de Castilla-La Mancha
ET	evapotranspiration
GWB	groundwater body
JCCM	Junta de Comunidades de Castilla-La Mancha
MAGRAMA	Ministerio de Agricultura Alimentación y Medio Ambiente
MAPA	Ministerio de Agricultura Pesca y Alimentación
MARM	Ministerio de Medio Ambiente y Medio Rural y Marino
OCA	Oficina Comarcal Agraria
RS	remote sensing
SIAR	Servicio Integral de Asesoramiento al Regante
SIIA	Sistema Integrado de Información del Agua
SUGP	Special Upper Guadiana Plan
WF	water footprint
WF_{add}	additional water footprint out of the irrigation season (Campo de Dalías case study)
WFD	Water Framework Directive
WMA	Western Mancha Aquifer

Notes:
- Symbols used in equations are presented at the beginning of the chapter where they appear.
- Water volumes figures usually refer to consumption per year, however hm^3 is used instead of hm^3/year for more convenience (1 hm^3 = 1 million m^3).

Chapter I

INTRODUCTION AND GENERAL METHODOLOGY

I. INTRODUCTION AND GENERAL METHODOLOGY

I.1 Background

Access to groundwater is essential for many areas of the world since, in addition to its general good quality, it constitutes a resource available through space and time (Llamas & Custodio, 2002; Shah, 2010; Margat & van der Gun, 2013). On the one hand, groundwater is almost ubiquitous (BGR/UNESCO, 2008), which is a major benefit compared to surface water sources that can be expensive to mobilize because they require big infrastructures development (Llamas & Martínez-Santos, 2005). On the other hand, groundwater is available when other water sources fail, especially during droughts, whether this is in emergency situations or a planned conjunctive management with surface water (Sahuquillo, 1985; Garrido et al., 2006). Thus, groundwater allowed many regions to secure their access to water and develop, particularly through groundwater-based irrigation (Shah et al., 2006; Giordano & Villholth, 2007; Foster & Garduño, 2013). Worldwide groundwater withdrawals have been estimated at about 1500 km^3, representing 35 % of world water withdrawals, a share that rises to 42 % for irrigation with 1300 km^3 (Siebert et al., 2010; Döll et al., 2012). Groundwater may become even more important in the future as climate change may reduce access to surface water (Taylor et al., 2012).

The spatial and temporal availabilities of groundwater are also a major benefit for the environment. Spatially, extended wetland areas depend on shallow aquifers and are sensitive to moderate changes in groundwater levels. Temporarily, groundwater constitutes the base flow of many rivers or sustains wetlands during the dry season. Thus, in addition to the ‘direct’ service offered by a cheaper and reliable water supply, groundwater sustains a full range of ecosystem services (Bergkamp & Cross, 2006; Brauman et al., 2007). Yet the boom in groundwater use experienced in the last seventy years means that many of these services have been altered. For instance, rising pumping costs or quality degradation have limited the use of some aquifers or dependant wetlands and rivers have dried up (Custodio, 2002; Sophocleous, 2002).

The complexity of groundwater systems means that the full magnitude of this change has still to be assessed. With respect to environmental aspects, these tended to be disregarded in the initial development phase because of a general lack of knowledge and awareness, and consequently not included in policies and regulation. This has changed in the last twenty years as the potential impact of pumping on ecosystems, and the services they generate, have been increasingly recognised. It is especially acknowledged in regulations that explicitly consider surface water and groundwater integration and the conservation of ecosystems at the core of their objectives, such as the South Africa’s National Water Act of 1998 (Seward et al., 2007) or the European Union Water Framework Directive (WFD) (European Commission, 2000).

In Spain, like in many other places in the world, groundwater resources had a decisive and strategic role, e.g. for economic development in rural areas or for drinking water supply (Llamas & Garrido, 2007). In spite of its strategic importance, the emergent nature of groundwater use implied that conditions of use, such as the place and amount of withdrawals, have been poorly known. Before the implementation of the WFD, the last official review at the scale of the whole country of the situation of groundwater resources was undertaken in the year 1995 (*Libro Blanco de las Aguas Subterráneas*; Ministerio de Industria y Energía, 1995) and was updated in the year 2000 (*Libro Blanco del Agua*; Ministerio de Medio Ambiente, 2000).

The application of the WFD constitutes therefore an opportunity to have a renewed vision on the situation of groundwater use in Spain. The Water Plans elaborated by the river basin authorities should compile an unprecedented amount of data at aquifer scale in order to characterize the state of groundwater resources and the path towards compliance with the Directive in terms of 'good status' of water bodies. The transparency required in the whole process of the WFD implementation involves potentially having access to detailed data on the use of groundwater at the scale of the groundwater bodies, the basic management units introduced by the WFD, like the amount of withdrawals by economic sector or a detailed assessment of flows to and from aquifers[1]. It is noteworthy that, with regards to groundwater, the WFD contemplates the impacts of groundwater pumping on the ecosystems and surface water, which may contribute to the availability of new data on this specific issue.

I.2 The thesis

This section presents successively the objectives of the thesis, the methods introduced to complete these objectives, and the general structure of the thesis.

I.2.1 Objectives of the thesis

The main objective of this thesis is to gain insights for groundwater governance based on the case of Spain, through a characterization of groundwater and its use in this country, i.e. its physical extent, the human footprint, and the values and visions that incentivize its use and shape the decisions on its allocation, in the general context where the WFD constitutes a new framework for water resources management focusing on the environment. This is undertaken at two levels (Figure 1.1): 1) a general overview at the scale of the whole country and 2) a more comprehensive assessment of three case studies in contexts that differ in terms of both the scale of impacts from groundwater pumping, and the incentives for groundwater withdrawals.

[1] At the time the thesis begun, a first synthesis, the assessment of the groundwater bodies at risk in Spain, had been published in compliance with the WFD (Ministerio de Medio Ambiente, 2006). However, this corresponded to a specific task of a work still in progress (see Chapter 4 on groundwater in Spain).

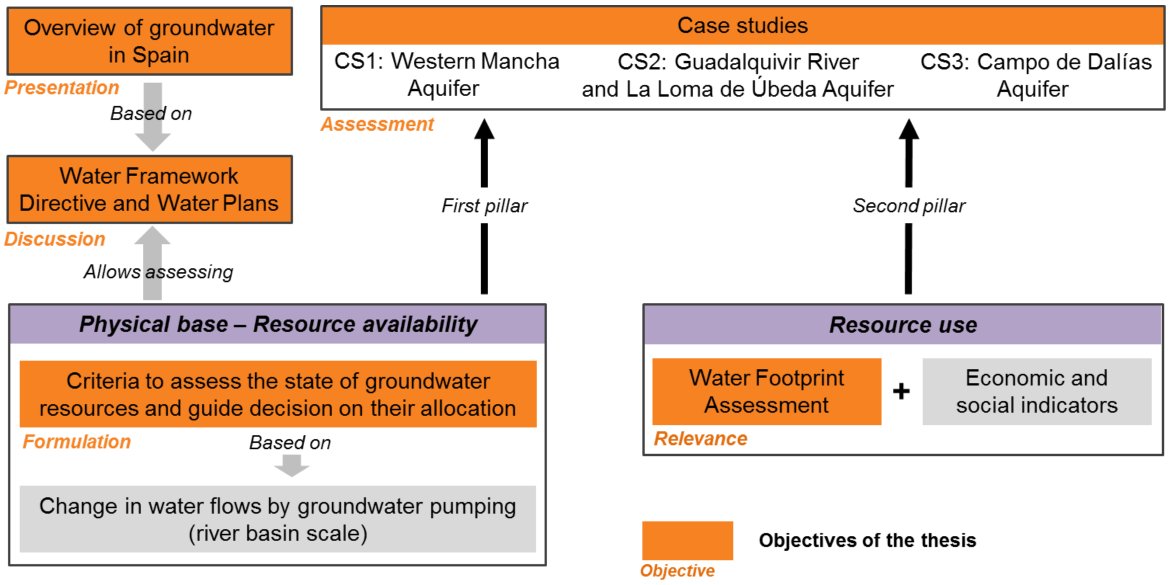


Figure 1.1. Objectives and general organization of the thesis, and methodological approach based on two pillars: 'physical base' and 'resource use'.

Three specific objectives directly related to this assessment are also contemplated (Figure 1.1):

1. *The formulation of a framework to guide decision on groundwater allocation within water resources management at the river basin scale*. This includes the central issue of the definition of resource availability for the users of an aquifer. The reason is that criteria are needed to assess both the WFD implementation in relation to groundwater quantity and the conditions of groundwater use and the policies in the case studies. More generally, the fact that usual rules for groundwater resource availability and allocation – such as the identification of recharge as the available resource – integrate poorly the role of groundwater flows in the watershed, specially for the environment, also justifies this objective.

2. *The evaluation of the criteria introduced in the WFD for groundwater availability definition, the assessment of the quantitative status of groundwater bodies and their implementation in the Water Plans in Spain*. A first reason for this task is to contemplate the ability of the criteria to effectively respond to the objectives of the WFD, particularly in terms of a better integration of environmental issues in water resources management. Another reason is that the overview of the use and state of groundwater in Spain is undertaken from the data and results presented in the Water Plans and depends therefore on the quality and relevance of the information presented (Figure 1.1).

3. *The interest and relevance of the water footprint (WF) approach to report the use of groundwater resources and improve decision-making*. The WF approach is introduced

to report water use in the different case studies. This thesis is initially the follow-up of WF assessments in Spain, where groundwater issues have been poorly integrated. For instance Garrido et al. (2010) presented a detailed assessment of the WF, in terms of both consumption of internal resources and imports and exports of virtual water. In this work, groundwater is included within blue water use, mainly because of the lack of specific data on this resource. Hence there is a need for a more in depth assessment the role groundwater plays. Moreover, the WF approach has only recently taken hold (see Hoekstra et al., 2011; Zimmer, 2013) and the actual contributions and advances of an approach in terms of WF are an unresolved issue. More insights on the actual potential of this indicator are necessary.

In addition to these general objectives of the thesis, the three case studies – the Western Mancha Aquifer, the Guadalquivir River Basin with a special focus on the La Loma de Úbeda Aquifer, and the Campo de Dalías Aquifer – have their own specific objectives that are detailed in Section 2.3 of this chapter.

I.2.2 General methodology of the thesis

- Overview at the scale of Spain

The overview of the state of groundwater resources in Spain is based on the data contained in the Water Plans elaborated in application of the WFD (Figure 1.1). All Water Plans in Spain were published over the period 2010-2013, at least in their draft version. This step mainly consists in the compilation of data relative to the use of groundwater by sector of the economy (agriculture, industry, urban supply, etc.), the resource availability quantification for each groundwater body and the quantitative and chemical status of the groundwater bodies.

- General approach for the three case studies

The general approach for the three case studies consists in two pillars (Figure 1.1):

1. *A 'physical base' pillar*. The framework formulated in this pillar, as a specific objective of the thesis, allows assessing the issues related to the availability and impacts of groundwater use in the different case studies, which change according to the physical setting, e.g. groundwater flows can be essential for a valuable ecosystem, to maintain river flows during droughts, or to prevent seawater intrusion. The adequacy of the resource availability definition in the current policies, decision-making processes and actors' visions on the resource can be then discussed for each case study.
2. *A 'resource use' pillar*. It contemplates the purpose of groundwater resources use, its drivers and the value generated. The physical accounting of water use is based on the WF approach. The agricultural WF is detailed for the main crops grown in the study

areas, since agriculture is responsible for the majority of groundwater use in Spain. The WF of the other relevant economic sectors is considered as a whole. The WF is complemented by economic indicators in terms of revenue and employment generated by the use of water.

Specific methods relative to the particular objectives of the different case studies are detailed in the corresponding chapters. It is noteworthy that information and data on the case studies have been specifically obtained during a series of interviews and meetings with different stakeholders (see Appendix 1).

- Criteria to guide decision on groundwater resources allocation and availability

Groundwater outflows from aquifers or high groundwater levels can sustain rivers, lakes, wetlands or dependent ecosystems. Thus, the starting point relative to this objective is to highlight the trade-offs in space and time regarding the availability of water flows in the river basin on the premise that, as stated by Theis (1940), 'all water discharged by wells is balanced by a loss of water somewhere' (Figure 1.1). These trade-offs should be identified and integrated in decision-making to define the acceptability of the impacts. The physical flows description is mainly based on the review and analysis of the hydrogeology literature that relates the dynamics of groundwater pumping to the implications for surface water flows and groundwater dependant ecosystems. An innovative aspect in this thesis is that it is not only the consequences of aquifer development that are considered but also the 'exit situation', when pumping stops or is reduced. These issues relate to the debate on 'sustainable use' of groundwater resources, which is reviewed critically in this thesis.

- Assessment of the WFD criteria on quantitative status and groundwater resource availability

The assessment of the methods to determine the available groundwater resource and the groundwater bodies' quantitative status in both the WFD text and its application in the Water Plans is based on the criteria for the allocation of groundwater resources and for groundwater resources state characterization that are previously introduced in the thesis (Figure 1.1).

- Relevance of the water footprint approach

This point is mainly assessed through the application of the WF in the case studies. This adds to the critical literature review on this concept that discusses its origins and the main developments and challenges in its formulation and clarifies the perspective adopted when using the WF in this thesis.

I.2.3 Selection of case studies and specific objectives for each case study

In order to show the conditions of use and challenges for the management of groundwater in different contexts, three main case studies are analysed (Figure 1.2). They represent diverse conditions, from the physical and economic point of view (Table 1.1).

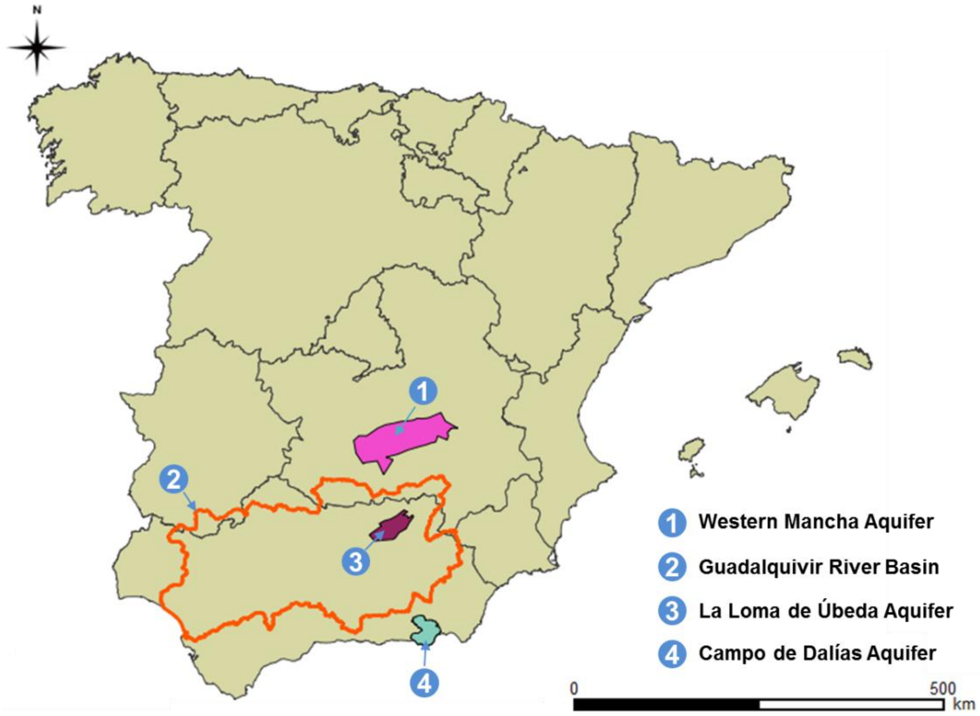


Figure 1.2. Localization of the case studies.

- <u>Western Mancha Aquifer.</u>

The Western Mancha Aquifer is a shallow aquifer with an easy access to groundwater resources under natural conditions. The main issue in relation with groundwater management is the conservation of the Tablas de Daimiel wetland, a National Park and an UNESCO Man and the Biosphere Reserve classified under the Ramsar Convention. The intensive use of groundwater, which is almost entirely dedicated to irrigation, has impacted this wetland during the last three decades with an important reduction of the flooded area during dry periods. The case study is located within La Mancha vineyard region. Vine irrigation is the main destination of groundwater followed by cereals. As well as the overall objective of characterising the use of groundwater in the area, there is also a specific aim to assess the illegal use through a WF approach. A recent major policy, which has tackled both the issue of illegal use and necessity to reallocate flows for environmental recovery of the Tablas de Daimiel wetland, the Special Upper Guadiana Plan (CH Guadiana, 2007), is also reviewed.

- <u>Guadalquivir River Basin, with a focus on the La Loma de Úbeda Aquifer.</u>

The Guadalquivir River Basin covers the major part of the Andalusia region. It is one of the largest in Spain and presents a diversity of local conditions. This case study is the only one to deal with the river basin scale as well as green water consumption. The main reason for having selected this river basin is the large and uncontrolled increase in groundwater use experienced in the last twenty years. It is mainly ascribed to olive groves irrigation in the upper part of the basin. La Loma de Úbeda Aquifer, a carbonate aquifer that is confined on much of its area, serves as an illustration for this dynamic regarding olive groves irrigation and the problem of groundwater resource availability definition in an aquifer that is directly associated with a river.

Table 1.1. Specificities of the different case studies relatively to the physical setting, the use of groundwater and their objectives.

Case study	Physical setting	Main uses and issues	Specific objectives
Western Mancha Aquifer	Groundwater sustaining a wetland	Vines irrigation as the main use Cereals and outdoor vegetables irrigation	Accounting the illegal activity in terms of WF and socio-economic implications Drivers for illegal use Implications for policies and assessment of the Special Upper Guadiana Plan
Guadalquivir River Basin	View at river basin scale All the aquifers receive inflows that potentially contributes to sustaining river flows and ecosystems	Role of groundwater in the whole river basin Inter-sector allocation of water resources	Green and blue WF accounting with a focus on groundwater Whole balance of water flows at the scale of a river basin Assessment of the possibilities of water resources reallocation
La Loma de Úbeda Aquifer	Aquifer connected to a river stream	Almost entirely used for olive groves irrigation	Assessment of a local economy based on olive groves irrigation
Campo de Dalías Aquifer	Coastal aquifer subject to sea water intrusion	Export-oriented greenhouse agriculture Conjunctive use of desalinated water and groundwater Groundwater also essential for urban supply and tourism	Extent of sea water intrusion Discussion of the rationale for use of desalinated water Assessment of the future 'supply mix' (water / energy footprints) Cost-benefits and cost recovery of the future 'supply mix' Future of greenhouse agriculture

- Campo de Dalías Aquifer.

The Campo de Dalías Aquifer is a coastal carbonate aquifer located in the Almería Province that is subject to sea water intrusion. It is almost entirely covered with greenhouses for an export-oriented vegetable production during winter that led to high economic returns for farmers. The aquifer is also a major source for urban supply and tourism. A specific issue of the case study is the introduction of desalination to secure future water supply and to reduce groundwater withdrawals to comply with the WFD. In addition to assessing the extent of sea water intrusion, specific objectives include the analysis of the rationale for the choice of desalination and the assessment of the costs of the new supply mix in energy and financial terms.

Although they differ through a series of parameters, the main common characteristics of the three case studies are: first, agriculture is the main use; second, the aquifers receive inflows from surface water bodies and rainfall percolation, i.e they are unconfined, at least partially; third, they all are located in area with a semi-arid climate, characterized by high intra- and inter-annual rainfall variability, which is accentuated for recharge since the relation between rainfall and recharge is not linear. This has two main implications: 1) the inflow variability represents a challenge for management and, 2) it means that theoretically the level of the water table can be recovered in a few years, if withdrawals are strongly reduced, as there is a recharge[2].

I.2.4 Organization of the thesis

The thesis begins with the description of temporal and spatial trade-offs associated with the access to groundwater (Chapter 2) to establish a set of criteria to assess the state of groundwater resources and to guide decision-making on their allocation (Chapter 3). The completion of this objective is a central pillar for the correct introduction and characterization of the other aspects of the thesis (see Figure 1.1). Chapter 3 also clarifies a series of common misleading approaches to groundwater management and terms that are sometimes considered as robust technical terms, but which are in fact, as will be discussed, associated with a partial vision of the groundwater, such as ‘availability’, ‘renewable resource’, or ‘groundwater mining’. Chapter 4 discusses the criteria introduced in the WFD text to assess the quantitative status of groundwater bodies and their available resources, and the actual implementation of these criteria in the Spanish Water Plans. It particularly allows understanding the scope and limits of the overview of groundwater resources in Spain that is presented at the end of the chapter.

Chapter 5 is dedicated to the introduction of the WF approach, the methodological issues involved, as well as the discussion of the relevance of some aspects linked to its interpretation. The specific methods for the case studies, including the WF application and the other economic indicators and general data used are then introduced in Chapter 6. The three case studies are presented in Chapters 7 (Western Mancha Aquifer), 8 (Guadalquivir River Basin and La Loma de Úbeda Aquifer) and 9 (Campo de Dalías Aquifer). The main contributions of the thesis are then synthetized in the concluding chapter (Chapter 10).

Thus, two parts can be distinguished in the overall thesis structure. The first part is basically dedicated to the formulation of criteria to guide decisions on groundwater resources allocation and the assessment of the WFD criteria based on the example of Spain. The second part focuses on the case studies and the WF, which is the main common approach for the case studies.

[2] This situation is different from, for instance, a desert where inflows to aquifers are almost inexistent or a totally confined aquifer, which cannot be considered as having any interaction with surface water bodies.

Chapter II

GROUNDWATER PUMPING AND FLOWS IN THE WATERSHED

List of symbols of this chapter

$c(t)$	capture rate (m^3/s)
c_f	future capture (m^3/s)
$d(t)$	decrease in discharge rate (m^3/s)
$D(t)$	current rate of outflows from the aquifer (m^3/s)
D_0	natural rate of outflows from the aquifer (m^3/s)
$d_i(t)$	rate of decrease in discharge in the area of discharge *i* (m^3/s)
d_{max}	maximum rate of decrease in discharge for the whole aquifer (m^3/s)
$d_{max,i}$	maximum rate of decrease in discharge in the area of discharge *i* (m^3/s)
d_s	rate of decrease in discharge for the whole aquifer at steady state (m^3/s)
$d_{s,i}$	rate of decrease in discharge in the area of discharge *i* at steady state (m^3/s)
$P(t)$	rate of pumping (m^3/s)
P_s	pumping at steady state (m^3/s)
$r(t)$	increase in recharge rate (m^3/s)
$R(t)$	current rate of inflows to the aquifer (m^3/s)
R_0	natural rate of inflows to the aquifer (m^3/s)
$r_j(t)$	rate of increase in recharge in the area of recharge *j* (m^3/s)
r_{max}	maximum rate of increase in recharge for the whole aquifer (m^3/s)
$r_{max,j}$	maximum rate of increase in recharge in the area of recharge *j* (m^3/s)
R_{max}	rate of total recharge of the aquifer (natural and induced by pumping) (m^3/s)
r_s	rate of increase in recharge for the whole aquifer at steady state (m^3/s)
$r_{s,j}$	rate of increase in recharge in the area of recharge *j* at steady state (m^3/s)
$s(t)$	rate of change in groundwater stock (also $dV(t)/dt$) (m^3/s)
s_s	rate of change in groundwater stock for the whole aquifer at steady state (m^3/s)
$V(t)$	volume of the groundwater stock (m^3/s)

II. Groundwater Pumping and Flows in the Watershed

II.1 Introduction

The potential adverse consequences of aquifer development have been considered for a long time by hydrogeologists. While aquifer stocks were sometimes initially viewed as inexhaustible underground seas, intensive withdrawals allowed by engine pumps and borehole techniques since the second quarter of the twentieth century have highlighted a number of limits to groundwater use. In addition to decreasing levels and rising pumping costs, this intensive development led to impacts such as the drying of wetlands, land subsidence, or decreases in spring or river flows, which suggested the necessity for limiting pumping in a number of situations to maintain the benefits of groundwater use or avoid undesired consequences.

The reference to ‘sustainable use’ or ‘sustainability of groundwater use’ has been common in the last two decades, in line with the popularization of the concept of sustainability (Alley & Leake, 2004; Devlin & Sophocleous, 2004). This concept may appear to be adapted to guide groundwater management, since it refers clearly to the issue of intergenerational equity and has a strong environmental dimension (see Chapter 3, for a more detailed introduction of the sustainability concept). Nevertheless, these considerations are not new and can be associated with the notions of ‘safe yield’, ‘sustained yield’ or ‘over-exploitation’ of aquifers (Figure 2.1). The determination of the availability of resources from an aquifer is an issue that is also directly related. Part of the economic literature tackles the optimal or efficient use of an aquifer. These are different approaches to a same basic issue: the determination of the amount of withdrawals that can be obtained from an aquifer, according to the value of groundwater for society.

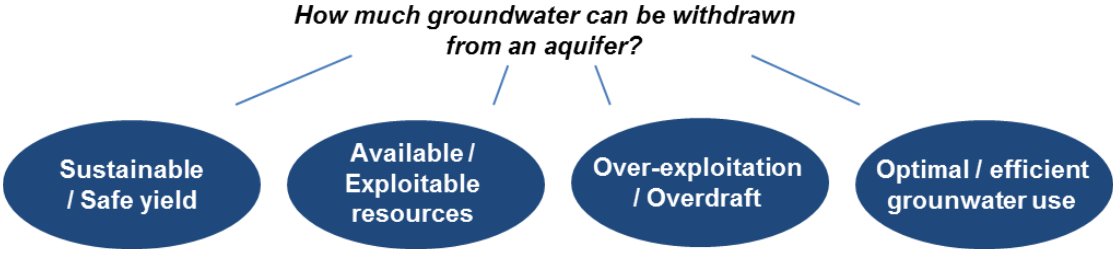


Figure 2.1. Different concepts related to the withdrawals that can be obtained from an aquifer.

This chapter and the following one are in line with these debates as they aim at establishing a framework for decision on the allocation of groundwater. The multiple criteria for defining sustainability or optimal use that have been proposed make it necessary to present a clear framework on how these questions are contemplated in this thesis. This is also pertinent to discuss the implementation of the WFD and the situation of the different case studies on a sound basis. This chapter introduces a detailed description of the change in water flows in the watershed that are due to pumping. It mainly shows that pumping generates, on the one hand, aquifer stock consumption and, on the other hand, increase in recharge and decrease in discharge, i.e. disruption of flows towards surface water bodies and ecosystems, with varying proportion through time.

The premise is that the criteria used for decisions must reflect all relevant implications, in terms of both value generated by withdrawals and disruption of natural water flows, thus highlighting the trade-offs. The traditional vision of an aquifer as an underground reservoir replenished by a recharge constituting the renewable resource fails to identify a number of dimensions linked to the allocation of groundwater. Therefore the description of the physical dynamics of changes in water flows in this chapter highlights aspects that are often disregarded in traditional visions of groundwater.

II.2 Natural situation, the role of groundwater in sustaining flows in the watershed

In the same way the topography of the land surface controls surface water flows, the destiny of water that infiltrates into the ground can be visualised through the ‘piezometric topography’ (Figure 2.2). This ‘topography’ is defined by groundwater heads and constitutes a basic determinant of hydrological flows since it governs inflows and outflows to the underground[3].

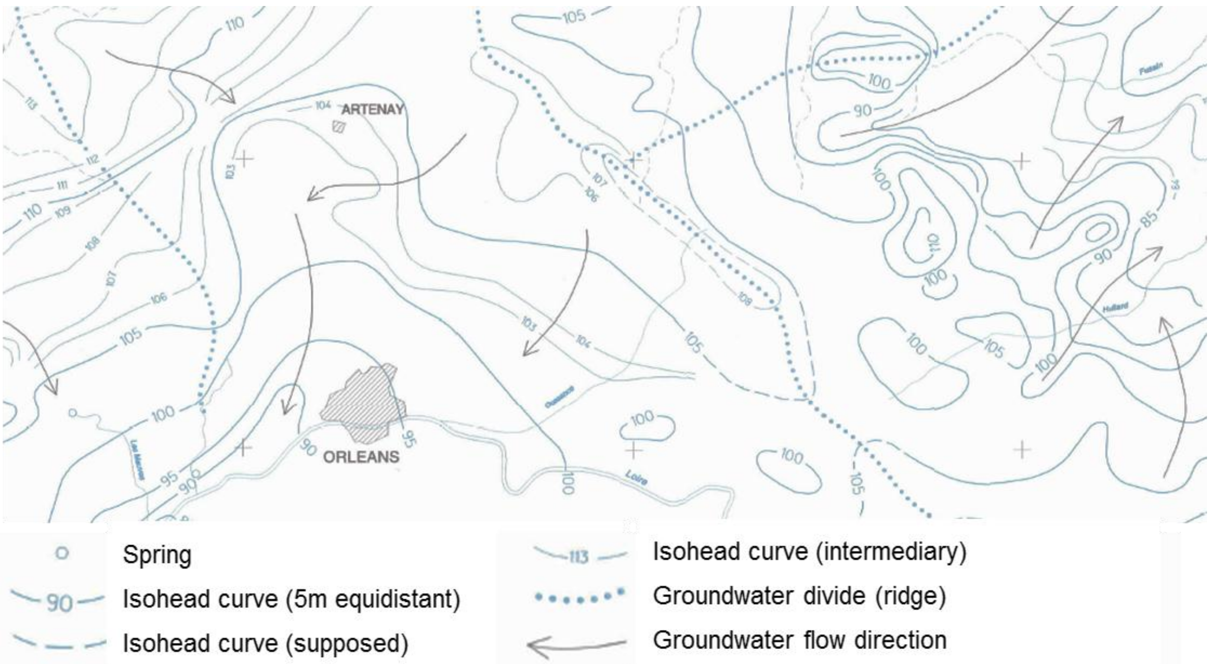


Figure 2.2. Topography of groundwater table defined by the isohead curves. Source: BRGM (1995).

[3] The ‘piezometric topography’ is constituted by groundwater itself. It implies that the dynamic of groundwater flow should not be compared to the dynamic of surface water flowing on the land surface.

The groundwater table sustains surface water flows and groundwater dependent ecosystems, which is observed when the groundwater and land 'topographies' intersect or are close one to another. It basically corresponds to two situations:

- Areas of discharge flows from the aquifer (e.g. springs, wetlands or rivers), directly or through evapotranspiration (phreatophyte plants and capillarity) (Figure 2.3 and 2.4).
- Areas where the groundwater table reaches the land surface and a 'rejected recharge' occurs – water that cannot percolate into the aquifer because it is full (Theis, 1940, see Figure 2.4). Surface water flows are then generated thanks to the high groundwater level. In particular, a share of most river flows can be considered as a 'rejected recharge', since it would percolate in the alluvial aquifer if groundwater levels were lower (i.e. a losing stream instead of a gaining stream).

Thus, through these interactions, high groundwater tables induce surface water flows and evapotranspiration. The specific temporality of groundwater flows implies that river base flows or many wetlands are fed mostly by groundwater, particularly under arid and semi-arid climate, with a high seasonal and annual rainfall variability.

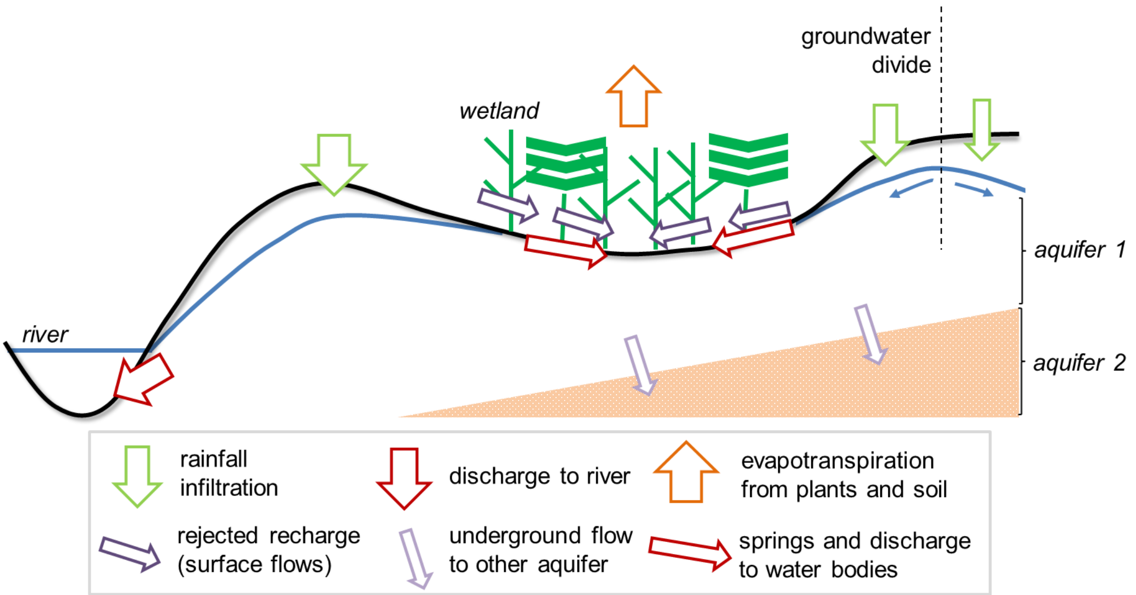


Figure 2.3. Natural budget of an aquifer and the groundwater divide.

Another element indicating the destiny of groundwater is the line of highest head, which forms a ridge that delimitates the different groundwater basins: the groundwater divide (Figure 2.2). It is similar to the drainage divide of a watershed, with the difference that it can move according to the variations in groundwater levels on both sides. Finally, the underground flows between different aquifer layers should also be accounted for.

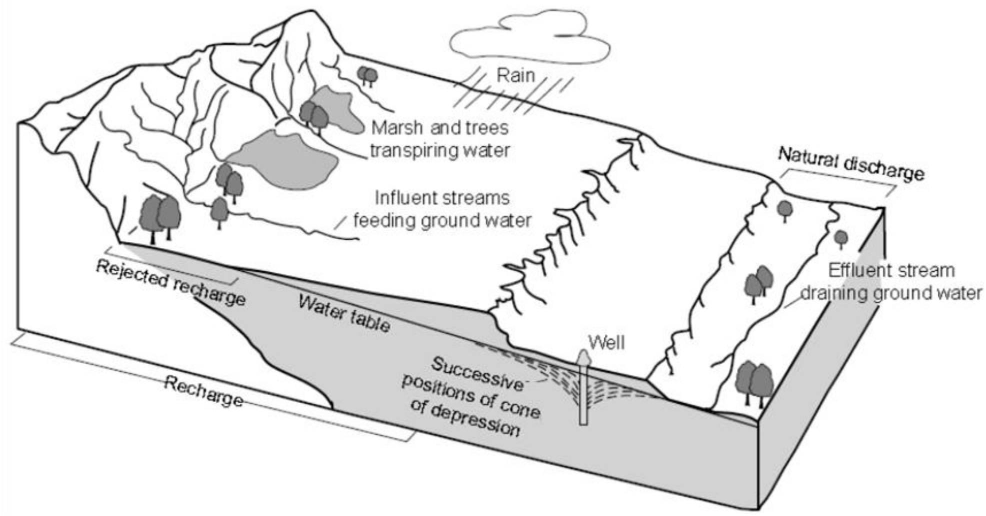


Figure 2.4. Factors controlling the response of an aquifer to discharge by wells. Source: Sophocleous (1998) (adapted from Theis, 1940)

II.3 Alteration of water flows: mobilizing the capture

II.3.1 Change in the water budget and the capture concept

When groundwater is pumped from an aquifer, the 'groundwater head topography' is modified. The groundwater table drops initially in the surroundings of the well and this perturbation is progressively propagated within the aquifer (Theis, 1940; Alley et al., 1999; Custodio, 2002). Thus, pumping creates a drawdown and it is illusory to assume that the level can be maintained in its natural state. Initially, pumping comes entirely from aquifer storage. Once the perturbation reaches an area of interaction between the groundwater and land topographies or a groundwater divide, the rates of inflows into and outflows from the aquifer change (based on Theis, 1940):

- In areas of discharge, a lower groundwater table generates a decrease in the discharge rate (d(t), in m^3/s), e.g. flows to springs and rivers decline or the phreatophyte plants are no longer able to reach the groundwater table.
- In areas of rejected recharge, more water percolates into the ground as groundwater does not reach the surface anymore. Pumping results in an increase in recharge rate (r(t), in m^3/s) and less surface water flows are generated. Typically, increased recharge can take place over the desiccated areas of a wetland or more water infiltrates from a river stream (conversion of a gaining stream in a losing stream).
- In case the perturbation reaches a groundwater divide, its position moves and groundwater that was previously flowing towards an adjacent groundwater basin flows towards the basin where pumping occurs, which triggers an increase in inflows[4].

[4] The change in the groundwater divide implies that the adjacent aquifer experiences a reduction in discharge that should be considered.

These are the three only possibilities for an aquifer system to get water (through increased inflows) or retain water (through decreased outflows) to balance pumping (Theis, 1940)[5]. Moreover, in the case of a coastal aquifer, it can be noted that the sea constitutes a source of increased recharge in case of flow inversion, which triggers seawater intrusion.

The sum of the increased inflows and the decreased outflows has been termed as 'capture' (c, in m^3/s) (Lohman et al., 1972; Bredehoeft et al., 1982):

$$c(t) = d(t) + r(t) \tag{2.1}$$

It makes sense to sum both factors because they consist in a reduction of surface flows generation or evapotranspiration from the aquifer as compared to the natural situation. These flows are 'captured' by pumping. It is important to note that the capture is mobilized thanks to the necessary drop in the groundwater level resulting from pumping.

A water budget of the aquifer allows formulating pumping as a function of capture. Assuming a situation where the groundwater flows out the aquifer from N areas of discharge and can potentially get in the aquifer through M areas of induced recharge, we have[6]:

'outflow rate' = 'inflow rate' + 'stock depletion rate'

$$D(t) + P(t) = R(t) + dV(t)/dt \tag{2.2}$$

$$D_0 - \sum_{i=1}^{N} d_i(t) + P(t) = R_0 + \sum_{j=1}^{M} r_j(t) + dV(t)/dt \tag{2.3}$$

with $D(t)$ and D_0 the current and natural rate of outflows from the aquifer respectively;
$R(t)$ and R_0 the current and natural rate of inflows to the aquifer respectively;
$P(t)$, the rate of pumping;
$V(t)$ the volume of the groundwater stock;
$dV(t)/dt = s(t)$ the rate of variation of the groundwater stock;
$d_i(t)$ the decrease in discharge in the area of discharge i;
$r_j(t)$ the decrease in discharge in the area of discharge j.

In the natural state, and considering a sufficient period of time to take into account the variation of inflows and outflows, the water budget can be expressed as:

$$D_0 = R_0 \tag{2.4}$$

[5] In addition, underground flows transmitted between aquifers should also be considered. It is also noteworthy that inversion of the direction of the flows is a common case of groundwater contamination, when an adjacent geological layer contains contaminants.

[6] This type of budget is common to show consequences of pumping in terms of capture and, particularly, to debunk the 'Water budget myth', which is a common erroneous budget formulation (e.g. Bredehoeft et al., 1982; Devlin & Sophocleous, 2004). Here, it is specifically generalized for multiple sources of capture.

hence, $$P(t) = \sum_{i=1}^{N} d_i(t) + \sum_{j=1}^{M} r_j(t) + dV(t)/dt \qquad (2.5)$$

or: $$P(t) = d(t) + r(t) + s(t) = c(t) + s(t) \qquad (2.6)$$

Equation 2.6 illustrates that groundwater pumping induces capture and aquifer stock depletion[7].

II.3.2 Transient stage

On the basis of Equation 2.6, it can be deduced that, at the beginning of pumping, when the perturbation generated by pumping has not reached an area of capture (i.e. $d(t) = r(t) = 0$) or a potential groundwater divide, pumping exclusively induces aquifer stock depletion:

$$P(t) = s(t) = dV(t)/dt \qquad (2.7)$$

Subsequently, the capture is progressively mobilized ($r(t)$ and/or $d(t)$ rises) and the contribution of the aquifer stock is progressively reduced, if pumping is constant (see Equation 2.6). Figure 2.5 presents the case of an aquifer connected to a river stream, a well-known and illustrative case on the evolution of the contribution of storage and capture in response to pumping[8].

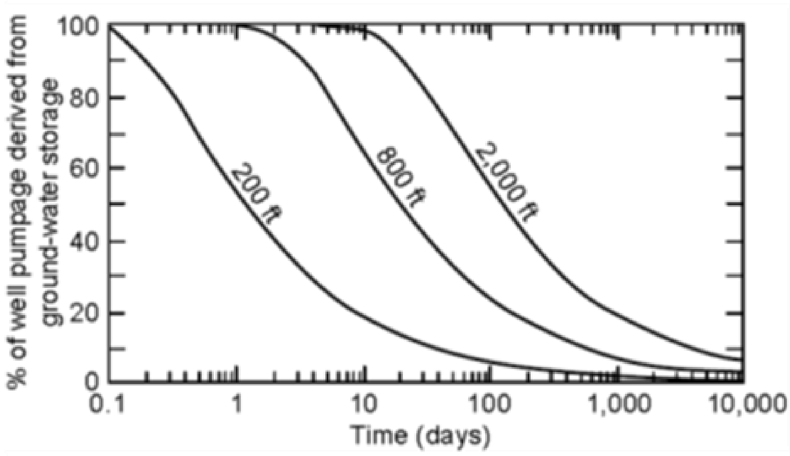


Figure 2.5. Transition of growth curves for wells located at three different distances from a stream. Source: Sophocleous (1998)

The distance to the river stream (i.e. distance to a potential source of capture) is an essential parameter, since it constitutes one of the factors that govern the time to start mobilizing the source of capture, the time needed to reach the steady state, and the total final amount of stock depletion (Figure 2.5). Kalf & Woolley (2005) present a more general case that distinguishes

[7] Equation 2.6 defines a water budget at the scale of the whole aquifer; it does not imply that the water that is pumped is effectively the increased recharge and decreased discharge. Conversely, it is not because the groundwater that is effectively pumped comes entirely from the stock (e.g. if it is testified that it has infiltrated centuries ago) that no capture takes place at the boundaries of the aquifer.

[8] It is noteworthy that the river stream is potentially subject to decreased outflows (less water flows to the river) and increased inflows (some portions of the river are turned into a losing stream).

the evolution of both outflows and inflows. Theis (1940), Sophocleous (2002), Bredehoeft et al. (1982), Custodio (2002) and Zhou (2009), among others, present similar graphs to underline the evolution during the transient stage.

II.3.3 Identification of two regimes at the steady state

From now on, it will be assumed that the potential sources of capture can be mobilized in a reasonable planning horizon, i.e. few years or decades. The specific case where a continuous consumption of the storage takes place will also be discussed in Section 5 of this chapter, which contemplates the situation where the time to mobilize the capture is high.

Equation 2.6 can be simplified for the steady state that is attained after a certain time of pumping (with a constant pumping, P_s). It is characterized by values of r(t), d(t) and s(t) that are constant (r_s, d_s and s_s, respectively). Two situations should be contemplated:

(1) The value of capture is sufficient to compensate for pumping, the groundwater level is stabilized ($dV(t)/dt = s_s = 0$) (Figure 2.6, c and d) and:

$$P_s = r_s + d_s = c_s \quad (2.8)$$

That is to say that pumping equals capture at steady state.

(2) The value of capture is not sufficient to compensate for pumping and there is a constant contribution of the stock s_c, accompanied by a constant drop of the level (Figure 2.6, e). In this case, r(t) and d(t) have necessarily reached their maximum value, r_{max} and d_{max} respectively:

$$P_s = r_{max} + d_{max} + s_s \quad (2.9)$$

Reaching d_{max} means that the natural outflows have been totally appropriated by pumping, i.e. there is no more outflows and d_{max} is equal to the natural recharge (R_0).

In situation (1), the final impact of pumping is on the surface flow and ET generation[9]. Thus, it can be assumed that pumping triggers only *flow impacts* (r_s and d_s in Equation 2.8).

In situation (2), Equation 2.9 can be generalized for a variation of P according to time. This expression remains valid as long as the value of capture remains the same (at its maximum):

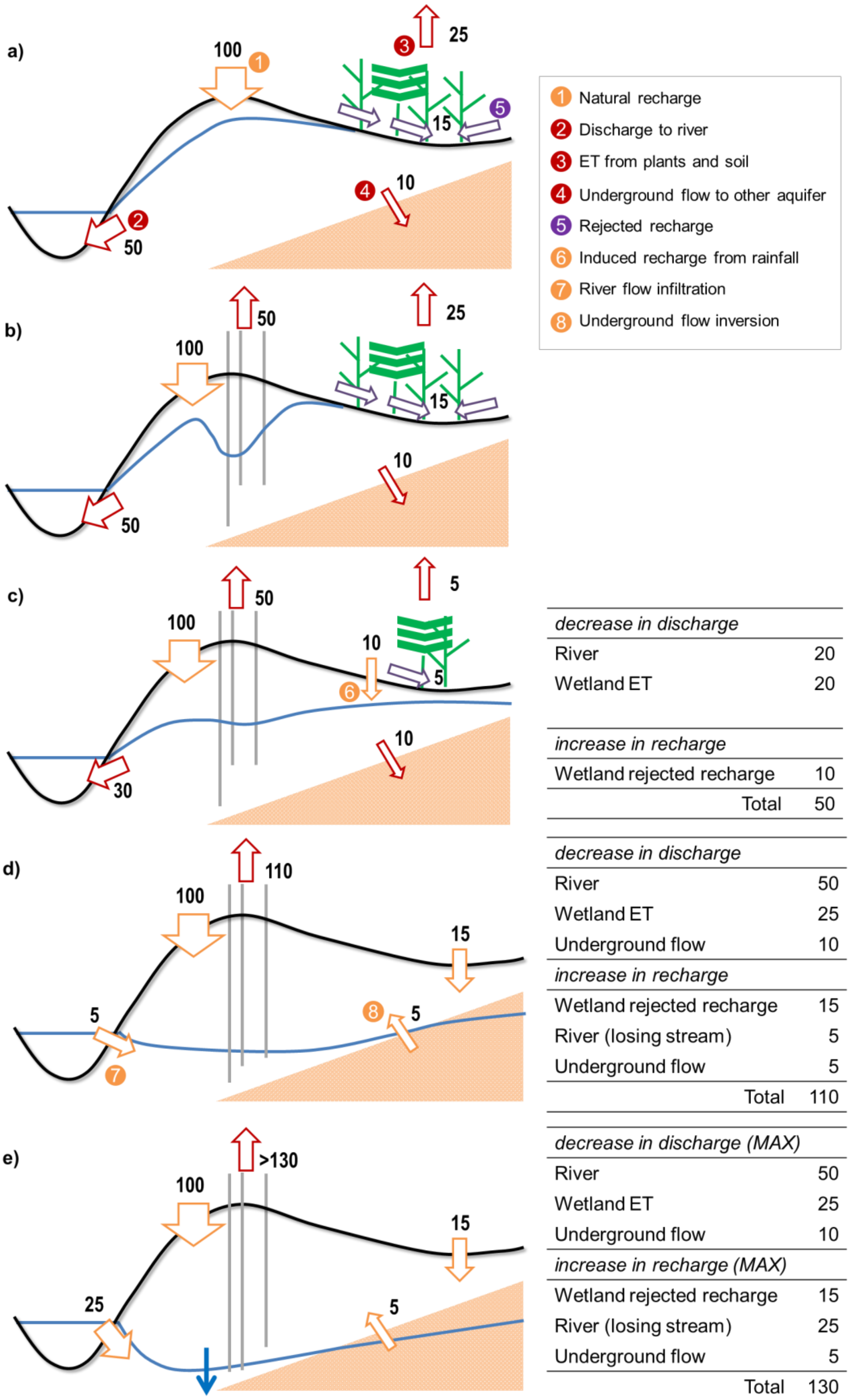
$$P(t) = r_{max} + d_{max} + dV(t)/dt \quad (2.10)$$

or $$P(t) = R_{max} + dV(t)/dt, \quad (2.11)$$

since $R_{max} = r_{max} + d_{max} = r_{max} + R_0$

with R_{max}, the total recharge or inflows to the aquifer (natural and induced by pumping).

[9] The stock depleted during the transient stage should also be considered. However, it has been assumed in this section that the steady state is reached quite 'rapidly', in order to simplify, which implies the depletion is limited. In a real case, it should be assessed with more details.



decrease in discharge	
River	20
Wetland ET	20
increase in recharge	
Wetland rejected recharge	10
Total	50

decrease in discharge	
River	50
Wetland ET	25
Underground flow	10
increase in recharge	
Wetland rejected recharge	15
River (losing stream)	5
Underground flow	5
Total	110

decrease in discharge (MAX)	
River	50
Wetland ET	25
Underground flow	10
increase in recharge (MAX)	
Wetland rejected recharge	15
River (losing stream)	25
Underground flow	5
Total	130

Figure 2.6. Transient stage and the new water budget at the steady state for various pumping rates (P) and corresponding value of capture from the different origins.
(a) Natural state / (b) Transient stage (P=50) / (c) Steady state with limited flow impacts (P=50) / (d) Steady state with more intensive flow impacts (P=110) / (e) Pumping is too high to be compensated by capture, which has reached its maximum value (P>130) / In the situations c and d, the amount of pumping equals the amount of capture.

Equations 2.9, 2.10 and 2.11 mean that pumping captures all the water that infiltrates into the aquifer (initial and induced recharge) and consumes storage, i.e. *stock impacts* take place. No more outflows go to surface water and evapotranspiration and the rejected recharge now percolates entirely, which can potentially have large environmental consequences. In other words, stock impacts are cumulative to the maximum flow impacts (Figure 2.7)[10].

The issues associated with the consumption of groundwater stock, e.g. land subsidence or rising pumping costs and depletion of reserves that will affect future use and other users, usually constitute a real problem to take into account only when a substantial consumption of the stock or drop of the water table has occurred.

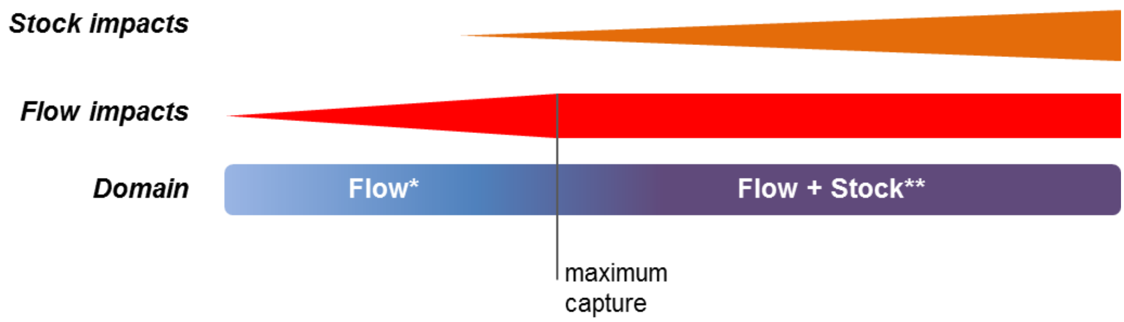


Figure 2.7. Domains of the 'flow impacts' and 'stock impacts'.
*Domain of Equation 2.8 (pumping equals capture) / **Domain of Equation 2.9 (steady state) or 2.10 (pumping equals the sum of maximum capture and stock depletion).

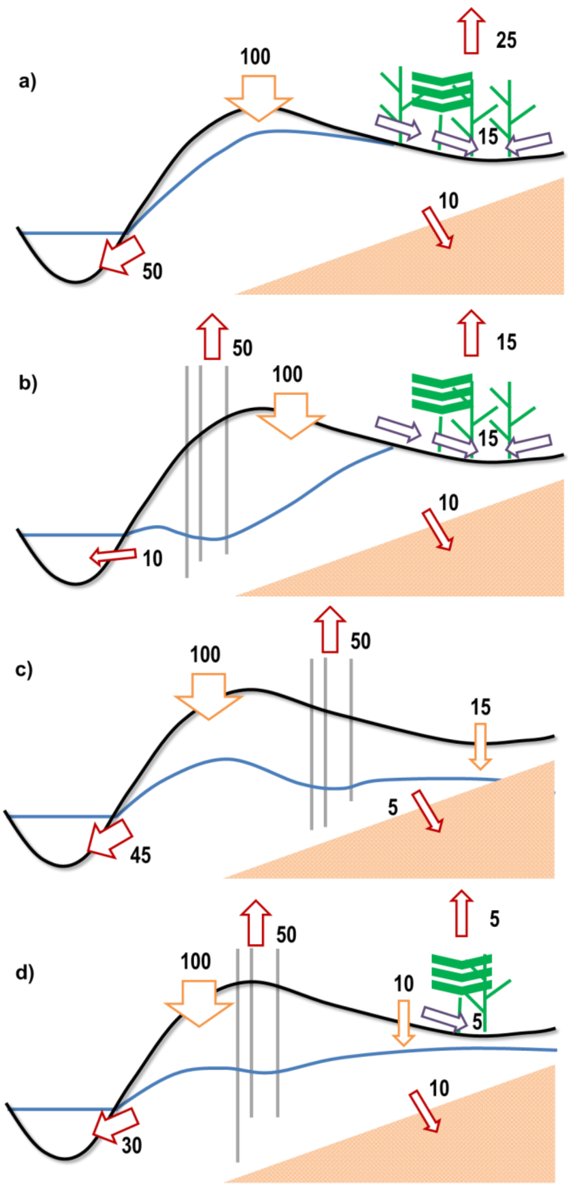
II.3.4 Mobilization of different sources of capture for a same pumping rate

Equations 2.6 to 2.11 show the general dynamics of the aquifer development. Depending on the hydrogeological conditions, there are potentially various sources of capture (N areas of increase in recharge and M areas of decrease in discharge). Thus, there is an infinity of steady-state situations that satisfy the budget of Equation 2.8, since the constant value of each of the sources of capture ($d_{s,i}$ and $r_{s,j}$) can be mobilized between 0 and its maximum value ($d_{max,i}$ and $r_{max,j}$):

$$P_s = \sum_{i=1}^{N} d_{s,i} + \sum_{j=1}^{M} r_{s,j} \quad \text{with } d_{s,i} \in \left[0; d_{max,i}\right] \text{ and } r_{s,j} \in \left[0; r_{max,j}\right] \qquad (2.12)$$

The share of each source of capture depends on the location of the wells and their respective pumping rate (Figure 2.8).

[10] The occurrence of the flow impacts before stock impacts is relevant in many situations. Yet it is usually not identified. Even if both types of impacts are often considered, they are not hierarchized.



decrease in discharge	
River	40
Wetland ET	10
Total	50

decrease in discharge	
River	5
Wetland ET	25
Underground flow	5
increase in recharge	
Wetland rejected recharge	15
Total	50

decrease in discharge	
River	20
Wetland ET	20
increase in recharge	
Wetland rejected recharge	10
Total	50

Figure 2.8. Contribution of the different sources of capture according to the well location. (a) Natural situation / (b) The river constitutes the main source of capture / (c) The wetland and the underground flows are the main source of capture / (d) Intermediary situation.

For instance, if pumping takes place near a river, it will constitute the source of capture that will be firstly and mainly mobilized. Only a detailed study of the behaviour of the aquifer and the process of mobilization of the sources of capture can tell us what will be the final sources of capture and their amounts. The value of the pumping rate cannot *ex ante* tell us the intensity of the impacts by itself. Other parameters must be specified, such as the well location and the pumping rate for each of the wells. It implies also that the well location can be changed to modulate the mobilisation of the sources of capture (Bredehoeft, 1982; Seward et al., 2007).

It is noteworthy that, when the maximum capture is mobilized (Equation 2.9), the final state is independent of the aquifer development path and location of the wells because all the sources of capture are fully mobilized. The time to reach this situation can vary according to the location of wells. We have then:

$$d_{s,i} = d_{max,i} \text{ and } r_{s,j} = r_{max,j} \text{ for every i and j.} \quad (2.13)$$

$$P_s = \sum_{i=1}^{N} d_{max,i} + \sum_{j=1}^{M} r_{max,j} \quad (2.14)$$

II.4 Replenishment of the aquifer: introducing future capture

In many analyses on groundwater management strategies, it is considered that the current pumping will remain the same for many years. However, in addition to the effect of lowering the groundwater level to a point where pumping is too expensive, there are many factors that can contribute to a decrease in pumping. For instance, in agriculture, the profitability of a crop might change or other sources of water could be obtained. The price of energy may also rise in the future, with direct consequences for the pumping cost. These are just some examples.

When pumping stops or decreases, the groundwater level logically rises. However, this is not without consequences for the associated surface flows and groundwater dependant ecosystems. Capture takes place until the initial level is attained again[11]. Particularly it will remain at its maximum value until the level attains the point where surface water flows are generated, through direct outflows from the aquifer or rejected recharge. Finally the total amount of the capture that will be obtained once pumping has stopped corresponds to the total amount of stock depletion that has been attained. The affection of surface water flows and groundwater dependant ecosystems will potentially last many years after pumping stops.

The stock depletion can be viewed therefore as a *future capture* (c_f) and Equation 2.6 can be reformulated as:

$$P(t) = c(t) + c_f(t) \quad (2.15)$$

i.e. groundwater pumping is the sum of the current and future capture.

Since an initial consumption of the stock is necessary for the mobilization of capture, these delayed temporal effects happen for every aquifer development and are the opposite effects of the delayed impacts of groundwater pumping at the beginning (the transient stage in Figure 2.6).

An implication of highlighting the future capture is that in many situations, the issue of stock consumption relative to the intergenerational equity of groundwater use, which is a frequent debate regarding groundwater management, could be relativized. Contrary to oil or mineral resources, the recovery of the storage can be relatively quick (i.e. on a scale of years or decades

[11] In case it is possible, e.g. compaction of the desaturated layers may prevent the rise of the level again.

rather than million years). The future capture and the associated impacts should, however, be fully contemplated.

II.5 When the capture takes time to be mobilized

For many situations, it can be considered that the steady state is attained with a limited drop of the water table, i.e. relatively quickly (a few weeks or years, see Figure 2.9), which has been the main assumption in Section 3 of this chapter. However, in some cases, a continuous and potentially substantial drop in the groundwater level (i.e. stock consumption) can occur during the transient stage (Figure 2.6.b). Depending on the aquifer characteristics and the local setting (e.g. the distance to the source of capture, the confined or unconfined nature of the aquifer), the transient stage can last many years or decades and an important stock depletion can take place, with, among other issues, the associated problems of rising pumping costs and land subsidence. Here, these stock impacts are relevant before the impacts in terms of capture.

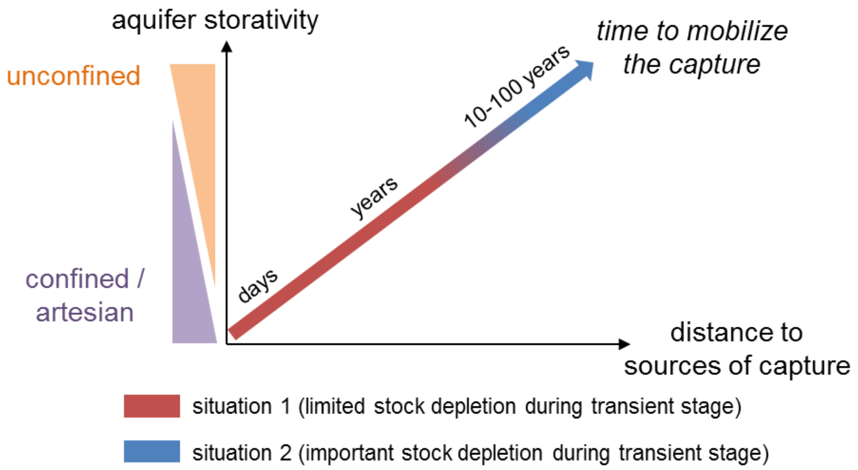


Figure 2.9. Two situations relative to the time to mobilize the capture.
Storativity depends on the geological nature of the aquifer and its confined or unconfined situation (as illustrated by the triangles on the figure). Confined aquifers usually have a lower storativity than unconfined ones and are more prone to being in a situation where the time to reach the steady state is small.

This configuration would correspond to another, more complex, situation for groundwater management and sustainability (situation 2 on Figure 2.9), as compared to the previous description that emphasizes the mobilization of capture and the associated impacts (situation 1). In this second case, the stock depletion during the transient stage can be huge and implies a future capture in the mid- to long-run (few decades), which should be taken into account.

II.6 Synthesis:

- Aquifers are commonly presented as stocks of groundwater, which implies the focus is on water that infiltrates (recharge or aquifer inflows), as it will reconstitute the stock.
- The approach that has been introduced in this chapter focuses on the role of groundwater in the watershed to sustain flows towards surface water and dependent ecosystems. The change in groundwater heads triggered by pumping modifies the water balance of the aquifer, with less water leaving the aquifer (decreased outflows) and/or more water entering the aquifer (increased inflows), i.e. flow impacts. The sum of these factors is termed ‘capture’. The mobilization of the capture is made possible thanks to a drop in the water table, i.e. a stock depletion, which is only temporary if enough sources of capture are available. At the steady state, pumping equals capture.
- If pumping cannot be balanced by capture, a continuous stock depletion takes place. In many situations, intense stock depletion and the associated issues are relevant when flow impacts are already strong. Indeed a continuous depletion of the stock requires the mobilization of the maximum capture. Thus, there is a hierarchy of impacts and two domains can be distinguished: a domain where only flow impacts are relevant and a domain where flow and stock impacts should be managed.
- In cases where the capture can take decades to centuries to be mobilized, stock impacts are first relevant. However, flow impacts will be also generated in the future. This is therefore a complex situation to assess.
- Depending on the location of pumping, the contribution of the different sources of capture and the temporality of their mobilization changes.
- Two characteristic times should be identified: when the capture starts being mobilized and when the steady state is attained (capture fully mobilized).
- Capture (i.e. flow impacts) is not reduced until levels are recuperated in the place of capture, which happen time after pumping has stopped or has been reduced. This is a symmetric effect to the initial stock consumption in the transient stage. The term ‘future capture’ is proposed to take into account this effect. Future capture replenishes the aquifer and current pumping can be expressed as the sum of the current capture and future capture.

Chapter III

GROUNDWATER ALLOCATION

List of symbols of this chapter

$d_{cons,i}$	agreed decrease in discharge rate to limit capture in the area of discharge *i* (m^3/s)
$d_{max,i}$	maximum (physical) decrease in discharge rate in the area of discharge *i* (m^3/s)
$d_{s,i}$	decrease in discharge rate in the area of discharge *i* at steady state (m^3/s)
$f_{cons,i}$	agreed value for the rate of decrease in flows for a source of capture (m^3/s)
$mf_{cons,i}$	minimum flow rate that should be respected for the area of discharge *i* (m^3/s)
P_s	pumping rate at steady state (m^3/s)
R_0	natural rate of inflows to the aquifer (m^3/s)
$r_{cons,j}$	agreed increase in recharge rate to limit capture in the area of discharge *j* (m^3/s)
$r_{max,j}$	maximum (physical) increase in recharge rate in the area of recharge *j* (m^3/s)
$r_{s,j}$	decrease in discharge rate in the area of discharge *j* at steady state (m^3/s)

III. GROUNDWATER ALLOCATION

III.1 Introduction

The aim of this chapter is to establish a framework to guide defining of the amount of groundwater that can be pumped from an aquifer. The basic approach contemplates the trade-offs associated with groundwater use in terms of mobilization of the sources of capture and stock consumption (or future capture) as introduced in Chapter 2. While groundwater pumping is motivated by the direct value for aquifer users and the local economy, decisions on groundwater use should also include considerations on the acceptability of the different impacts.

Balancing costs and benefits from the use of natural resources, relative to environmental, social and economic values is directly linked to the concept of sustainability, which has been extensively debated in relation to groundwater resources (Sophocleous, 1998; Alley et al., 1999; among others). This chapter begins with some considerations on the sustainability concept in order to clarify the specific approach of this thesis in terms of highlighting trade-offs.

The proposed framework follows; first, dealing with the acceptability of flow alteration (capture) and, second, with the consumption of the groundwater stock. The formulation of this framework finally allows clarifying and discussing a series of issues, such as the different interpretations of the concepts of ‘safe yield’ and ‘sustainability’ applied to groundwater or terms such as ‘renewable resources’ or ‘groundwater mining’ that influence the perception of allocation issues or ‘sustainability’. The relevance of the definition of the groundwater as a Common-Pool Resource is also assessed.

This chapter primarily shows that the identification of the implications of pumping in terms of capture and stock consumption is essential in the process of decision-making. This allows establishing commonly agreed caps that take into account the contribution of aquifer outflows to water resource availability at basin scale for both human uses and ecosystems now and in the future. It is also shown that the concepts of ‘safe yield’ and ‘sustainable groundwater use’ have multiple meanings and are often associated to oversimplified views of groundwater dynamics or relate to an overall objective in terms of combining environmental, social and economic implications that is too general to generate practical criteria needed for decision-making.

III.2 Balancing values: sustainability and trade-offs

III.2.1 Common definitions of sustainability

Two common definition of sustainable development are usually found. The first definition is from the World Commission on Environment and Development (1987) (also known as the Bruntland report), which made the concept widely known:

> "Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs."

The other main definition of sustainable development considers the integration of three pillars – the economy (or efficient use), society (or equitable use), and the environment in the 'sustainable development triangle' (see Figure 3.1) – with the objective to combine these three aspects in decisions over development options (Munasinghe, 2003).

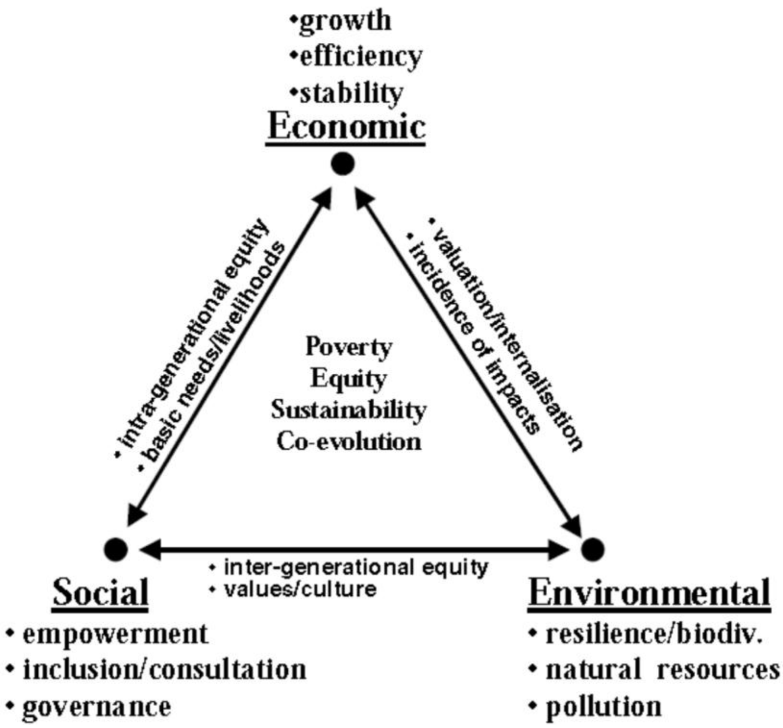


Figure 3.1. The 'sustainable development triangle'. Source: Munasinghe, 2003.

Both definitions imply that the environmental dimension should be integrated into decision-making. The provision of renewable resources and other ecosystems services in the long run requires a preserved environment. The emergence of this environmental awareness through the second half of the twentieth century comes from the growing evidences of environmental deterioration and depletion of resources. Additionally, sustainable development has a social component, with the view that development should benefit everyone.

III.2.2 Sustainability as a 'nirvana concept'

In the previous definitions, sustainability appears as a very general principle related to the final goal of a decision-making process. Yet it is potentially hard to qualify a situation as sustainable or unsustainable, as it relates to the complex balance of different values and criteria, which are not really mutually compatible (Barraqué, 2001). It constitutes one of the reasons why Molle (2008) introduces sustainability as a 'nirvana concept':

"Nirvana concepts are concepts that embody an ideal image of what the world should tend to (...). Although, just as with nirvana, the likelihood that we may reach them is admittedly low, the mere possibility of achieving them and the sense of 'progress' attached to any shift in their direction suffice to make them an attractive and useful focal point.

Nirvana concepts usually take the form of a 'photo-negative' of the real world. For example, as the social and environmental costs of conventional industrial development became apparent, the concept of sustainable development proposed a vision whereby contradictions would be dissolved, negative impacts internalized, and antagonisms reconciled. Likewise, the concept of good governance emerged as a model in which inefficient, corrupt, biased and discriminatory governments would – "as a result of" or "through" growing transparency and power-sharing – become accountable to their populations and act for the common good. Participation or empowerment, at some level of generalization, also appear as desirable counterpoints to exploitation and disfranchisement. All these words are "warmly persuasive" (Williams, 1976), nice-sounding, sanitized, and endowed with "almost unimpeachable moral authority" (Cornwall and Brock, 2005), at least in the spheres of development professionals."

Sustainability as a 'nirvana concept' relates also to the formulation of a state where the three dimensions (efficiency, equity and environment) could be balanced to reach a situation where decisions on the environment would be universally accepted as the right solution. This approach disregards the fact that decision-making on the environment and use of natural resources consists in balancing trade-offs, which inevitably benefits some and imposes costs on others. Win-win processes are possible but are often more difficult than anticipated. The message of peaceful integration and harmony conveyed by the sustainability concept, which hides the necessary costs and concessions, is also recognized by Van der Gun & Lipponen (2010):

"(…) the designation 'sustainable' should not be interpreted too rigorously (…) [since one] has to sacrifice almost always something in exchange."

III.2.3 Approach in this thesis: trade-offs and acceptability

Faced with the two main problems of the sustainability concept – the lack of recognition of the complexity of decision-making and the consideration of an ideal and inclusive state – the general framework for decision-making on groundwater in this thesis consists in highlighting the trade-offs involved in the allocation of groundwater, based on the capture concept. This allows identifying the real impacts of groundwater pumping and the concrete decisions to be taken in terms of acceptability of these impacts. Alley et al. (1999) have a similar approach relative to environmental impacts:

"The tradeoff between water for consumption and the effects of withdrawals on the environment often become the driving force in determining a good management scheme.(…) the public should determine the tradeoff between groundwater use and changes to the environment and set a threshold at which the level of change becomes undesirable."

This approach could be considered as contributing to the debate on sustainability, if one chooses to recognize the value of this terminology, as in fact done by Alley et al. (1999).

It is also noteworthy that, in many cases, when more sustainable practices are demanded or an unsustainable situation denounced, it is a call for better integration of aspects such as the environmental side or future generations. Sometimes, sustainability is simply a synonym with the 'good' or 'virtuous' use of natural resources. Mentioning sustainability may have more impact since it goes against a supposedly unquestionable general objective for society: it has a discursive power.

Referring to sustainability, however, makes it possible to be criticized on the basis of the concept itself, because other dimensions may not be taken into account. Typically when the use of a resource, such as water, generates environmental impacts, it can be argued that the situation cannot be qualified as unsustainable as long as the economic dimension is not considered. Thus, it would be clearer to make a direct reference to the concrete environmental value that is infringed, e.g. speaking of 'impact on dependent ecosystems' or 'infringement of the WFD' instead of 'unsustainable use'.

III.3 A framework for the allocation of groundwater

III.3.1 Decisions within the flow domain

In the flow domain, defined as a situation where pumping is entirely compensated by capture at steady state (see Chapter 2, especially Figure 2.7), the main consequence of groundwater pumping is an alteration of flows to surface water and groundwater dependent ecosystems. Decisions on the amount of groundwater that can be used should be based on the acceptable level of capture. It requires characterizing the different sources of capture and their mobilization, i.e. *where, when, how long and how much* outflows decrease and inflows increase. Capture affects water availability in the river basin. Thus, caps on capture should be agreed in relation to various elements:

- Water management in the whole river basin, i.e. integrating all water resource users while conserving environmental flows, as river flows are potentially captured by groundwater pumping ('downstream impacts' in Table 3.1).
- Conservation of the ecosystems that depend directly from the maintenance of the water table ('direct ecological impacts on dependent ecosystems' in Table 3.1).
- Conservation of groundwater flows to connected aquifers since these flows may be essential for users of the connected aquifer and indirectly imply a capture.
- Prevention of seawater intrusion since capture reduces freshwater flows to the sea.

Table 3.1. Cumulative impacts associated to groundwater use.

Impact		Scale / Who is concerned?	Example of data to guide decisions
Flow impacts	Downstream impacts	All water users and stakeholders in the river basin, integrating environmental flows	Availability of water resources at river basin scale
	Direct impacts on dependant ecosystems	Aquifer and associated ecosystems users / Ecological value[a]	Resilience of ecosystems to a drop in groundwater level
	On connected aquifers	Direct users of the connected aquifers and all users impacted by the consumption of groundwater from the connected aquifers	Capture of flows to the connected aquifers
	Seawater intrusion	Aquifer users	Maximum pumping to prevent marine intrusion
Stock impacts		Current and future aquifer users	Maximum inflows to the aquifer
Future capture		All the above mentioned users	Time to recover the stock = Time with operant capture

All these impacts take place until the aquifer is replenished to its natural initial state.
[a] The 'ecological value' implies that the conservation of a dependent ecosystem can be relevant at many scales, such as the existence value attached by people living far away, or for the contribution to ecosystem services, such as carbon sequestration, in addition to potential local uses in terms of materials source. Integrating all these values and scales in the decision-making process on water resources remains a challenge.

The assessment of available resources is consequently the result of a complex process, as it requires the technical assessment of the different types of capture and also the involvement of many different stakeholders depending on the issues relevant for the case analysed (Table 3.1). For instance, groundwater rights allocation should be coordinated with surface water rights allocation. In this case, capture is acceptable only if the captured flows are not previously allocated to users and environmental flows downstream in the river basin. In the particular case of a 'closed river basin', i.e. where no more water is available without affecting the environment, water should be obtained from reallocation of other users' rights. Then there is initially no water 'available' from the aquifer. This has been recognized in some Western States in the United States, where groundwater pumping is restricted for its effect on surface water flows (Alley et al., 2002). However, it is an exception, motivated by the action of senior surface water rights holders who saw their right jeopardized by river flow capture.

As regards with the direct ecological impacts, the challenge is to assess the acceptable limit for ecosystem degradation based on technical data. In case of a clear threshold (tipping point), the limit could be established on this basis. However, if the alteration is more progressive, setting the acceptable limit is harder since the loss of ecological value is changing progressively with the level of resource use. The process of defining the acceptable level of capture can also be challenged by the uncertainties in the technical data (as illustrated in the case studies of this thesis: Chapters 7, 8 and 9).

For coastal aquifers, we shall notice that, in absence of an ecologically valuable discharge area[12], the specific impact is on the current (and potential future) users of the aquifer, since there are no downstream users (Table 3.1).

It is noteworthy that knowledge on the system is essential to make decisions and scientists have to provide a series of data to support decision-making (Table 3.1). This situation contrasts with the traditional criteria for groundwater sustainability, where the role of hydrogeologists is limited to quantify the recharge.

Thus, the process of decision on the acceptability of capture results in the definition of constraints for each source of capture:

$$\mathrm{d}_{\mathrm{s,i}} \leq \mathrm{d}_{\mathrm{cons,i}} \quad \text{and} \quad \mathrm{r}_{\mathrm{s,j}} \leq \mathrm{r}_{\mathrm{cons,j}} \tag{3.1}$$

with $d_{cons,i}$ and $r_{cons,i}$, the agreed decrease in discharge rate and increase in recharge rate to limit the impacts from pumping on surface water and groundwater dependent ecosystems.

It has been shown in Section 3.4 of Chapter 2 on the mobilization of the different sources of capture that, in case of multiple sources of capture, the location of wells influences the amount mobilized from each source. Hence the possibility to optimize the localization of wells and aquifer development to respect the conditions formulated in Equation 3.1. However, this is difficult to achieve in practice since there are constraints on the location of wells (or these already exist). Thus, as stated by Maimone (2004), the debate on the amount of pumping from an aquifer includes also the issue of the place of pumping:

> "(...) understanding where withdrawals can best be made and identifying areas where impacts are to be minimized (or maximized) are critical considerations in defining sustainable yield."

[12] The value of freshwater discharge flows into the sea is a question that is hardly contemplated. Yet the contact between freshwater and sea water and the difference of temperature of these flows within the groundwater discharge areas can create specific valuable ecosystems (Alley et al., 2002; Van der Gun & Lipponen, 2010).

Custodio (2002), in the same line, affirms that, in many situations, what is perceived as over-exploitation may be only the result of local problems (e.g. location of pumping).

When there is only one source of capture, even if this source can be object of both decreased outflows and increased inflows (e.g. a river stream section that can be turned from losing to gaining stream), the condition of maintaining a certain amount of flow (Equation 3.1) allows defining the maximum acceptable level of pumping:

$$d_s + r_s \leq d_{cons} + r_{cons} \qquad \text{hence: } P_s \leq f_{cons,i} \qquad (3.2)$$

with $f_{cons,i}$, the maximum value for the decrease in flows for the source of capture[13].

The variations of the groundwater table under natural conditions are also worth considering, particularly when the characteristics of the aquifer imply there is a small residence time for groundwater in the aquifer. For instance, under arid or semi-arid climates characterized by high rainfall variability, with recurrent periods of droughts over several years, the natural discharge to surface water and groundwater dependent ecosystems can be highly reduced for several years, compared to the mean value. Consequently, it should be the same for capture. The pumping restrictions cannot be based on average outflows. This point is particularly relevant for the Western Mancha case study as will be illustrated in Chapter 7.

III.3.2 Decisions within the stock domain

- Access to the stock domain and interest of traditional groundwater visions

In the stock domain, all sources of capture are mobilized at their maximum value (see Section 3.3 of Chapter 2, especially Figure 2.7). It is acceptable if no constraint on capture, as presented in Equation 3.1, is stricter than the maximum capture from each source, that is to say:

$$d_{max,i} \leq d_{cons,i} \qquad \text{and} \qquad r_{max,j} \leq r_{cons,j} \qquad (3.3)$$

As the full capture is acceptable, the main issues at stake are the possibility to consume the aquifer stock and the repartition of the consumption over time, taking into account the adverse consequences, such as rising pumping costs, land subsidence or water quality degradation.

Management and sustainability issues of groundwater use are traditionally introduced under this approach. They overlook flow impacts and contemplate aquifers as a stock that is replenished thanks to recharge constituting the available resource. This view is found in basic groundwater considerations by non-specialists but also in more technical and detailed models as Alley et al. (2002) have also noticed. Even if this vision is simplistic and misleading as a general approach,

[13] Regarding the quality of pumped water, it is relevant to consider the origin of the capture (water from the river or entirely decreased outflow from the aquifer).

the different propositions and conclusions can be useful for the stock domain assessment (i.e. when Equation 3.3. conditions are satisfied).

It should be remembered that, in case of a delayed mobilization of capture (decades to centuries), stock impacts are relevant before flow impacts and they should be managed. However, future capture, which will irremediably occur, may not be acceptable, which means a different situation compared to the generic situation considered in this section, where access to the stock is conditioned by the acceptability of flow impacts.

- The approach by the 'economic modelling literature'

The approach by the 'economic modelling literature' on groundwater (Gisser & Sánchez, 1980; Howitt & Nuckton, 1981; Provencher & Burt, 1993, among others) usually disregards flow impacts. The example of the well-known work by Gisser & Sánchez (1980) is particularly illustrative. The controversial conclusion of this work is that, under a list of assumptions, no significant improvement (in terms of 'value' generated from the groundwater stock) results from the planned management of an aquifer, compared to the open-access situation (no regulation). However, the model considered in this work does not contemplate the outflows from the aquifer and, consequently, their value for the environment and other users (Figure 3.2). Therefore, it would be applicable only in the situation where flow impacts are irrelevant, which is a very specific situation. Esteban & Albiac (2011) introduced the value of a groundwater ecosystem to show the main conclusion of Gisser & Sánchez (1980) does not apply in this case, which is *a priori* the general situation[14].

In fact, the models found in the literature are based on defining an 'efficient' or 'optimal' groundwater withdrawal path, i.e. which generates the highest value for society[15]. This relies on a formulation of a specific problem or view on the issue of groundwater management. It consists in quantifying over time, with a changing pumping rate, the depletion of the stock based on the value it generates, as compared to the value lost because of higher pumping costs in the future. A discount rate is introduced, i.e. the value of the money earned now is considered higher than the value of the money earned in the future.

[14] This result has been also intensely discussed relatively to the reality of the some other basic assumptions (Gardner et al., 1997; Koundouri, 2004; Brozović et al., 2010, among others).

[15] This objective is virtually not reached without integrating the value of the outflows for the environment and downstream users, yet assigning a monetary value to the environment is controversial. Establishing the 'optimal' or 'efficient' withdrawals path is therefore much more evident when the environmental value is not integrated, since it only concerns the allocation of a stock over time for human use. In line with this point, even if the work of Esteban & Albiac (2011) shows that the inclusion of the value of the environment is essential, the method and valuation used in their work can be questioned since the relation between pumping and the impact on wetlands is simplified, plus the monetary valuation is based on the method of contingent valuation, which implies a series of assumptions, and is not universally accepted (see Spash, 2000).

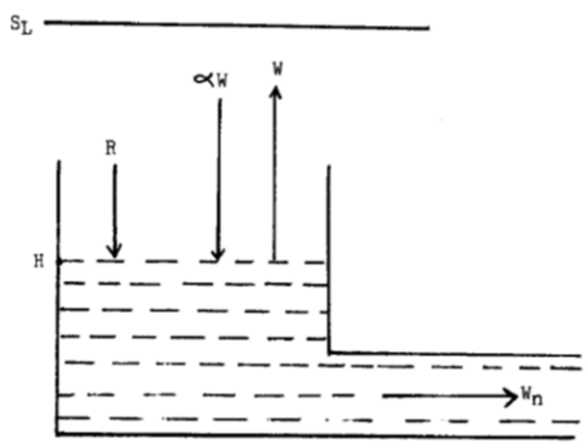


Fig. 1. A model of an aquifer.

The differential equation which describes the water table as a function of time is obtained by equating 'rate in' minus 'rate out' with the impact on the water table, as displayed in Figure 1, namely

$$AS \cdot \dot{H} = R + (\alpha - 1)W \qquad (6)$$

where R is natural recharge, α is return flow coefficient, and AS is area of the aquifer times storativity.

Figure 3.2. Extracts of the paper by Gisser & Sánchez (1980) showing that the factor accounting for the outflows in the water budget is disregarded. Source: Gisser & Sánchez (1980).

The factor W_n (natural outflows from the aquifer) is omitted in the rate balance. It is all the more striking since W_n appears in the model of the aquifer. Thus, the equation presented appears as a strong version of the 'Water budget myth' (Bredehoeft et al., 1982).

The problem formulation implies that:

- the value of groundwater pumped now (through its use for irrigation, industrial production, etc.) is higher than if it were pumped in the long run;
- the influence of future higher pumping costs is reduced since the economic losses generated have comparatively less value[16].

If the environmental value of the outflows would be introduced in this approach (i.e. necessarily through a monetary valuation), the discount rate implies also that the value generated by the outflows is potentially negligible after a certain period of time (Esteban & Albiac, 2011).

Howitt & Nuckton (1981) in their paper 'Is overdrafting groundwater always bad?' make a comparison of this problem with a bank account[17]:

[16] Without considering the discount rate, the mentioned problem could not be formulated because the value for the current use and the future use would be the same, which means there would not be any reason to allocate groundwater stock according to time. It is because of this discount rate that there is a possibility to optimize the consumption of the stock. The consideration of a constant production function through time, while the incentives for groundwater pumping are usually highly variable, is also problematic.

> "Continuing our banking analogy, the groundwater stock can be compared with capital that can be either invested in business (growing crops) or saved to draw interest. "Interest" accrued by the decision not to overdraft is collected in the future through savings in pumping costs. Overdrafting, of course, lowers the water table so that, as time goes on, pumps must draw more deeply at increased cost."

The irremediable result of this approach is that, after depleting the stock by pumping more than the inflows during a period of time, withdrawals have to be reduced to the value of recharge:

> "Again, using the banking analogy, the "steady state" of a groundwater source is the point at which deposits (recharge) equal withdrawals. A hydrologic system is in a steady state if net withdrawals equal the average rate of natural recharge. This state can be attained at various groundwater levels; the more overdrafting, the deeper the level. (…) Sooner or later all groundwater aquifers must reach steady state, although, it is hoped, not when empty. The management decision to be made for each aquifer is when and at what depth to stop overdrafting."

This citation is another evidence of the consideration of withdrawals that equal 'the average rate of natural recharge' as an objective for the management of aquifers.

Finally, three main obstacles remain to agree with the problem formulated, and thus the results, proposed by this economic approach to groundwater management, in addition to the lack of consideration of flow impacts:

- the problem formulation, i.e. significance of a discount rate and its implication in terms of sustainability and equity among generations; this approach is mainly linked to neo-classical economics, and has been challenged by other paradigms (e.g. ecological economics, see Daly & Farley, 2003);
- the acceptability of the many assumptions necessary to obtain the model and a solution to this problem of optimization, e.g. the simplification of the dynamics of groundwater in an aquifer or a constant production function, when in fact the value of the use of groundwater changes over time;
- the proper concept of an optimal path of withdrawals, which can be unenforceable in reality because it contemplates only the financial returns or economic efficiency as the sole criteria to allocate groundwater, without considering criteria like equity and legitimacy or the complex legal system associated to groundwater [18].

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[17] A bank account is a common metaphor associated with the view of groundwater as a stock, to illustrate that we should not pump beyond recharge, similar to how we should not take more money than our incomes. Keeping with this metaphor, what is forgotten is that some money has to be reserved each month for paying the bills and the taxes, which implies the whole income cannot be used. In the same way, the conservation of environmental and downstream flows limits the use of inflows to an aquifer.

[18] For instance, it means that what Gisser & Sánchez (1980) consider as the 'efficient' allocation that would be ideally implemented by the Public Authorities – and that is compared with the open-access to finally obtain that the regulation has no benefits – is the approach presented in this section. This is a very specific interpretation of the role of the Public Authorities. Furthermore, efficiency and optimality are value-laden and are not an absolute criterion.

- Other debates on the allocation of aquifer stocks

Concerns with the rights of the current generation to use the stock of groundwater for its development (which will also benefit future generations) is an issue contemplated beyond the problem described in the previous section. The view that development based on intensive groundwater use would allow for the local economy to diversify and thus would reduce the need of using groundwater has been proposed (Custodio, 2002; Llamas & Custodio, 2002). Another question concerns the right of the current generation to use groundwater, which should not necessarily be reserved for future generations (Ravenscroft et al., 2013). In the specific case of fossil groundwater[19], Llamas (2004) asks:

> "Fossil groundwater has no intrinsic value if left in the ground except as a potential resource for future generations, but are such future generations going to need it more than present ones?"

In relation to the consumption of 'non-renewable' groundwater resources, Foster & Loucks (2006), in a guidebook prepared for the UNESCO, propose the following approach:

> "To meet the requirement of 'social sustainability' in respect of development of the nonrenewable groundwater resource, the following criteria need to be addressed:
> • it should lead to clear improvements in social well-being and livelihoods;
> • the balance between short-term socio-economic benefits and longer-term 'negative impacts' must be positive;
> • an 'exit strategy' should exist, with an answer to the question 'what comes after the aquifer becomes seriously depleted';
> • the issue of inter-generational equity has to be considered.
> (…) Concomitantly, it should be recognised that predicting the longer-term evolution of any given case of groundwater mining will be subject to significant uncertainty (as a result of incomplete hydrogeological understanding, innovations in water technology, changing global agriculture and food markets, accelerated climate change, etc), which places limits on conventional resource management. This dictates the need to incorporate more flexible and adaptive risk-based approaches, which need to find political acceptance"

III.3.3 Decisions relative to future capture

The introduction of future capture that results from stock consumption (see Section 4 of Chapter 2) makes it necessary to consider the potential agreed constraints on impacts from pumping also once pumping has stopped or has been reduced. This will happen virtually for any aquifer, with a high uncertainty on the time horizon. With the exception of cases where aquifer layers have compacted or land use and climate change imply a change in the inflows to the aquifer, the aquifer will replenish (i.e. reconstitute the stock for the use of the future generations). However, this will be at the expense of future capture generated by stock depletion. As the rate of future

[19] See Section 4.7 of this Chapter for a review of the meaning of the concept of 'fossil groundwater', which is often used inappropriately.

capture is equal or lower than the rate of steady state capture, the main issue is ensuring that it is acceptable over the whole period it is operational, which should take into account the climatic changes.

III.4 Some precisions and notes on misleading approaches

III.4.1 Interpretation of the sustainability concept in groundwater resources management

In the case of groundwater, the introduction of the term 'sustainable yield' or 'sustainable use' can be quite confusing. The very use of this term implies that it is related to the concept of sustainability as a nirvana concept, as introduced at the beginning of this chapter. However, the term 'sustainable yield' is often used in a more restricted approach, as it is commonly understood as a yield for which:

(*a*) the groundwater level or the volume of storage remains the same (dV(t)/dt=0);
(*b*) pumping is limited to the 'renewable resources' of the aquifer, i.e. the inflows;
(*c*) by extension, pumping is less than recharge.

The criterion *c* is particularly misleading because it does not take into account the dynamic nature of the recharge and the potential increase in recharge. This is also the case of the two other criteria if induced recharge is not taken into account in the 'renewable resource'. These criteria also do not address clearly the issue of transient stage, as introduced in Chapter 2.

Regarding the criterion *a*, at the steady state (attained because dV(t)/dt=0), pumping would be entirely obtained from capture. This can hardly be considered as sustainable (in a general sense) without further assessment, as the impacts of the capture on surface water and groundwater dependant ecosystems are not integrated. In the same way, both 'renewable resources' (criterion *b*) or recharge (criterion *c*) are essential for surface water and groundwater dependent ecosystems, since they flow out of the aquifer in natural state. In fact, these criteria are derived from a common view of groundwater as constituting the stock of aquifers replenished by recharge, where the role of aquifers outflow is disregarded. As noted by Alley et al. (2002):

> "Many unfamiliar with its dynamic nature view groundwater as a static reservoir. Even specialists may overlook its linkages across the biosphere and consider it an isolated part of the environment. Yet, (...) the dynamic aspects of groundwater flow systems, their recharge, and interactions with surface water and the land surface are numerous and extend over many different time scales."

The gap between the common consideration of a 'sustainable' use or yield and the sustainability concept has been recognized by Scanlon et al. (2012):

> "(...) sustainable pumpage may not equate to the much broader concept of sustainability, which includes minimizing adverse environmental impacts".

In the same way, Kalf & Woolley (2005), who indistinctively use the terms sustained and sustainable yield, recognize that:

> "It is unlikely of course that the basin maximum sustained yield would be desirable since it could mean complete loss of both residual outflow and stream flow (…)"
>
> "(…) a particular sustainable yield (or non-sustainable yield) (…) [should] be selected based or constrained by other criteria (i.e. water authority ground and surface water usage limits, community needs, legal factors, economic issues, ecological requirements, water quality, effect of subsidence)".

Thus, it is argued that the use of the term sustainable use or yield is different from referring to the concept of sustainability (Scanlon et al., 2012) or taking a decision based on a series of factors (Kalf & Woolley, 2005) (here in a 'nirvana concept' approach, see Table 3.2). Van der Gun & Lipponen (2010) explicitly identify these two meaning of the term 'sustainable':

> "(…) 'sustainable' merely indicates 'capable of being sustained', physically spoken. (…) a pumping rate that in principle may be continued forever without exhausting the aquifer, irrespective of whether such a pumping activity is considered desirable or not. At a higher-level perspective, sustainable groundwater development is not only defined by the physical capacity of an aquifer to yield water permanently, but also by a range of subjective conditions to be fulfilled or considerations to be taken into account."

The use of the same term to refer to different concept clearly generates confusion. For instance, a 'non-sustainable yield' could be acceptable (e.g. in Kalf & Woolley, 2005) or, inversely, a rate of pumping exerting a potentially excessive surface flows capture could be qualified as a sustainable pumping (e.g. in Scalon et al., 2012). Thus, it would be better to use a more descriptive or neutral term to refer to the situation where the level is constant, such as 'perennial yield'.

The debates and confusion generated by the two interpretations of the term 'sustainable yield', and the formulations in terms of 'groundwater use sustainability' in fact mirror the discussion of the term 'safe yield', which has been used according to the two approaches: a requirement to conserve the groundwater level or a more general concept that aims at reconciling the values to society ('nirvana concept') (Table 3.2). This is recognized by Alley & Leake (2004):

> "It should be clear the concept of sustainability in relation to ground water resources is far from new and is closely aligned with that of safe yield. The differences represent more of a transition, or (…) a journey, in our understanding of the dynamic nature of ground water and its linkages across the biosphere and to human activities."

Indeed, as reported by Zhou (2009), 'safe yield' definitions have progressively integrated economic, quality or water rights considerations and also the depletion of stream flow by induced infiltration and land subsidence, thus questioning an approach based only on the conservation of the aquifer stock.

Table 3.2. 'Sustainable yield' and 'safe yield' as specific criteria or general nirvana concepts and the visualization of the debates of the two concepts relative to groundwater.

	INTERPRETATION	
	Groundwater balance point of view[a]	*General societal objective / 'Nirvana concept'*
SUSTAINABLE YIELD	e.g.: approach contemplated by Kalf & Woolley (2005) "a pumping rate that in principle may be continued forever without exhausting the aquifer" (van der Gun & Lipponen, 2010)	e.g.: "Sustainable groundwater development at global and local scales is (...) the maintenance and protection of the groundwater resource to balance economic, environmental and human (social) requirements" (Hiscock et al., 2002) "Development and use of ground water in a manner that can be maintained for an indefinite time without causing unacceptable environmental, economic, or social consequences" (Alley et al., 1999)
SAFE YIELD	e.g.: "The limit to the quantity of water which can be withdrawn regularly and permanently without dangerous depletion of the storage reserve" (Lee, 1915).	e.g.: "The amount of water which can be withdrawn from [a groundwater basin] annually without producing an undesirable result" (Todd, 1959)

Main debates and logics found in the literature:

Debates on the definition of sustainable yield

Debates on the definition of safe yield

The safe yield concept questioned by the sustainability concept

[a] i.e. conservation of the level or stock and associated criteria (e.g. criteria a, b, c, see text). Some debates, such as the denunciation of the 'Water budget myth' (Bredehoeft et al., 1982, see Section 4.2 of this chapter), deal with the validity of the conditions to reach this equilibrium as they are mistakenly presented in criteria a, b and c for instance. However, these debates take place within the same approach in terms of conservation of aquifer stock or 'groundwater balance'.

The sustainability concept also serves sometimes as the basis to question 'safe yield' (Sophocleous, 1997; Devlin & Sophocleous, 2004, among others), 'safe yield' being understood as a limited criterion, that does not include all the aspects that the sustainable management of groundwater would require, i.e. 'safe yield' is interpreted exclusively in terms of groundwater balance. However, as shown on Table 3.2, both terms have been interpreted under the two approaches of groundwater balance.

III.4.2 The water budget myth and the necessity to know the value of recharge

Conceptually, contrary to the traditional approach, taking decisions on groundwater allocation does not require introducing the value of the total or natural recharge since it is the characterization of capture – i.e., *where, when, how much and how long* surface water flows and groundwater dependent ecosystems are affected – that constitutes the essential information.

After Bredehoeft et al. (1982), who introduced the 'Water budget myth', Devlin & Sophocleous (2004) explain what it consists in and give an explanation of why it is so common:

> "The appeal of the Water Budget Myth comes from the seemingly logical reasoning that if an aquifer is pumped more than it is recharged, it will one day run out of water. In a fashion consistent with the [Law of Conservation of Matter], this thinking can be expressed as a mass balance (erroneous in its oversimplification, as will be shown) of the form, R = P + D [R being the recharge, P the rate of pumping and D the discharge of the aquifer]
>
> The Water Budget Myth, fails in two respects: first, it assumes that recharge is independent of pumping, a condition not necessarily true in all cases, and second it assumes that recharge rates must be known to calculate sustainable pumping rates, a tenet that is clearly in error."[20]

Alley et al. (1999) also denounce this myth:

> "It is a myth because it is an oversimplification of the information that is needed to understand the effects of developing a ground-water system. (…), a predevelopment water budget by itself is of limited value in determining the amount of ground water that can be withdrawn on a sustained basis."

In this approach, it is mainly the common misconception of a 'static' budget that is denounced and the fact it is not necessary to know the recharge to characterize the 'sustainable' (perennial) yield. The common introduction of the water budget is not linked to the approach in terms of acceptability of capture and thresholds as discussed in this chapter (Section 3). This limits the significance of the concept of capture, which can serve to highlight the trade-offs linked to groundwater pumping, as we have shown earlier. Devlin & Sophocleous (2004) even affirm that sustainability (here, as a nirvana concept) is another problem for which it is necessary to know the recharge.

In fact, the usefulness of the value of recharge remains an issue of debate among hydrogeologists. The knowledge of natural initial recharge or dynamic recharge may help in certain contexts, for instance as an indirect way to know the maximum decrease in discharge. However, its determination cannot be considered as a necessary step and it can be misleading as a general approach. Recharge is also a basic requirement in models but its value may be difficult to obtain and be quite uncertain (Sophocleous & Devlin, 2004).

[20] 'Sustainable pumping rates' represents here a situation such as $dV(t)/dt=0$.

III.4.3 Subtraction of a reserved flow: only in particular situations

In order to take into account the necessary transmission of flows from an aquifer, it has been proposed to subtract the amount necessary for dependent surface water bodies and ecosystems to mean recharge to assess the availability of groundwater resources. This is particularly the case for the WFD as it will be discussed in Chapter 4. Considering that recharge remains unchanged, as assumed by this criterion – which is a first problem for its applicability in a general case –, this would mean that:

$$P_s \leq R_0 - \sum_{i=1}^{N} mf_{cons,i} \quad (3.4)$$

that is to say $$P_s \leq \sum_{i=1}^{N} d_{cons,i} \quad (3.5)$$

because $$R_0 - \sum_{i=1}^{N} mf_{cons,i} = R_0 - \left(D_0 - \sum_{i=1}^{N} d_{cons,i}\right) = \sum_{i=1}^{N} d_{cons,i} \quad (3.6)$$

with $mf_{cons,i}$, the minimum flow that should be respected for the area of discharge i.

P_s is the sum of the increase of recharge and decrease in discharge (see Chapter 2). However, here, only the later factor has to be included since increase in discharge is overlooked. From Equation 3.5:

$$\sum_{i=1}^{N} d_{s,i} \leq \sum_{i=1}^{N} d_{cons,i} \quad (3.7)$$

Realizing this condition is not equivalent to realizing the constraints presented in Equation 3.1:

$$d_{s,i} \leq d_{cons,i}, \text{ for each i}$$

If the maximum value of pumping is established on the basis of Equation 3.7, it is probable that one of the $d_{s,i}$ will not respect the constraint from Equation 3.1 if no other conditions are established (particularly the wells location). The criterion of Equation 3.4 is only valid in the specific situation where there is only one source of decrease in discharge and no increase in recharge:

$$P_s \leq R_0 - mf_{cons} = d_{cons} \quad (3.8)$$

since $P_s = d_s$, which implies $d_s \leq d_{cons}$

Then, Equations 3.1 and 3.4 / 3.7 are valid simultaneously.

This may be a common case, but this approach cannot be considered as valid as a general rule and may lead to undesirable outputs in some specific situations, where there are diverse sources of capture (see Chapter 4 on the application of the WFD).

III.4.4 Groundwater management under an arid climate: mobilization of the capture as a necessary evil?

Two domains of groundwater management have been distinguished. Access to the second domain is possible only in case the impacts of capture, which attains its maximum value, are acceptable (see Section 3.2 of this chapter). As noticed by Custodio (2002) and Kalf & Woolley (2005), among others, this condition would virtually prevent any use of groundwater in many situations and particularly under arid and semi-arid climates, where the availability of capture sources is limited. It implies that it is hard to remain within the flow impacts domain. Sophocleous & Devlin (2004) affirm also that in case of small recharge, the decision can be to mine the aquifer, i.e. consuming the stock at steady state.

The problem here is that, under this view, the mobilization of the maximum capture and its associated impacts is conceived as a necessary evil, which should be accepted in any case. Yet, the role of groundwater for groundwater dependent ecosystems and surface flows is particularly essential when surface water is scarce. In many parts of the world, this role has been lost due to the drop in groundwater levels (e.g. in Egypt or the U.S. high plains as reported by Gleeson et al., 2010)[21]. Conversely, when the potential inflows to the aquifer are high, it has been proposed that the rule could be to preserve a certain amount of outflows (Sophocleous & Devlin, 2004; Scanlon et al., 2012). Nevertheless, if there is plenty of water in the receiving surface water bodies, this may not be useful.

Thus, an approach based on the value of capture could be proposed as the rule, since it allows integrating the cases where its mobilization is effectively a necessary evil, but does not *a priori* disregard the cases where potential capture sources have a great value. On the other hand, it does not require minimum flows conservation when it is not necessary.

III.4.5 Current and future affection of surface water rights

The fact that groundwater pumping can reduce surface water flows and subsequently jeopardize the water rights associated with these flows is an issue that has been recognized for a long time, particularly in the U.S. where the specific doctrine of prior appropriation brought about numerous claims of surface water right holders (Alley et al., 2002). For example, Theis (1940) identifies implications for water rights that show also some similarity with the approach in terms of acceptability of the impact developed above:

> "If the recharge was previously rejected through transpiration from non-beneficial vegetation, no economic loss is suffered. If the recharge was rejected through

[21] Sometimes the storage depletion is such that the 'disconnection' between surface water and groundwater can be recognized and it could be made as if groundwater was not giving birth to surface water flows (as advised by Scanlon et al., 2012). However, this cannot be considered as a general rule.

> springs or refusal of the aquifer to absorb surface waters, rights to these surface waters may be injured. (…) after sufficient time has elapsed for the cone to reach the areas of natural discharge, further discharge by wells will be made up in part by a diminution in the natural discharge. If this natural discharge fed surface streams, prior rights to the surface water may be injured."

Thus, for Theis (1940), the principal issue is relative to water rights, while the environmental consequences are not taken into account as much, which is understandable in the 1940's[22].

The consequences for water rights holders are also described by Sophocleous (1998) who proposes a measure to adapt groundwater rights according to these impacts. They could correspond to the totality of inflows during the transient stage (because surface water flows are not affected) but they should consider surface water rights when surface water flows are diminished. However, this recommendation may be problematic since it does not include the 'future capture' linked to stock consumption during the transient stage.

III.4.6 Conjunctive use of groundwater and surface water

Based on the view developed in this chapter, it can also be remembered that the conjunctive use of surface and groundwater should integrate the impacts of groundwater pumping relative to current and future capture. Many hydrologic or economic models of conjunctive use introduce groundwater as an additional resource that is available in case of lack of cheaper surface water, without integrating these impacts.

III.4.7 Renewable resources, fossil groundwater and the meaning of mining

Terms such as 'groundwater mining' and 'non-renewable groundwater resource' are commonly used to refer to 'aquifer stock consumption'. In addition to the fact that this situation is often incorrectly defined based on withdrawals overpassing recharge, using these terms is questionable in their meaning and interpretation. 'Groundwater mining' makes a clear reference to mineral resources or oil, which are effectively non-renewable resources, with one of the main issues being the intergenerational equity in their allocation. Thus, this term should be reserved to cases where there are effectively no significant current inflows to the aquifer, i.e. the aquifer has been replenished in the past (see Custodio (2002) and the definition by Foster & Loucks (2006) in Table 3.3). The analogy with mineral resources is then justified.

[22] Theis (1940) also affirms: "The pumps should be placed as close as economically possible to areas of rejected recharge or natural discharge where groundwater is been lost by evaporation or transpiration by non-productive vegetation, or where the surface water fed by, or rejected by, the ground water cannot be used. By so doing this lost water would be utilized by the pumps with a minimum lowering of the water level in the aquifer."

Table 3.3. Glossary of key terms in an UNESCO publication. Source: Foster & Loucks (2006).

Term	Definition adopted	Explanatory comments
Non-Renewable Groundwater Resource	Groundwater resource available for extraction, of necessity over a finite period, from the reserves of an aquifer which has a very low current rate of average annual renewal but a large storage capacity.	Possible limiting criterion sometimes suggested is that the renewal period should be more than 500 years (average aquifer renewal less than 0.2% of aquifer storage). Some argue that a limiting average rainfall (say 300 mm/a) ought also to be included in definition. Usually expressed as the total volume of extractable groundwater or as an annual average flow rate for a fixed period, under practical field conditions (drilling accessibility and hydraulic productivity), realistic economic considerations (maximum affordable cost) and with consideration of potentially undesirable side-effects. The absence of significant replenishment is usually a consequence of very low rainfall in the unconfined areas of the aquifer but can also result from hydraulic inaccessibility in some confined aquifer.
Fossil Groundwater	Water that infiltrated usually millennia ago and often under climatic conditions different to the present, and that has been stored underground since that time.	Concept is thus both genetic and kinematic, since linked with both the mechanism of emplacement and the groundwater age (time passed since water under consideration entered aquifer) respectively. Groundwater age relates to period of residence of groundwater concerned within the aquifer, rather than (of necessity) the present-day absence of recharge of the aquifer system as a whole, and thus <u>*concept does not necessarily imply a non-renewable resource*</u>. Should not to be confused with connate groundwater, which is trapped in a geological strata since its formation and is thus often saline and frequently occurs in aquitards (rather than aquifers).

continues next page

Aquifer Overexploitation (Overdevelopment)	Prolonged (multi-annual) withdrawal of groundwater from an aquifer in quantities exceeding its average annual replenishment, bringing about a persistent fall in groundwater levels and reduction of aquifer reserves with undesirable side effects.	Concept of overexploitation is intended to indicate an imbalance within the groundwater budget of the aquifer system under consideration and the term aquifer overdraft is also sometimes used to indicate 'the amount of groundwater withdrawn from aquifer reserves'. However, the definition of time period and geographical area over which to evaluate this budget is always subjective and *interest is usually more in the side-effects of groundwater depletion on aquifer users, third parties and the environment (well yield reductions, saline water intrusion, land subsidence, ecosystem impacts, etc) than the process itself*. Important not to confuse aquifer overexploitation with active exploitation of an aquifer as a regulating reservoir (between seasons or in drought years) without any rupture of its long-term equilibrium.
Groundwater Mining	Extraction of groundwater from an aquifer having predominantly non-renewable resources with depletion of aquifer reserves.	*Process distinguishable from overexploitation of an aquifer with renewable groundwater resources, since in its case the reduction of aquifer reserves (with or without side-effects) is essentially permanent*.

Note: Highlights by the author of this thesis. / The definition of 'Aquifer Overexploitation' given here is questionable. It introduces the usual criterion of withdrawals higher than replenishment. Even if there is a reference to the environmental impacts, these occur also for pumping much lower than replenishment and an overuse of the aquifer could be considered then.

Meanwhile, when referring generically to the consumption of the stock, i.e. also in the existence of inflows to the aquifer, the term 'mining' is misleading since the characteristics of the physical access to groundwater and sustainability issues are quite different from mineral resources. Particularly:

- the groundwater stock may be recovered in a relatively short time after a reduction in pumping thanks to future capture;
- stock consumption occurs in the initial stage for all pumping (see Chapter 2) and it is thus not relevant to speak about mining in this case.

The same remarks apply for the term 'non-renewable resources', which can be considered as equivalent to 'groundwater mining' (see Table 3.3). One of the main problems relative to these terminologies is that they engage to sustainability debates relative to inter-generational equity and the consumption of a stock, often disregarding, and detracting, from issues related to the continuous impacts on surface water flows and groundwater dependent ecosystems. These concepts are also deeply linked to the traditional view of groundwater as a stock that can be safely pumped until reaching the amount of recharge.

However, having no current recharge does not necessarily imply that ecological impacts do not occur as noticed by Custodio (2002):

> "Groundwater mining means that the total volume of fresh water is reduced and/or replaced by poor quality or saline water, that part of the aquifer may become depleted, and that some springs, oases or other types of surface discharge may dry up. (…) the small renewable part, or the residual flow resulting from past climatic conditions, often plays an important environmental role, sustaining springs and wetlands (oases), creating valuable landscapes and habitats, and providing the required, shallow groundwater resources for the local population. Therefore, some ethical issues should be considered."

Finally, the identification of 'paleo-water' or 'fossil groundwater' pumping is also a trap, as it also generally triggers the debate on 'non-renewable resources' and 'groundwater that accumulated in the aquifer during a long time', with the idea that the issue at stake is intergenerational equity. However, the pumping of fossil groundwater can be compensated with current capture, producing flow impacts today (see the definition of 'fossil groundwater' in Table 3.3 and the case of La Loma de Úbeda in Chapter 8). Thus, lengthy residence time of groundwater in the aquifer under natural conditions does not allow identifying groundwater mining, contrary to a common consideration, e.g. by Gleeson et al. (2010), since pumping can greatly reduce the residence time under dynamic conditions and generate capture.

III.4.8 Should groundwater be defined as a Common-Pool resource?

Groundwater is often presented as a typical example of a Common-pool resource (CPR). This section will show that the formulation of groundwater as a CPR is directly related to the vision of aquifers as stocks of groundwater. Since the definition of a CPR constitutes one of the main justifications for the proposal of collective management by the users of the aquifer, the adequacy of this institutional arrangement will be also contemplated.

- <u>Stock impacts, basis for the definition of groundwater as a Common-pool resource</u>

The consideration of groundwater as a CPR is based on the combination of two characteristics: rivalness and non-excludability. While the issue of non-excludability as an intrinsic character of groundwater can be debated – since excludability depends on the institutions that can regulate the access to the resource (see Daly and Farley, 2010) –, it seems clear that the consumption of a volume of groundwater by a user will prevent another one from using this water. However, groundwater is usually defined as a CPR through the stock consumption, which deprives other and future users from the resource and implies higher pumping costs for all users. A main implication is that only the users of the aquifer, i.e. the users that have access to the stock and that will suffer higher pumping costs, are considered as affected by groundwater withdrawals.

Nevertheless, groundwater withdrawals potentially impact directly dependant ecosystems and the availability of water resources in the whole river basin for downstream users and the environment (Table 3.1). The traditional consideration of groundwater as a CPR disregards these uses and values. The decision-making and institutional arrangements that would integrate them are not contemplated in the CPR framework. Presenting groundwater as a CPR is based on its perception as a stock and contributes to reproduce this view. In fact, water coming from an aquifer is a CPR; it is not, however, due to this particular origin but to its inclusion in a wider CPR, the water resources of the whole river basin, which are rival for all the users in the river basin and hardly excludable at this scale.

- Common-Pool Resources problem and users involvement

The CPR framework is the basis for the proposition of the direct management of groundwater resources by users of an aquifer, or at least their greater involvement. The definition of a resource as a CPR implies that it is facing a specific challenge regarding its management. The non-excludability (at least for some users) implies that any user is able to withdraw as much resource as she/he desires, supposedly to maximize her/his own private benefits. This leads to the overuse of the resource in many occasions, which affects its capacity to regenerate and be a continuous source of supply in the long run. However, if users would coordinate to maintain the capacity of the system to regenerate, the outcomes would be better for all. It constitutes what Ostrom (1990) calls the 'CPR problem'. This is a specific collective choice paradox. Collective choice paradoxes apply to situations where the coordination of different individuals would lead to a better outcome for everyone than under individual action. Thus, CPR users have an inherent incentive toward cooperation to limit their use of the resource to ensure a continuous use through time (Figure 3.3).

The limit of use that does not jeopardize the resource base, i.e. resource availability, would emerge from full collaboration since it corresponds to the amount that maximizes users' profits in the long-run. This would supposedly lead to a win-win situation:

- The users of the CPR would have their profit maximized in the mid-to-long run.
- The resource would be conserved. This is an outcome that benefits the society as a whole, within an objective of conservation of the resource through time.

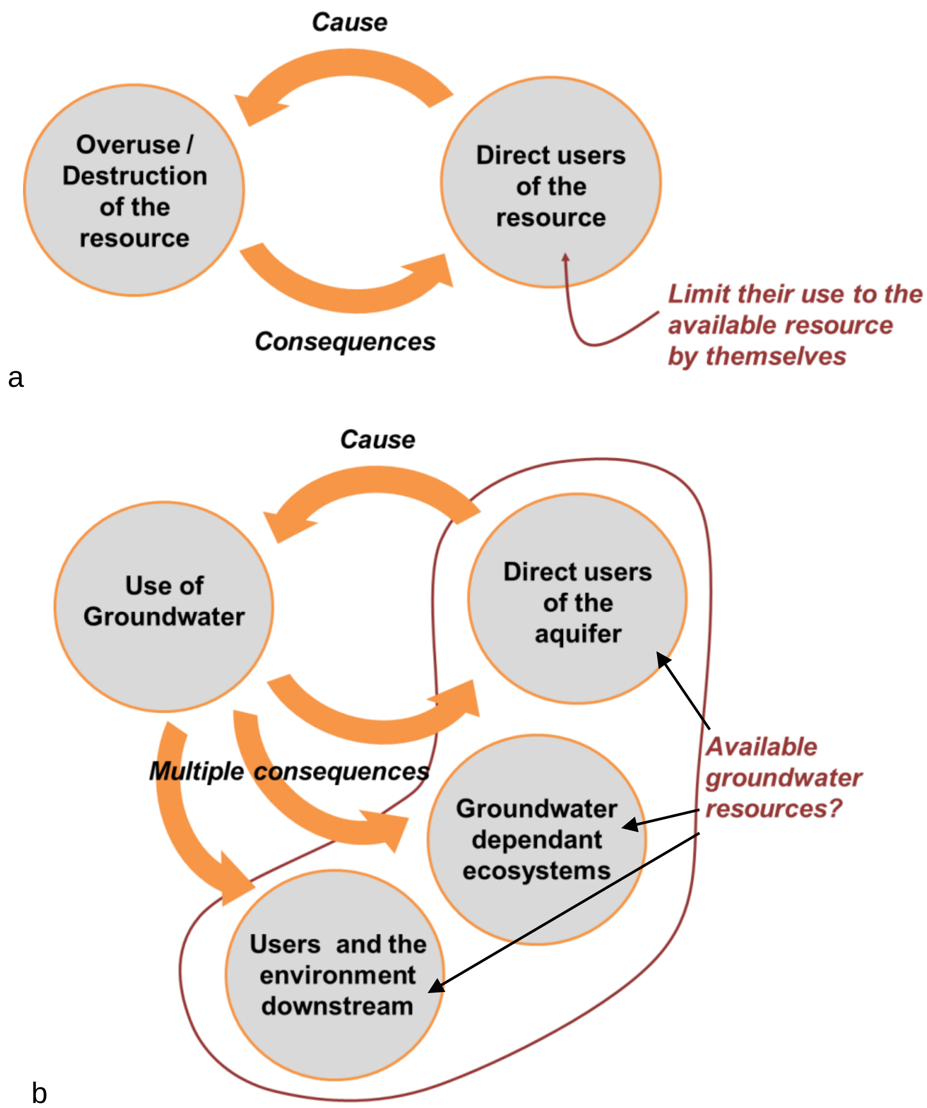

Figure 3.3. (a) Reciprocity between the cause and consequences of resource overuse and adjustment to the available resource (CPR framework); (b) Integration of multiple consequences of groundwater use to define available resources.

This framework constitutes a first justification to focus on the users of a resource and to propose a resolution of potential overuse based on their collective action. Regarding groundwater, the proposition for the collective action of users as a mode of governance is therefore based on this premise. An evident objective of a coordinated action by the users of the same aquifer would be to stabilize groundwater levels to limit the pumping costs. However, this can correspond to a situation where the capture of surface water flows is maximal with consequences for other uses in the river basin, as shown in Chapter 2. In fact, the interest of aquifer users would be to maximise capture. Thus, solving the 'CPR problem' does not guarantee that the resource is conserved for the other uses. This is not the win-win solution it initially appears. Consequently, the promotion of collective action by users of an aquifer following this line of argumentation is based on a mistaken premise.

Depending on the affected users from pumping, a series of other stakeholders and values would have to be integrated in the decision-making, such as the downstream users or the value of ecosystems (Figure 3.3 and Table 3.1). The definition of the available groundwater resources for a specific aquifer would result from this integration, as described in this chapter. This is a framework that is related to the 'Integrated Water Resource Management' at river basin scale. Foster & Ait-Kadi (2012) have notably presented an approach to include groundwater in this framework. Ross & Martinez-Santos (2009) have also reviewed the challenges to find governance mechanisms that integrate groundwater within the river basin and critically consider the approach as a CPR. It is noteworthy that even if the water in the whole river basin has the characteristic of a CPR (rival and no excludable), there are few examples of application of the governance framework based on the CPR approach at this scale (e.g. Van Oel et al., 2009a; Dinar & Nigatu, 2013). Management by users based on the CPR approach has been traditionally applied to more limited resource systems such as irrigation districts or aquifers.

A limit to the CPR analysis, at least in its initial approach, is introduced by Ostrom (1990, p.31):

> "[the] analysis [of CPR problems] relates to situation in which CPR appropriators have no power in a final-good market, nor do their actions have significant impact on the environment of others living outside the range of their CPR."

Groundwater is one of the main examples of CPR detailed in the book and was the object of E. Ostrom doctoral thesis. However, this citation reveals that the consequences of groundwater pumping outside the aquifer are not included. It also strengthens the framework shown on Figure 3.3 that illustrates that the CPR approach initially considers a bounded resource and its direct users only, who have an inherent reason to conserve it. The framework of a CPR problem and the general approach developed by Ostrom (1990) could be valid, however, for the specific cases where flow impacts of groundwater use are either irrelevant or accepted.

- <u>Justification based on successful experiences of groundwater management</u>

The institutional approach on CPRs has also contemplated both historical institutions that led to the conservation of CPRs and more recent intents of coordination in contexts where the CPR is potentially jeopardized. Successful experiences serve as an argument to propose the governance by local communities or users to be reproduced for CPRs. Thus, the successful experiences of collective action constitute another (more empirical) basis for its promotion.

This approach can also be questioned in the case of groundwater. The indicators that allow qualifying an institution as 'successful' should reflect all the impacts associated to groundwater pumping. Flow impacts are often disregarded, because of the view of aquifers as a stock of groundwater, and experiences described as successful may in fact overlook some aspects. Indeed the common view of groundwater as a stock implies that the usual indicators are the traditional ones, such as stability of levels or equality between recharge and pumping, as illustrated by Ostrom (1990, p.31):

> "As long as the average rate of withdrawal does not exceed the average rate of replenishment, a renewable resource is sustained over time."

The reproduction of successful modes of governance should be promoted in similar contexts and the success attested in specific situations (where impacts on surface waters and ecosystems are unperceived or acceptable) should not serve as the basis to support collective action as an institution for groundwater governance in general[23]. In other words, groundwater cannot be considered as a unique category of resource that could be managed under the same institutions, e.g. the management by direct aquifer users.

- Redefining the role of the users of an aquifer

The previous critiques of the ability of the CPR framework to deal with the governance of groundwater do not imply that more involvement of the users of an aquifer in its management should not be recommended. For instance, compliance with rules that are locally designed may be higher than externally-imposed regulation. Withdrawal monitoring would be also easier with the collaboration of the users. However, the space of this management should be well defined in relation with the other stakeholders. It should be limited to the repartition of the available resource established at the river basin scale. As a general rule, the allocation of water rights should be made in the whole basin, integrating all the stakeholders. The necessity to include a full range of stakeholder, and not only aquifer's users, is recognized by Custodio (2002):

> "Groundwater users should be involved in the management process, as should other people and organisations (stakeholders) who are directly or indirectly affected by aquifer use and who may be concerned for aquifer overexploitation. This includes groundwater developers, farmers, town authorities, water managers, land-use planners, decisionmakers, environmentalists, ecological organisations, groundwater scientists and experts, local people, and the mass media".

III.5 Synthesis

- In its common meaning, sustainability is a general objective for society implying that a harmonization of multiple (economic, social and environmental) values in an ideal and universally accepted state – a nirvana – is reachable. Thus, the concept disregards the

[23] For instance, various examples from California presented by Ostrom (1990) as successful concern coastal aquifers or aquifers sustaining intermittent river streams, whose conservation may not have been recognized as an issue, thus this is a specific context. Interestingly, Barraqué (2002), in the particular case of water markets, had already claimed for a better integration of the context to explain the emergence of this type of policy instruments and warned against generalization. Water markets in fact appear as linked to a theoretical posture rooted in neo-classical economics.

multiple trade-offs and necessary costs that most decisions on the use of natural resources and the environment entail.

- The multiple consequences of groundwater pumping in terms of capture, i.e. *where, when, how much and how long* the sources of capture are mobilized, should be clearly communicated during the decision-making process. This highlights the importance of knowledge, in order to take into account the full implications of groundwater pumping. Caps on flows to surface water and dependent ecosystems should be agreed in an integrated view at basin scale.
- A common recommendation to meet groundwater sustainability is to conserve the aquifer stock to allow for the continuous use of the aquifer, which is commonly interpreted as a situation where pumping is below recharge. This approach has been challenged based on two axes. First, a misconception of the groundwater dynamics conveys mistaken conditions to reach the steady state. Second, many authors insist on the fact that a sustainability assessment must integrate and balance a series of values, i.e. they build their critique from the general definition of sustainability.
- In fact, the traditional criterion of conserving the aquifer stock especially fails since it disregards the role of groundwater in the whole basin for all users and the environment, which would impose a cap on water flow consumption. This is often overlooked when a general integration of environmental, social, and economic values is required.
- Even in the absence of induced recharge, locally available resources cannot be obtained by subtracting the minimum agreed flows for the different source of capture to recharge. Reaching this level of pumping without specifying more conditions, such as the location of wells, would entail the respect of at least one agreed cap.
- The Common-Pool Resource approach of groundwater stems from the vision of aquifers as a stock of groundwater. Thus, it identifies the direct users of an aquifer as the only users who have an interest in the conservation of the resource, and this is a mutual interest with society (win-win). Therefore the proposition of management of aquifers by the direct users and other recommendations from this approach are based on a false premise. These are the whole resources of the river basin, including groundwater, that constitute a CPR.
- Flow impacts are also disregarded by the traditional approach in economic modelling of groundwater. Thus, these models elude the issue of ecosystems valuation. This adds to the arguable objective of an ‘efficient’ or ‘optimal’ extraction path based solely on the financial aspect and the simplifying hypotheses on the economic and physical contexts for models formulation. Moreover, the ‘history’ of the local area is disregarded, in particular the situation of rights initially granted. More fundamentally, the problem of

optimizing the consumption of the stock of an aquifer is based on a discount rate that implies future use of groundwater has less value than current use. Consequently, the results or governance propositions of these approaches are arguable, e.g. the much debated ‘Gisser-Sánchez’ effect, which affirms that groundwater regulation brings little benefits, compared to the open access situation.

- Based on the vision on groundwater dynamics and allocation developed in this thesis, it comes to light that terms such as ‘non-renewable resources’, ‘groundwater mining’, or ‘fossil groundwater’, which are often understood as technical terms, in fact need clarification. Beyond the challenges in their definition, these are also the common implications for potential actions and related policy recommendations that are questionable. Particularly, they engage in intergenerational equity debates that might detract from more urgent issues in terms of flow consumption, thus identifying mistaken policy goals.

Chapter IV

THE WATER FRAMEWORK DIRECTIVE AND GROUNDWATER OVERVIEW IN SPAIN

IV. THE WATER FRAMEWORK DIRECTIVE AND GROUNDWATER OVERVIEW IN SPAIN

IV.1 Introduction

One of the objectives of this thesis is to present an overview of the situation of groundwater resources in Spain, in terms of availability, quantification of the withdrawals and destination of the resource. As a result of the Water Framework Directive (WFD) implementation, 712 groundwater bodies (GWBs), the elemental management entity for groundwater introduced by this directive, have been defined (Figure 4.1)[24]. In the Water Plans, much data regarding their physical characteristics, groundwater flows and the use of the resource have been compiled to comply with one of the main goals of the WFD, an improved and detailed knowledge of water resources, which is a necessary step before any transparent decision on water planning.

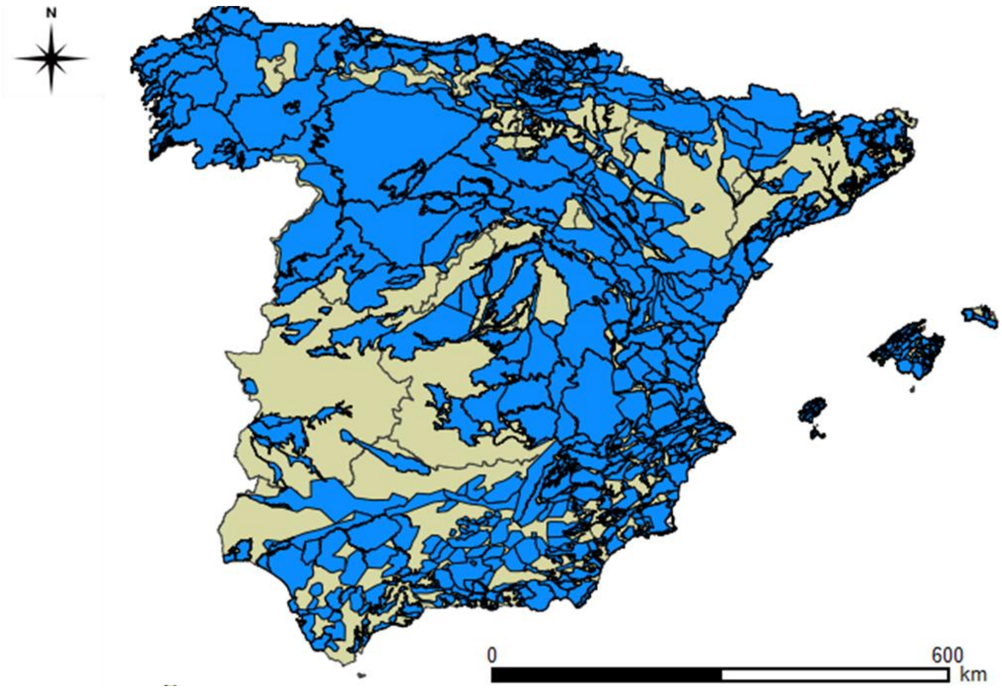

Figure 4.1. The 712 groundwater bodies defined in Spain. Source: Adapted from SIIA (2013).

The characterization of the quantitative situation of groundwater resources, i.e. their availability and the status of GWBs, is directly related with the framework for groundwater availability and allocation introduced in Chapter 3. The general objective of the WFD to have a special focus on the environmental role of water and its management at the scale of the river basin is also related. Thus, in addition to presenting an overview of groundwater resources in Spain, a second goal of this chapter is to analyse the criteria introduced in the WFD to assess the state of GWBs, the actual implementation of these criteria in the Spanish Water Plans, and whether these are in line with the objectives of the directive.

[24] The Canary Islands are not contemplated in this overview as each island has its own Water Plan, which complicates the access to data. The geology is also different from the rest of Spain, which may imply specificities that are difficult to assess. Identically the Ceuta and Melilla districts are not considered.

The first section of this chapter reviews the criteria introduced in the WFD and their transposition in the Spanish legislation, which is compared in turn to the actual application in the Water Plans. The second section presents an overview of groundwater resources in Spain by river basin district and at the scale of the whole country. The average inflows to the aquifers, the 'available resources' (as understood by the Water Plans) the withdrawals by sector of the economy and the status of the GWBs are successively presented. Even if the focus for the whole chapter is on the quantitative aspect, the GWBs general status includes also the chemical status.

IV.2 Quantitative status and available resources: the Water Framework Directive and its implementation in Spain

IV.2.1 The definition of the Groundwater bodies

The boundaries of the GWBs are mainly based on geological and hydrogeological criteria since these water bodies are constituted by an only aquifer or, more generally, a group of aquifers. The boundaries are defined therefore relatively to impervious contacts or discharge areas (rivers). In Spain, the process has consisted in the redefinition of the previous Hydrogeological Units. However, the physical criterion is not the only criterion since the GWBs are defined also taking into account the practical management of groundwater resources (Ministerio de Medio Ambiente, 2005a). For instance, some subdivisions have been established to reduce the area where a specific problem, such as pollution or intensive withdrawals, should be contemplated. Moreover, aquifers that used to not be integrated into groundwater planning, mainly local aquifers, had to be defined in compliance with the WFD when they supply more than 10 m^3 per day for domestic use.

IV.2.2 Risk of non-compliance versus groundwater bodies status

Two different characterizations of the state of the GWBs in the implementation of the WFD should be distinguished:

- *The risk of non-compliance* with the objective of the WFD at the end of the planning process is the first assessment that has been realized, in partial application of Article 5 of the directive. It was due for 2005 and consisted in the detailed characterisation of the river basin: pressures, impacts and economic analysis[25]. It should include a review of the impact of human activities on the status of surface and ground water. This

[25] More precisely, Annex II of the WFD states that 'Member States shall carry out an initial characterisation of all groundwater bodies to assess their uses and the degree to which they are at risk of failing to meet the objectives for each groundwater body under Article 4 relative to the environmental objectives' (European Commission, 2000). It is this requirement that gave birth to the expression 'GWB at risk' to refer more concisely to the GWB that fail to meet this requirement.

characterization should be more detailed for the GWBs ‘at risk’. The specific results relative to groundwater have been compiled by the Ministry of environment in 2006 (Ministerio de Medio Ambiente, 2006) (see Appendix 2).

- *The GWBs chemical and quantitative status assessment* characterizes the state of the GWB at the time of the preparation of the Water Plans. The result is the classification of GWBs in ‘good status’ or ‘poor status’. It represents the real state of the GWB, contrary to the risk of non-compliance which is an evaluation on the future state (Figure 4.2).

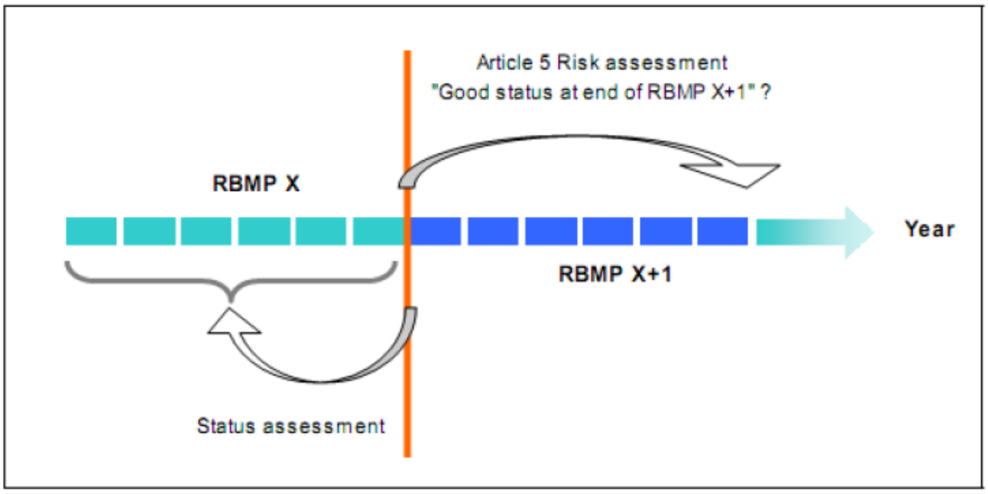

Figure 4.2. Risk assessment regarding the status objective looks into the future whereas status assessment looks back on the performance. Source: European Commission (2009).

Even if there are differences in their conceptualization, the distinction between these two steps may not be so clear in reality and the term ‘GWB at risk’ is sometimes used instead of ‘GWB in poor status’. The real problem from not differentiating these two steps is that the criteria used in the two assessments are not the same, which potentially leads to different results. For example, as will be illustrated for the case of Guadalquivir River basin district, in Section 2.6 of this Chapter, some GWBs have been assessed as ‘at risk’ of no-compliance, yet they are not considered as in ‘poor status’. The opposite situation can also be found.

The criteria for the determination of the GWBs ‘at risk’ are not detailed in the text of the WFD, contrary to the criteria for the GWB status assessment, which are detailed and discussed in the next section. The criteria introduced in the Guadiana district (Tables 4.1 and 4.2) are an example of the application in Spain. They are based on the concepts of pressure and impacts that are introduced in the Annex II of the WFD associated with the implementation of Article 5.

As the approach is quite open, the methods and criteria used in the different districts have been variable (see Appendix 2). It means that any comparison between the results of each district should be made with caution, as results depend on the criteria used for the assessment. In these conditions, no detailed comments of the criteria and results of the assessment undertaken for Article 5 will be made in this chapter. The discussion will instead focus on the criteria on good quantitative status assessment, which is the assessment included in the final Water Plans (see next section).

Table 4.1. Risk of non-compliance with the objectives of the WFD according to the combination of pressures and impacts from the human activity on the water body. Source: translated from Ministerio de Medio Ambiente (2005b).

		Impact			
		Verified	Probable	Without impact	Without data
Significant pressure	Yes	High	Medium	Low	Medium
	No			Without	Low
	No data			Low	---

Table 4.2. Example for the Guadiana District for pressure (a) and impacts (b) characterization relative to the groundwater withdrawals. Source: translated from CH Guadiana (2005).

Pressure: Withdrawals

Criterion	Threshold
$Use\ Indicator = \frac{\sum d}{R}.100$ $\sum d$: Sum of the demands of the Hydrogeological Units corresponding to the GWB (urban supply and irrigation) R: Available resources respective to the Hydrogeological Units corresponding to the GWB (infiltrations + irrigation return flows – environmental restrictions – transfers to other Hydrogeological Units)	> 40 %

Impact assessment on resource quantity

Diagnosis	Criterion	Threshold
Verified impact	- Noticeable trend toward decrease in levels over time (y=ax + b)	Slope (a) < (-1)
Probable impact	- Moderate trend toward decrease in levels over time (y=ax + b)	0 < Slope (a) < (-1)
Without visible impact	- No significant trend toward decrease in levels over time (y=ax + b)	Slope (a) = 1
	Or	
	- Trend of increase in levels over time (y=ax + b)	Slope (a) > 0

In addition to groundwater withdrawals, the other 'pressures' are: diffuse pollution, point pollution, artificial groundwater recharge, seawater (and other) intrusion.

The result of the assessment undertaken in application of Article 5 (Ministerio de Medio Ambiente, 2006) is presented in Table 4.3, with the update presented by Varela (2009). The number of GWBs was 699 at this time.

Table 4.3. Assessment of the groundwater bodies 'at risk' (chemical and quantitative). Source: Ministerio de Medio Ambiente (2006) and Varela (2009).

	Number of groundwater bodies	'At risk'				In study	'No risk'
		P	D	I	W		
Ministerio de Medio Ambiente, 2006	699	259				256	184
		80	167	72	164		
Varela, 2009 (Ministerio de Medio Ambiente data source)	699	343				96	260
		82	225	84	214		

Chemical risk: P: point contamination; D: diffuse contamination; I: seawater intrusion – Quantitative risk: W: withdrawals.

IV.2.3 'Good status' and 'available resources' under the Water Framework Directive: a new approach integrating environmental requirements?

Compared to traditional approaches, the criteria introduced by the text of the WFD to assess the quantitative status of the GWB seem to represent a renewed and ambitious approach. It clearly associates the 'good status' of a GWB to the status of the associated surface water bodies and groundwater dependant ecosystems (Table 4.4). The compliance with the requirements for surface water bodies established in Article 4, which deals with environmental objectives, should not be jeopardized and groundwater dependent ecosystems should be preserved. These conditions imply a limit to pumping and the definition of acceptable impacts. This seems in line with the framework for groundwater availability developed in Chapter 3.

More precisely, the criteria presented in the WFD can be separated in two sets that can be viewed as more or less equivalent or complementary (Table 4.4): criterion *a* complemented by the definition of the term 'available groundwater resource', on the one hand, and criteria *b*, *c* and *d*, on the other hand. Whereas the term 'accordingly' suggests that the second set of criteria makes explicit the first one, they are not strictly equivalent. The second batch of criteria is more general and is in line with the approach that has been detailed in the Chapter 3, which focused on the impacts of pumping on surface water bodies and groundwater dependent ecosystems. Applying these criteria would require a detailed assessment of these impacts, i.e. characterizing the mobilization of capture through time and the conditions that prevent 'significant damages'. The main limit is that these criteria are very general. However, these should be clearly established as the ultimate goal and not replaced by criteria that are overly restrictive.

Table 4.4. Definition of the good quantitative status of groundwater bodies and available groundwater resources according to the Water Framework Directive. Source: adapted from European Commission (2000).

Elements	Good status
Groundwater level	**(a)** The level of groundwater in the groundwater body is such that the available groundwater resource[a] is not exceeded by the long-term annual average rate of abstraction. Accordingly, the level of groundwater is not subject to anthropogenic alterations such as would result in: **(b)** – failure to achieve the environmental objectives specified under Article 4[b] for associated surface waters. **(c)** – any significant diminution in the status of such waters. **(d)** – any significant damage to terrestrial ecosystems which depend directly on the groundwater body. **(e)** And alterations to flow direction resulting from level changes may occur temporarily, or continuously in a spatially limited area, but such reversals do not cause saltwater or other intrusion, and do not indicate a sustained and clearly identified anthropogenically induced trend in flow direction likely to result in such intrusions.

[a] The available groundwater resource is defined as **"The long-term annual average rate of overall recharge of the body of groundwater less the long-term annual rate of flow required to achieve the ecological quality objectives for associated surface waters specified under Article 4, to avoid any significant diminution in the ecological status of such waters and to avoid any significant damage to associated terrestrial ecosystems."**

[b] The Article 4 of the WFD specifies the environmental objectives, which are the core of the directive.

Compared to this general approach, the criterion *a* is more detailed and practical. The definition of 'available groundwater resource' can, however, be questioned taking into account two issues:

- It mentions the 'long-term annual average rate of overall recharge'. Speaking of 'overall recharge' implies that the induced recharge should be taken into account. However, there is no reference to this term and the potential impacts linked to the increase in recharge (see Chapter 2). If the induced recharge is implicit in this definition, this remains a static vision since it suggests the 'rate of overall recharge' is a fixed value.
- A 'long-term annual rate of flow' that should be maintained towards surface water bodies and groundwater dependent ecosystems shall be subtracted to the rate of recharge. As explained in Section 4.3 of Chapter 3, this is problematic as a general rule in case of multiple sources of capture, since the different minimum flows that should be conserved are not cumulative. Only a detailed assessment of the dynamic of groundwater as a result of pumping can define an aquifer development path compatible with the maintenance of all the acceptable rates of outflow.

- Specifically within the previous axis, there is no reference to the conservation of the underground flows between GWBs, which often take place. This is all the more relevant when the boundaries are based on management criteria.

Consequently, the definition of available resources would be valid only in the simple situation presenting only one source of capture, with a decrease in outflows potentially accompanied by an increase in inflows occurring for the same source (see Section 4.3 of Chapter 3).

To sum-up, even if the definition of available resources includes a reference to an environmental flow, the introduction of the ‘long-term annual average rate of overall recharge’ is a reminiscence of the traditional approach and fails in integrating a dynamic view. The focus is still on the flows that enter the GWB, as recharge constitutes basic data. A renewed approach and a real change of paradigm would have considered primarily the acceptability of the impacts from pumping on the surface water bodies and dependent ecosystems. The approach also do not contemplate explicitly the potential increase in inflows. On the contrary, the criteria *b*, *c* and *d* are in line with a view based on assessing the impacts of capture.

Another major issue in the definition of ‘available groundwater resources’ in the WFD is the lack of reference to the role of groundwater for water resources availability in the whole river basin. Although this is implicit, since the principle of integrated management at the scale of the whole river basin is a basic element of the WFD, the view here is focused on directly dependent surface water bodies and ecosystems, as evidenced by the reference to the ‘associated surface waters’ and ‘associated terrestrial ecosystems’. Thus, the contribution of the GWB outflows for all the water bodies located downstream is overlooked.

In fact, it would be better to distinguish the quantification of groundwater that is effectively available for direct users of a GWB from the assessment of the status of the groundwater body. As stressed in Chapter 3, the definition of available resources for users of an aquifer must be integrated within the whole river basin. A view at the local scale makes sense if the objective is to determine the amount that can be pumped without harming the directly associated ecosystems and surface waters. In this case, a term different to ‘available resources’ could be used, such as ‘accessible resources’, and introduced in the criteria for the quantitative status assessment. Inversely, a value of available resources based on a view at the scale of the whole basin would not be meaningful to be assessed if the GWB is in ‘good status’. It would imply that a GWB is in ‘poor status’ if the outflows are committed for downstream users, without having problems relative to the state of its waters and the locally immediate dependant ecosystems and surface waters.

The tendency to overlook downstream impacts is usual beyond the WFD. The interaction of groundwater with dependent ecosystems and surface water is increasingly better understood, leading to the growing integration of environmental flows and ecosystem requirements in the debates on groundwater allocation (Sophocleous, 2002; Boulton & Hancock, 2006; Klove et al., 2011). However, in general, there is still a lack of vision at the scale of the whole river basin.

IV.2.4 In Spain: the Water Planning Regulation

In Spain, the detailed criteria for the assessment of the good quantitative status of groundwater bodies in application of the WFD is presented in the Water Planning Regulation (*Instrucción de Planificación Hidrológica*, MARM, 2008) (Figure 4.3 and Table 4.5). Faced with the variability of the methods for the assessment of the GWB 'at risk', as described in Section 2.2 of this chapter, one of the aims of the regulation was to harmonize the criteria. A first limit is that the Spanish central State is only competent for water planning in the river basins shared between different Autonomous Communities, meaning that the application of the regulation is only a guide for the districts managed by the Autonomous Communities.

In general, the transcription in the Water Planning Regulation for the criteria introduced in the text of the WFD is quite straightforward, with the introduction of two sets of criteria (Figure 4.3). Again, it is not clear if these sets should be considered as equivalent to each other or if all the criteria should be assessed.

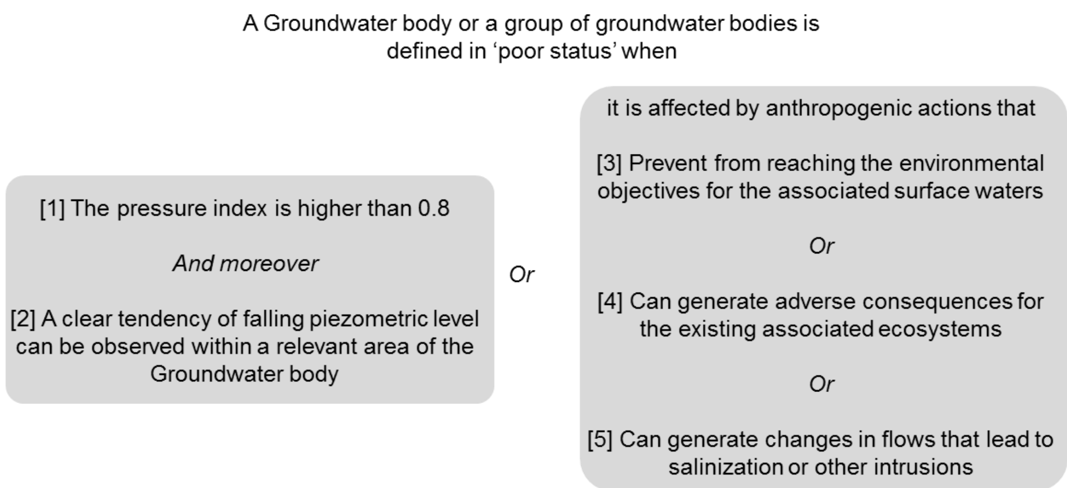

Figure 4.3. Criteria to assess the quantitative status of groundwater bodies in the Water Planning Regulation. Source: adapted from MARM (2008).
The definitions of the terms 'pressure index', 'available resources' and 'renewable resources' are presented in Table 4.5. The section relative to these criteria taken from the Water Planning Regulation is presented in Appendix 3 (in Spanish).

Table 4.5. Definition of the pressure index, available resources and renewable resources of a groundwater body according to the Spanish Water Planning Regulation. Source: MARM (2008).

Term	Definition
Pressure index	The ratio between abstractions and available resources. This indicator is obtained based on the average value of resources for the period 1980/81 – 2005/06 and the abstractions data corresponding to normal conditions of delivery in the last years.
Available resources	The inter-annual average value of the total recharge rate of the groundwater body, less the inter-annual average flow required to reach the objectives of ecological quality for the associated surface water to prevent any significant diminution in the ecological status of these waters, and any significant impact on the associated terrestrial ecosystems. (…) this will be obtained as the difference between the *renewable resources* (…) and the environmental flows required to comply with the ecological flows regime and to prevent the negative effects caused by marine intrusion.
Renewable resources	Recharge from rain infiltration, recharge from irrigation return flows, losses in the riverbeds and transfers from other groundwater bodies.

The original definitions in Spanish are presented in the Appendix 3.

A similar approach as for the WFD consisting in comparing withdrawals with the available resources is introduced. It is based on the difference between total recharge and environmental flows that would allow maintaining sufficient flow towards surface water and groundwater dependent ecosystems and prevent marine intrusion. The same critiques made before for the WFD text are also applicable: the problem with potential induced recharge, the absence of a dynamic approach, the reproduction of the traditional view focusing on recharge, and the issue of including multiple sources of capture. A main difference is the introduction of a safety factor of 0.8, i.e. withdrawals shall be less than 80 % of available resources.

The most relevant difference is the inclusion of another criterion that should also be verified, in addition to the comparison of withdrawals and available resources. A clear trend of falling groundwater levels should be observed on a significant area of the GWB. A continuous drop of the groundwater level corresponds to two potential situations (see Section 3 of Chapter 2): the transient stage or a pumping rate that is higher than the maximum capture that can be mobilised. Thus, two common situations may not be picked up, if the cross-compliance of the two criteria *1* and *2* (Figure 4.3) has to be verified:

- The transient stage can produce a substantial drop in the level if it lasts years or decades, even if the withdrawals are only a limited fraction of the available resource. This situation should be identified as problematical, all the more so since some local sources of capture can be mobilized (see Chapter 2 on groundwater dynamics). Thus, adding the temporal dimension in the criteria of a lowering groundwater table appears essential. It should be noted that this situation would not be identified with the criterion of the WFD text, which does not include the consideration of the transient stage.

- Inversely, the groundwater level can be constant with potentially a high share of the capture mobilised, with the associated impacts in terms of flows.

Another questionable point is the inclusion of the transfer from other GWBs as a full component of recharge (Table 4.5). A correct definition would have mentioned the ‘net transfer’ from other GWBs – underground transfer received less the necessary underground transmission of flows to receiving GWBs, to allow them to comply in turn with the criteria. This point is further discussed and illustrated in the next section.

IV.2.5 Status assessment on the ground: the Spanish Water Plans

Beyond the debate on the criteria to report the state of groundwater and available resources in theory, the actual application of the Water Planning Regulation criteria in the different river basin districts in Spain illustrates the practical challenges. The criteria are not identically addressed, which implies a potential additional distance from the general principle of the WFD and the framework presented in Chapter 3. The review of the Water Plans undertaken for this section allows drawing attention to a series of issues[26]:

- consideration of impacts on surface water and associated ecosystems, and intrusions, which is a basic characterization to comply with the objectives of the WFD;
- combination of the pressure index and the continuous drop in levels (see Figure 4.3);
- factors included in the renewable resources, particularly underground transfers;
- calculation of environmental flows and integration in the river basin – these two last points being the basis of the calculation of available resources.

Comparison is also possible for the districts that are managed by the Autonomous Communities, since these must also comply with the requirements of the WFD, even though they do not have to follow the Water Planning Regulation.

[26] The references for the Water Plans that were analysed are: Agencia Catalana del Agua (2010), Govern de les Illes Balears (2011), Agencia Vasca del Agua (2012), Augas de Galicia (2012), Junta de Andalucía (2012a, 2012b, 2012c), CH Cantábrico (2013a, 2013b), CH Duero (2013), CH Ebro (2013), CH Guadalquivir (2013), CH Guadiana (2013), CH Júcar (2014), CH Miño-Sil (2013), CH Segura (2014), CH Tajo (2014). It is noteworthy that data and results are not presented uniformly within the different Water Plans. This makes it hard to find information, which is presented in different sections of the main reports or different annexes from one plan to another. An effort of harmonization to make the access to data more direct and transparent is advisable for the next rounds of water planning.

- Consideration of impacts on surface water and associated ecosystems, and intrusions

One of the criteria for good quantitative status of a GWB according to the Water Planning Regulation, in concordance with the objectives of the WFD, is that groundwater pumping should not jeopardize the compliance with the objectives for associated surface water bodies, affect groundwater dependent ecosystems or generate intrusions (criteria *3*, *4* and *5* on Figure 4.3). Seawater intrusion is usually adequately contemplated. On the contrary, the direct impacts on surface water and groundwater dependent ecosystems is hardly assessed; some exceptions are the Guadalquivir, Segura, and Mediterranean Andalusia districts, which take into account affected ecosystems or the decrease in spring flows, for instance.

- Combination of the pressure index and the criterion of continuous drawdown

The quantitative status assessment relies almost entirely on the pressure index (criteria *1* on Figure 4.3) and the drop in water levels (criteria *2*), since the other criteria are hardly taken into account in the Water Plans. Yet there are differences in the way these criteria are introduced in the different districts (Figure 4.4). Only a few districts, such as Duero and Ebro, apply strictly the Water Planning Regulation and combine both criteria[27]. In the case of Ebro, it has resulted in the classification of only one GWB in ‘poor status’, even if ten GWBs present a pressure index higher than 0.8. These ten GWBs would have been classified in ‘poor status’ applying the criteria in the same way than the majority of the districts: they consider that any of the two criteria *1* or *2* leads to ‘poor status’, which is less restrictive and identifies more ‘poor status’ situations. Applying both criteria together would have resulted in four GWBs less in ‘poor status’ in the Mediterranean Andalusia district[28].

It can be also observed that some GWBs have been classified in ‘good status’ despite having their pressure index close to the established limit of 0.8, e.g. the Doñana ‘Almonte-Marismas’ GWB in the Guadalquivir District, with 0.797, or the Madrid Manzanares-Jarama GWB in the Tajo District, with 0.78. These are two well-known cases, as these two aquifers are essential for public water supply or have ecological value, and there have been measures in place for long to improve groundwater management. Nevertheless, this situation is noteworthy from the point of view of reporting due to the uncertainty of the factors of the pressure index and the implications of a classification as in ‘poor status’ or ‘good status’, particularly for future Water Plans since, under the WFD, there is no possibility for water bodies to deteriorate.

[27] In the case of the Duero district, a GWB has been defined in ‘poor status’, despite a pressure index lower than 0.8, as the drop in levels was identified to be ‘clearly related to human activity’.

[28] In the Segura District, GWBs are classified in ‘good status’ despite having a pressure index higher than 0.8, in case this index is lower than 1 and it is proven that there is no decrease in levels.

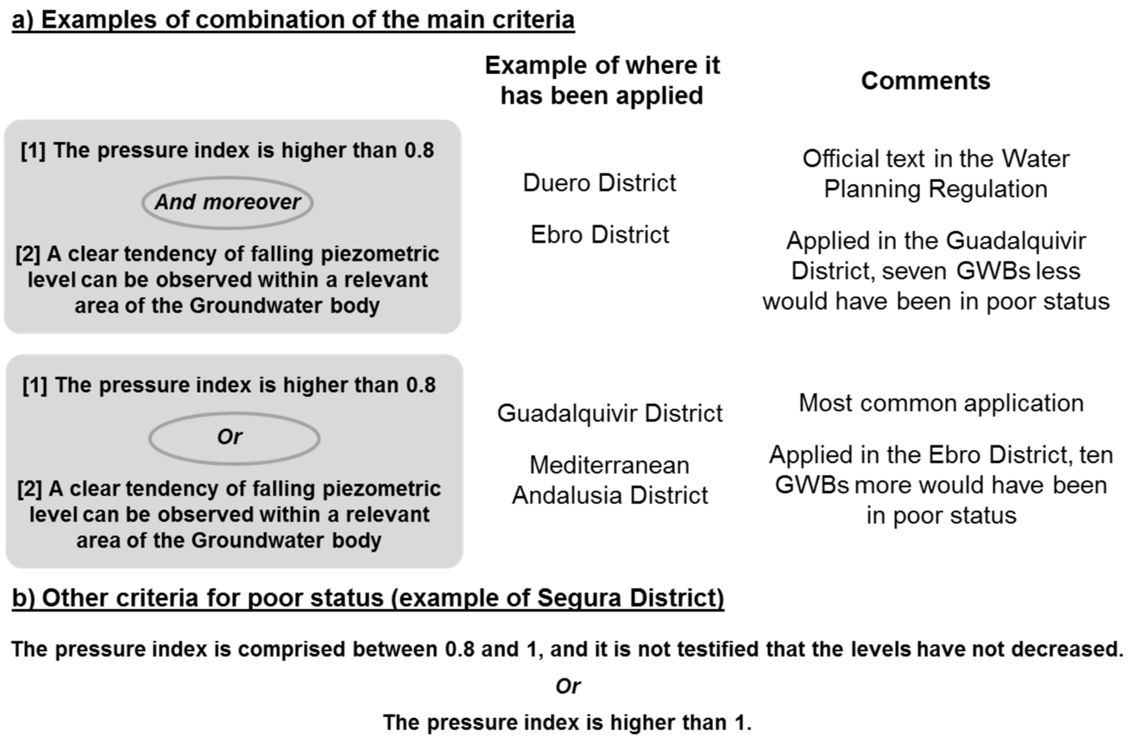

Figure 4.4. Different combinations of the pressure index and decrease in levels for the status assessment of the groundwater bodies in the Water Plans.

- <u>Estimation of resources in the Water Plans with a particular focus on underground transfers</u>

'Available resources' are one of the factors of the pressure index. Thus, its quantification is determinant for the GWB status. The analysis of the Water Plans has shown that its calculation is very different between districts for two main reasons. First, recharge or inflows – often called 'natural resources' or 'renewable resources' – can include different terms (Figure 4.5). For instance, inflows from rivers or return flows from irrigation are not always included. Depending on the district, the natural state or the dynamic state is considered as the reference, which has also some influence on the final result. The evaluation of the different terms is also based on methods and tools (e.g. computer programs) that were usually developed by each district.

Second, underground transfers, i.e. flows between adjacent GWBs, are contemplated under four main approaches:

1) The net transfer in natural conditions, i.e. flows naturally received less the flow naturally transferred, is included in 'renewable resources'. This approach appears to be the most conservative for environmental and downstream impacts since the natural underground flows are conserved (see the case of Ebro on Figure 4.5). The net transfer is usually considered as a component of recharge or 'natural resources'.[29]

[29] It is noteworthy that in the Duero District the underground flows from and to adjacent GWBs generated by pumping are accounted as additional resources and additional withdrawals respectively.

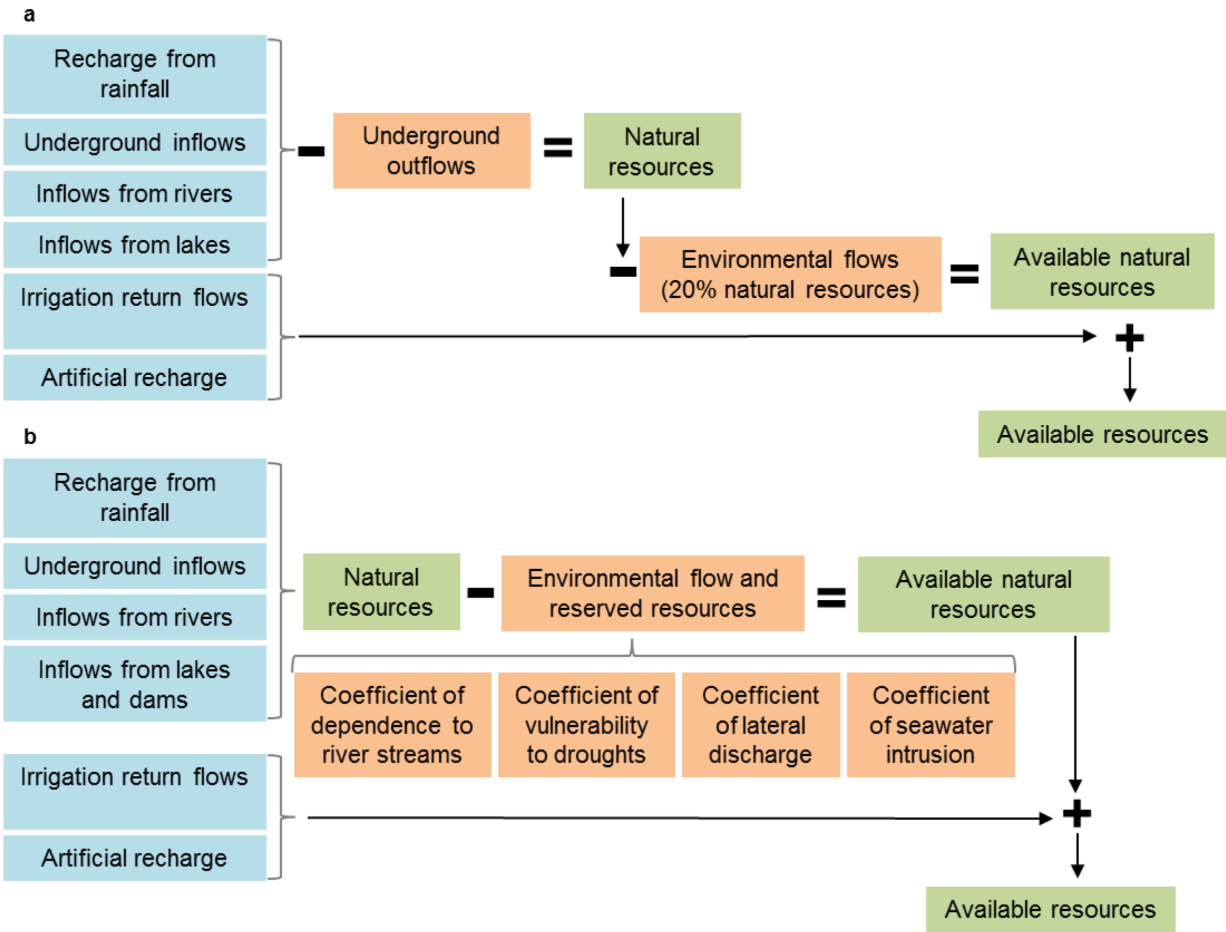

Figure 4.5. Available resources definition in Ebro and Guadiana Districts (a) and in the Mediterranean districts of Andalusia (b). Source: (a) CH Ebro (2012) and CH Guadiana (2011); (b) Agencia andaluza del agua (2011).

2) In other cases, such as Cataluña and Mediterranean Andalusia districts, a minimum underground outflow towards the receiving GWBs is established (Figure 4.5). Thus, underground outflows are viewed in the same way as environmental flows to surface water and dependent ecosystems or flows to prevent seawater intrusion. The inflows from other GWBs are fully integrated in 'natural resources'. In other words, the flows towards 'downstream' GWBs are included as a source of capture like the others. In Chapter 3 (Section 4.3), it has been shown that, in case of various sources of capture, the different minimum flows cannot be added to compute the available resources. It is especially relevant in the case of underground outflows from a GWB, since in many situations, these flows would be the last ones to be affected by pumping (Figure 4.6).

3) It can also happen that only the inflows from adjacent GWBs are considered, the outflows being disregarded. This approach is the generalization of the traditional view that contemplates the whole inflows to an aquifer as the 'available resource', which can be very problematic in the case of flows that are essential for the receiving GWBs. This is the case for the Guadalquivir District in some situations or for the Júcar District,

where 80 % of underground inflows are integrated in the 'renewable resource' and the direct transfer to other GWBs is disregarded[30].

4) Finally, underground transfers are sometimes not contemplated, apart from the case when these are coming from another river basin, e.g. in the Tajo and Segura Districts. In the same way as the previous approach, resources are identified as available for the receiving GWB, despite being naturally transmitted to adjacent GWBs. However, for most of the GWBs underground inflows that may effectively constitute a potential additional resource are not accounted for.

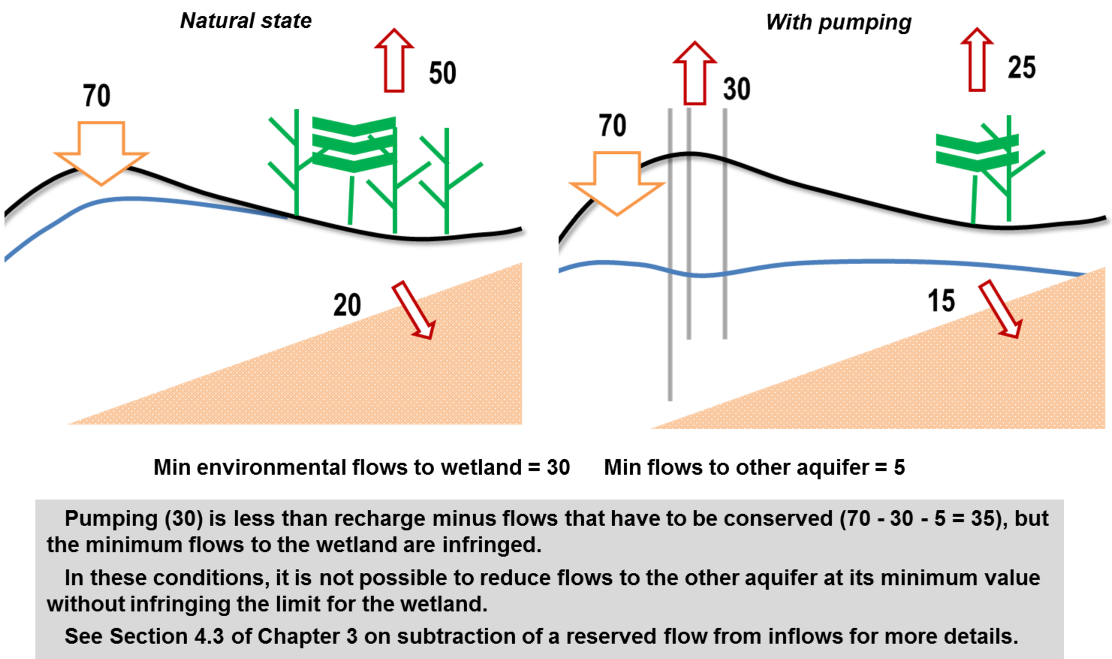

Figure 4.6. Subtraction of minimum flows from recharge in the case of underground flows.

In the approaches *2* and *3*, there is also a problem of multiple accounting for the same flows as available in different GWBs. This issue is especially relevant when the boundaries of the GWBs are not based on physical factors. In fact, this should be considered for all sources of inflows, as the identification of groundwater resources as available at the scale of a GWB should imply it is no longer available downstream.

It should be remembered that, even if the Water Planning Regulation introduces the concept of underground transfers, contrary to the WFD text, it does not explicitly refer to net transfers, which can explain the potential confusion in its application.

[30] The accounting in the Júcar District could be interpreted also under the approach *2*: a reserved flow of 20% of underground inflows. However, the focus is clearly on accounting inflows. Outflows are basically overlooked.

- Environmental flows and integration in the river basin

The determination of environmental flows is challenging since it requires the detailed knowledge of groundwater-surface interactions. The lack of studies and data can explain why a majority of districts introduced environmental flow as a fixed share of inflows. Apart from the major shortcoming of not reflecting the diversity of local conditions, this approach is problematic due to the value of the percentage considered, which is often only 20% of the inflows, or sometimes 50%, when ecosystems particularly dependant on groundwater are identified (e.g. in Guadiana, Guadalquivir, Duero). In other words, it means that up to 80% of the inflows to a GWB are identified as potentially usable within the limits of the aquifer, as the term ‘available’ suggests. In a context where interactions between groundwater and surface water and dependent ecosystems only begins to be acknowledged, this approach may harm ecosystems whose degree of dependence on groundwater has not yet been identified.

Some districts, such as the Júcar, Segura, and Cataluña Districts, present a more detailed assessment of groundwater / surface systems interaction, based on local studies, with varying values for each GWB. In the Segura district, environmental flows are null for many GWBs but reach values higher than 50% for four GWBs (Figure 4.7). Nevertheless, the questionable consideration of underground transfers in these districts (see approaches *2*, *3* and *4* in the previous section), implies that the ecological flow might ‘feed’ the underground transfers towards other GWBs (Figure 4.6).

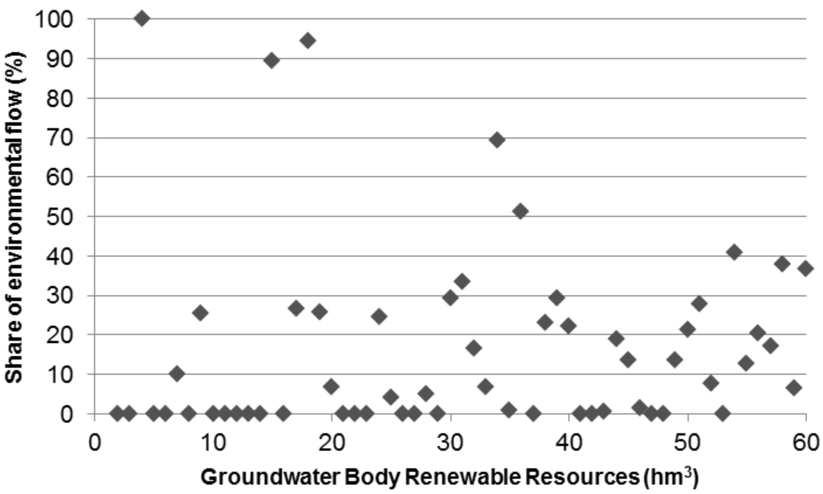

Figure 4.7. Renewable resources reserved for environmental flow in the Segura district. Source: own elaboration based on CH Segura (2014).

In the Tajo district, the percentage of environmental flows varies between 20% and 60%. However, it is not based on specific local studies, but depends on the situation of the GWB (upper or lower stretch of the river basin), the presence of dams downstream, and an estimation of the degree of dependence of the surface water bodies and ecosystems. Eventually, the case of the Northern districts (Galicia, Miño-Sil, Cantábrico Oriental, Cantábrico Occidental) is noteworthy since the value of environmental flows is based on the maintenance of flows during low water season, with a minimum set at 10% of the inflows to the GWB. This low value is not so problematic, due to the local climatic conditions and the limited groundwater pumping in these regions.

The approach consisting in identifying the available resources of a GWB as up to 80% of the inflows is a clear illustration that the availability of groundwater resources is not addressed at the scale of the whole river basin, suggesting that up to 80 % of inflows could be abstracted by the direct aquifer users.

IV.2.6 Synthesis and implications

The criteria introduced by the WFD to report the status of the GWBs and to determine the available resources, which are also introduced in the Water Plans through the application of the Water Planning Regulation, initially appears to be in line with a renewed approach that integrates the environmental impacts of groundwater pumping, in line with the general objectives of the WFD. Nevertheless, this may only be illusory since the most challenging criteria (criteria *3*, *4* and *5* in Figure 4.3), which require assessing in detail the impact of groundwater withdrawals, are not directly used in most cases. The most practical criteria of a high pressure index and dropping levels constitute the basis of the assessment. However, their questionable formulation and the inertia of the traditional approaches imply that, in the Water Plans, the integration of the environmental requirements is realized in its minimal form and there is no overall perspective at the scale of the river basin. Thus, it can be questioned if the assessment of the WFD process represents a renewed approach regarding the current challenges of groundwater resources management.

Regarding the variability of methods in the different districts, it can be said, first, that the choice of criteria to assess the situation of groundwater resources is a challenging task. No criterion is adaptable to every situation. In fact, in many districts, an additional ‘expert evaluation’ was also led on the basis of the Water Planning Regulation criteria or based on additional data. This is a good point to consider specific situations; however, some guidelines appear necessary to have some points of comparisons and to be able to evaluate if the assessment was correctly done. The risk of an ‘expert evaluation’ is to reproduce the traditional criteria and it might have the added problem of a lack of transparency.

Second, the choice of criteria influences many parameters in the status assessment and different criteria have been used to assess the state of groundwater resources (Table 4.6). A main reason for the variability of the approaches is the lack of precision of the Water Planning Regulation and WFD texts. Third, the inertia of the traditional approaches is another reason that can explain why the actual application of some criteria hardly meets the objectives of the WFD. It means that the exceptions, like the detailed assessment of the environmental flows in the Júcar and Segura Districts, or the consideration of affected groundwater dependent ecosystems in the Guadalquivir District, generate a divergence.

Table 4.6. Synthesis of the main points of divergence in the quantitative status assessment.

Point of divergence	Comments
Factors included in the natural or renewable resource	- Irrigation return flows or river seepage are not always considered. - Different models and methods have been used to calculate rainfall infiltration and the other factors. - Natural state or dynamic state.
Underground transfers	- Three main approaches: net natural flows, a minimum outflow or no consideration of the outflows for 'downstream' GWBs.
Environmental flows	- Usually a fixed share of inflows (20% or 50%), but, in few cases, a detailed assessment based on local context.
Drop in levels and pressure index	- The Water Planning Regulation (combination of the two criteria) is followed in few cases and usually one of the two criteria leads to the 'poor status'. - Other criteria are sometimes introduced (e.g. Segura District).
Additional criteria	- An assessment of the impacts of withdrawals directly on flows, springs or wetlands is exceptional (e.g. Gudalquivir River Basin).
Expert evaluation	- Some of the documents make direct reference to the evaluation by experts. Even if a blind application of guidelines is not desirable, this might hinder the possibility of comparison and transparency.

The diversity of methodologies implies that a strict comparison between districts cannot be undertaken. The application of the criteria of one district to another would have resulted in a different result. The influence of the choice of criteria can also been viewed for the same district at different stages of the WFD implementation (Figure 4.8). The variability at the scale of one country is also a reality between the different countries of the European Union (Henriksen & Refsgaard, 2013).

Finally, the efforts led in the WFD process for the characterization of the state of groundwater resources, which should be recognized, may have been in vain without adequate frameworks and criteria to assess the situation. The critiques that have been addressed in this chapter are possible thanks to the new data presented in the Water Plans. Some districts have advanced significantly on particular points in the evaluation. However, this can be entirely questioned if there is a problematic point, such as the detailed consideration of surface-groundwater interaction accompanied by a questionable consideration of underground flows in the Júcar district. In addition, the goal of the status assessment is not only to have an overview of the situation of water resources, it is primarily to identify where means and efforts should be concentrated, which can have consequences for the GWBs not identified as in 'poor status'.

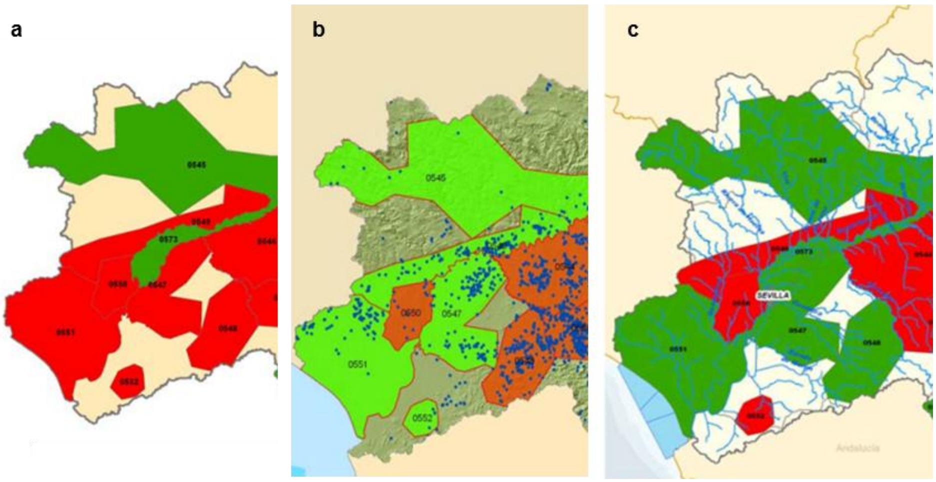

Figure 4.8. Changes in the groundwater bodies 'at risk' or 'in poor quantitative status' in the Guadalquivir District. Source: own elaboration based on CH Guadalquivir (2005, 2010, 2011).
(a) Article 5 report (CH Guadalquivir, 2005); (b) Intermediary document (*Esquema de temas importantes*) (CH Guadalquivir, 2010b); (c) River Basin Management Plan (CH Guadalquivir, 2010a)

IV.3 Overview of groundwater in Spain according to the Water Plans

IV.3.1 Number of the Groundwater Bodies and area occupied

The WFD implementation lead to the definition of 744 GWBs that cover more than two thirds of the Spanish territory with 355,000 km^2. This is around twice the area occupied by the former divisions of groundwater management, the 411 hydrogeological units that covered around 175,000 km^2 (De Stefano et al., 2013). The increase in area is mainly due to the integration of aquifers that are locally important, particularly for urban supply, following the requirements of the WFD. The increase in number is also linked to the subdivision of some hydrogeological units into smaller GWBs, mainly for management reasons, e.g. tackling local quality issues.

IV.3.2 Groundwater availability by district

Table 4.7 presents the average value of groundwater inflows to the GWBs and the 'available resources' defined in the Water Plans for each district[31]. In average, for the whole territory of

[31] See footnote 26 (p. 68) for the whole list of the Water Plans that have been analyzed in this thesis.

Spain, around 34,000 hm^3 of water infiltrates into the ground and reaches the aquifers. A small share of this amount is constituted by irrigation return flows that can be important locally for some GWBs. Because of the regional climatic variability, infiltration is higher in the North of the country (Galicia Costa, Miño-Sil, Cantabrico Occidental and Cantabrico Oriental).

Table 4.7. Number of groundwater bodies and available resources by district.

District	Number of groundwater bodies	Groundwater bodies area (km^2)	Inflows to aquifers (hm^3)	Environmental flows (hm^3)	Available resources (hm^3)
Galicia Costa	18	12,990	3,869	398	3,471
Miño-Sil	6	17,594	3,774	581	3,193
Cantábrico Occidental	20	13,875	4,217	889	3,328
Cantábrico Oriental[a]	14	3,523	1,272	182	1,090
País Vasco	14	2,267	508	91	417
Duero	64	75,885[b]	3,737	747	2,990
Ebro	105	54,125	4,207	841	3,366
Cuenca Fluvial de Cataluña	39	11,254	1,930	789	1,141
Tajo	24	21,842	1,736	658	1,078
Guadiana	20	22,484	555[c]	102	453[c]
Júcar	90	40,135	3,355	1028	2,327
Segura	63	18,500	692[d]	146	546[d]
Guadalquivir	60	35,609	2,700	738	1,962
Tinto, Odiel y Piedras	4	1,018	66	20	46
Guadalete-Barbate	14	1,927	166	114	52
Cuencas Mediterráneas Andaluzas	67	10,395	833	157	676
Islas Baleares	90	4,737	410	229	181
Islas Canarias[e]	32	7,425	-	-	360
Total	744	355,564	34,000	7700	26,700

[a] Area under administration by the Spanish State. The other part of the Cantábrico Oriental District is managed by the Basque Country Autonomous Community and appears as 'País Vasco'./ [b] This area corresponds to the lower or 'general' groundwater bodies; twelve GWBs have been defined upon these ones in the upper layers. / [c] Including irrigation return flows of 49 hm^3. These values correspond to the period 1980-2005 and were updated to 564 hm^3 of available resources in the final version of the Water Plan to take into account higher rainfalls in the period 2005-2012. / [d] Including irrigation return flows of 86 hm^3. / [e] Source: De Stefano et al. (2012).

IV.3.3 Use of the sectors of the economy

An important disclaimer to this section is that, similarly to the term of 'natural' or 'renewable resource', the term 'use' is imprecise. A main confusion is on the consideration of total use, i.e. withdrawals, or consumptive use only. This issue is extensively debated in Chapter 5 and 6 on the WF concept and case study methods, as the WF computes consumptive use. Nevertheless, in traditional water management at the scale of the river basin, figures usually refers to total withdrawals and the debate in this section is related to another source of confusion: is the figure relative to allocated rights, real use, or estimated demand? Water Authorities usually register rights, but these are not necessarily fully used each year, and an estimation of real use is also complicated by illegal use, particularly in the case of groundwater. Demand is also an imprecise variable, as not all the demand is necessarily satisfied. Consequently the results for withdrawals or demand presented in Table 4.8 can have different significance from a district to another. As a general rule, it can be acknowledged that the districts located in the North-West of the country overestimate real use as water rights are not fully used, and the other districts may underestimate real use due to illegal use.

Based on the information contained in the Water Plans, the total use of groundwater resources reaches 6,700 hm^3 per year as an average (Table 4.8). The main use of groundwater in Spain is for irrigation. It represents almost three quarters of the use (5,000 hm^3). The rest of the use is almost entirely destined for urban supply (1,400 hm^3). The share of groundwater on total water use, 25 % for agriculture and around 28 % for urban supply and industry, is quite consistent among the different sectors. For urban supply, this share is particularly low as compared to the situation in other countries. It is a paradox since supply from groundwater is safer as compared to surface water, particularly under a semi-arid climate with repeated periods of drought. This situation might be explained by the traditional lack of consideration of groundwater resources for water supply in Spain, compared to surface water development based on big infrastructures (Llamas & Martínez-Santos, 2005). Exceptions are the Cantábrico occidental, Cantábrico oriental, and Júcar districts, with around 60 % and Islas Baleares, with 80 % from groundwater.

It is also noteworthy that three districts, Guadalquivir, with 950 hm^3, Duero, with 1050 hm^3, and Júcar, with 1610 hm^3, concentrate more than half of the groundwater withdrawals. The Júcar district is also the only one, together with Islas Baleares, to have groundwater constituting more than half of its water resources supply.

Table 4.8. Total water and groundwater withdrawals / demand by sector.

	Agriculture		Urban supply		Own supplied industries		Leisure and other		Total	
	GW	Total[a]	GW	Total[a]	GW	Total[a]	GW	Total[a]	GW	Total[a]
Galicia-Costa		84	49	274	1	45	-	1	51	404
Cantábrico oriental	1	8	31	53	1	72	-	1	34	134
Cantábrico occidental	5	93	94	162	7	213	-	7.1	105	474
Pais Vasco	1	-	35	195	1	13	11	-	48	209
Miño-Sil	28	297	15	64	5	36	9	47.5	56	445
Tajo	135	1,797	48	787	55	62	-	-	237	2,646
Ebro	252	7,681	38	358	46	147	-	-	336	8,186
Duero	965	4,048	69	331	20	84	-	-	1,054	4,463
Cuenca Fluvial de Cataluña	197	388	198	632	59	110	-	8	454	1,138
Guadiana	438	1,995	46	151	4	44	-	-	488	2,190
Júcar	1,178	2,474	323	548	100	124	9	10	1,610	3,156
Segura	478	1,432	-	237	-	23	-	14	478	1,706
Guadalquivir	833	3,329	104	436	11	36	-	-	948	3,801
Tinto, Odiel y Piedras	31	151	4	56	-	46	2	2	37	255
Guadalete y Barbate	39	320	20	122	-	-	4	6	63	448
Cuencas Mediterráneas Andaluzas	377	838	140	336	3	23	19	28	539	1,225
Islas Baleares	49	68	142	174	3	3	0	8	194	253
Total[b]	5,000	25,000	1,400	5,000	300	1,100	50	130	6,700	31,000

GW: groundwater / [a] Groundwater and surface water / [b] Without Islas Canarias.

IV.3.4 Groundwater bodies status according to the WFD assessment

The result of the assessment on the status of the GWBs shows that 296 GWBs are in 'poor status' in Spain (without the Canary Islands), 166 for quantitative reasons and 229 presenting a poor chemical status, i.e. 414 are in 'good status' (Table 4.9 and Figure 4.9).

Table 4.9. Status of groundwater bodies according to the WFD assessment.

District	Number of groundwater bodies	Good status	'Poor status' Quantity	Chemical	Total	Number of groundwater bodies in 'good status' in the WFD next rounds
Galicia Costa	18	18	0	0	0	–
Miño-Sil	6	5	0	1	1	2021: 6
Cantábrico Occ.	20	20	0	0	0	–
País Vasco	14	13	0	1	1	2021: 14
Cantábrico Ori.	14	13	0	1	1	2015: 14
Duero	64	48	5	14	16	2015: 47 - 2027: 50
Ebro	105	82	1	23	23	2021/2027: 103
Cuenca fluvial de Catalunya	39	14	6	23	25	2015: 18 - 2027: 39
Tajo	24	18	0	6	15	2021: 22 - 2027: 24
Guadiana	20	5	11	13	15	2021/2027: 20
Júcar	90	50	30	27	40	2021: 57 - 2027: 87
Segura	63	16	41	24	47	2015: 17 - 2021: 19 – 2027: [illegible]
Guadalquivir	60	32	19	16	28	2015: 35 - 2021: 48 - 2027: 60
Tinto, Odiel y Piedras	4	2	0 (ID: 1)	2	2	2015: 4
Guadalete y Barbate	14	5	3 (ID: 8)	7 (ID: 2)	7 (UA:2)	2021: 7 - 2027: 10 (UA: 2)
Cuencas Mediterráneas Andaluzas	67	27	32	35	40	2015: 41 - 2021: 52 - 2027: 62
Islas Baleares	90	46	18	36	44	2015: 64 - 2021: 75 - 2027: 87
Total[a]	712	414	166	229	296	2015: 456 - 2021: 508 - 2027: 671

[a] Without Islas Canarias. The two groundwater bodies not assessed are only included in the total number of groundwater bodies. / ID: insufficient data; UA: under assessment

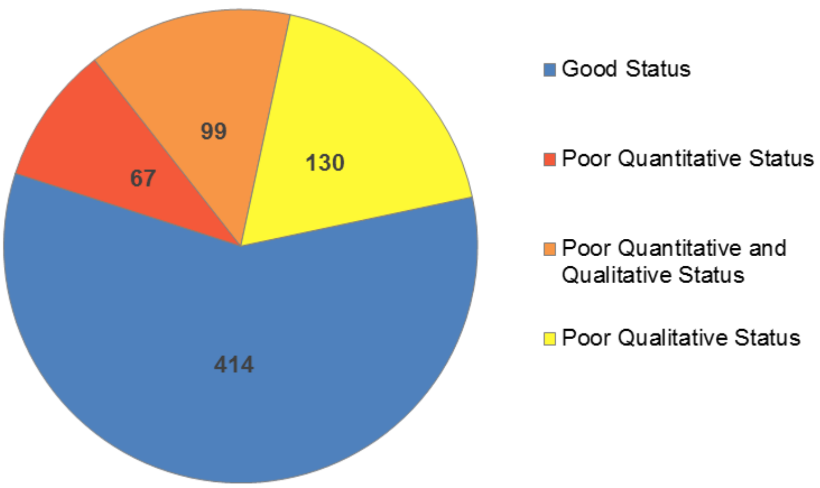

Figure 4.9. Groundwater bodies according to the reason for 'poor status' classification.

This view in terms of number of groundwater in ‘poor status’ allows having a general overview; however, it may not reflect the real implications for groundwater management. In addition to the debates on the definition of criteria to assess the status of the GWBs and their harmonization, as discussed previously, the relevance of a view in terms of number of GWBs may be questioned due to the high variability in their size, which means different challenges for their management (De Stefano et al, 2013). Another view of the situation could be obtained, for instance, considering the share of groundwater use associated with GWBs in ‘poor status’ (Table 4.10).

Table 4.10. Number of groundwater bodies in ‘poor status’ and the associated withdrawals in the Guadalquivir District. Source: Dumont et al., 2011a.

	Number	Associated withdrawals
Groundwater bodies	60	822.6 hm^3/year
Groundwater bodies in ‘poor status’	19 *(32%)*	618.5 hm^3/year *(75%)*

Regarding the period to reach the WFD ‘good status’, the Water Plans assess that the majority of the GWBs classified as in ‘poor status’ will reach ‘good status’ for the year 2027 (Figure 4.10). Less stringent objectives have been established only for 39 GWBs, i.e. around 5 %. Most of them are located in the Duero (14), Segura (10) and Andalusia districts (7). Thus, many districts have not opted for this classification, potentially since it should be precisely justified. This shows again a difference of approach between the different districts.

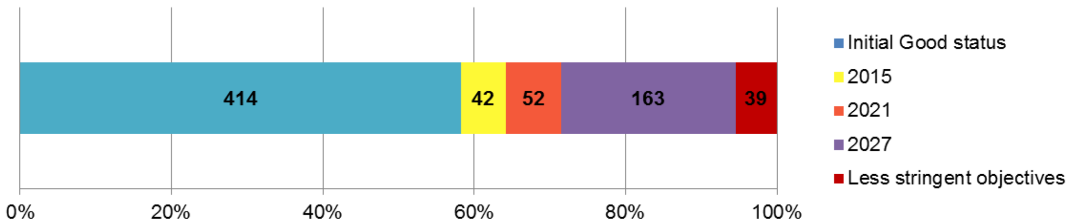

Figure 4.10. Time to reach ‘good status’ and less stringent objectives for groundwater bodies.

IV.4 Synthesis

- Despite the objective of the WFD to fully integrate the environmental dimension in water resources management, the practical assessment on the state of groundwater bodies and groundwater availability reproduces conventional approaches. It comes initially from the criteria of the WFD based on assessing a drop in levels and comparing withdrawals to ‘recharge’ minus an environmental flow. These criteria eventually prevail on the requirement to assess in detail the interaction between groundwater and surface water bodies and ecosystems, when the WFD is implemented, as evidenced in the Water Plans in Spain.

- Regarding the definition of ‘available resources’, in addition to environmental flows that are often roughly generally estimated, a major problem is the lack of integration at the scale of the whole river basin. Underground lateral flows between groundwater bodies are also a main issue, as there is a range of approaches in the different Water Plans, some of them questionable, mainly due to multiple accounting of flows.
- Despite the objective of a standardised approach by the Water Planning Regulation, there is a lack of consistency, with a disparity in the methods and criteria used in the different river basin districts, which complicates the comparison and has implications on the final perception and overview of the state of groundwater resources in Spain.
- A major benefit of the WFD, relative to the state and use of groundwater resources in Spain, has been the publication of much data and the formulation of measures to improve the state of groundwater. However, the presentation of basic information should be harmonized, for a more convenient access and a more robust and truly comparable overview.
- According to data presented in the Water Plans elaborated in compliance with the WFD, the total use of groundwater reaches around 6,700 hm^3, which represents 20 % of the water use in Spain. Three quarters of the use (5,000 hm^3) is dedicated to irrigation, followed by urban supply (1,400 hm^3) and own supplied industries (300 hm^3).
- 296 (40 %) of the 712 groundwater bodies (without Canary Islands districts) are classified in ‘poor status’. 166 for quantitative reasons and 229 for chemical reasons. Most of the groundwater bodies defined as in ‘poor status’ are planned to reach ‘good status’ in 2027 and less stringent objectives have been defined for only 39 groundwater bodies.

Chapter V

WATER FOOTPRINT: DEFINITION AND DISCUSSION

List of symbols of this chapter

WF_{ideal}	water footprint without losses in the supply chain (m^3/ha)
WF_{real}	water footprint with losses in the supply chain (m^3/ha)
w	percentage of losses in the supply chain (%)

V. WATER FOOTPRINT: DEFINITION AND DISCUSSION

In compliance with the objective of the thesis to assess the relevance of the WF, this chapter reviews the WF approach, introducing the general methodology, main applications, and main debates on its formulation and meaning. As the WF has been subject to a variety of interpretations, the aim of this chapter is also to show how the WF is contemplated in this thesis.

This chapter is organized as follows. The first section deals with the definition and the different perspectives in the application of the WF approach. It includes an analysis of the origin of the WF concept together with a broader assessment of the concept of 'footprint'. In the second section, the physical accounting of blue and green water is described, showing its specificity compared to traditional accounting for water resources management. The third section is dedicated to the interpretation of the WFs, with a special emphasis on the comparison between a volumetric WF, which is the traditional approach, and an impact-weighted WF, an approach in line with Life Cylce Assessment (LCA). Finally, some proposals to extend the scope of the WF approach are presented.

V.1 Water footprint assessment: definitions and origin of the approach

V.1.1 Direct and indirect appropriation according to multiple scale and perspectives

The WF measures human appropriation and contamination of freshwater resources according to two perspectives (Hoekstra et al., 2011)[32]: a) the production from a defined geographical area, usually referred to as the 'WF from the perspective of production', and b) products and entities of the supply chain like factories, retailers, or final consumers (individual consumers, households, the population of a city, a region or a whole country, etc.), referred to as the 'WF from the perspective of consumption'. In the latter case, the view from the final consumers and from businesses or sectors of the economy are sometimes distinguished (Figure 5.1).

A WF is constituted by the sum of the water resources consumption of unitary processes involving direct withdrawals from the environment (Figure 5.1). These basic components are then aggregated on part of or the whole supply chain (perspective of consumption) or for a geographically delineated area (perspective of production).

[32] Even if the general definition of the WF encompasses both the consumption and contamination of freshwater resources, this thesis focuses on the consumption aspect. Thus, the contamination aspect is sometimes omitted, particularly in the general presentation of the present chapter. This is also for more clarity in the explanations. However, it should be kept in mind that the original definition of the WF by Hoekstra et al. (2011) integrates both dimensions.

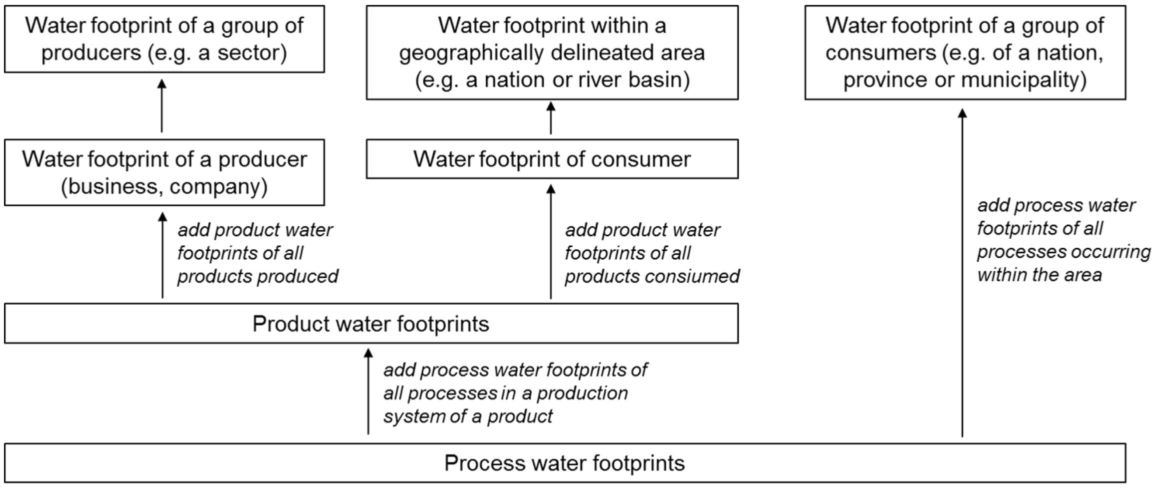

Figure 5.1. Process water footprints as the basic building block for all other water footprints. Source: adapted from Hoekstra et al. (2011).

From the perspective of consumption, the direct WF – water resources directly consumed by an entity – and the indirect WF – the sum of the direct water consumption of the previous (or upstream) entities – can be distinguished (Figure 5.2). While the reduction of the consumption of water resources has traditionally targeted the direct withdrawals, introducing a view in terms of indirect use shows the real impacts from the different entities of the supply chain and introduces new options for actions to reduce the pressure on water resources (Hoekstra et al., 2011). For instance, a change in the consumption pattern of a final consumer or in the suppliers for a company can have an influence on their indirect WF. A company can also help its suppliers manage water resources better. It is noteworthy that the WF of a product is the same as its 'virtual water content', i.e. the water resources that have been consumed to make it available.

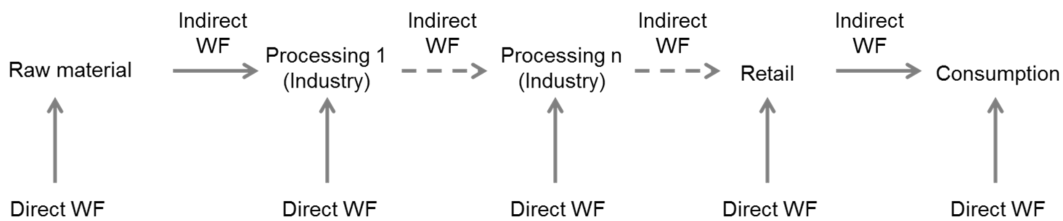

Figure 5.2. Direct and indirect water footprints along the supply chain. Source: own elaboration based on Hoekstra et al. (2011)

The WF concept has introduced, or at least reinforced, in the realm of water resources management a focus in terms of human footprint, that had previously taken hold in relation to the ecological footprint (Wackernagel & Rees, 1996; Hoekstra, 2009), or life cycle assessment (LCA) (Boulay et al., 2013; Guinée & Heijungs, 2005). It has emphasized the role of consumers and a view in terms of product and supply chain (Hoekstra et al., 2011; Zimmer, 2013). It has also highlighted that water resources management should not only be approached at the scale of the river basin: many drivers for water resources consumption, particularly associated with global trade, are located out of the river basin boundary (Chapagain et al., 2005). According to Hoekstra (2009):

> "[the WFs] illustrate the hidden links between human consumption and water use and between global trade and water resources management."

Traditional water resources management, which contemplates the appropriation of water within an area, usually river basins or aquifers, is also included in the WF framework, which appears as very inclusive. Thus, the WF is a flexible indicator that can be applied at any scale, for any entity or group or a geographical area to measure the water consumption that is generated by or within the object of study, whether directly or indirectly.

In this thesis, the focus is on the WF from a geographical area, as the WF approach is applied to various aquifers and a river basin. However, the perspectives from production and consumption are not incompatible points of view. The basic component under both standpoints is the direct WF of a process (Figure 5.1). Thus, the 'perspective of consumption' is sometimes adopted in this thesis, particularly for agricultural products.

V.1.2 Water footprint of nations, the first approach

- Initial formulation of the water footprint concept

The formulation of the WF presented in the previous section has progressively evolved since the first definition of the concept by Hoekstra & Hung (2002), who defined it under the perspective of water resources appropriation by nations in a report focusing on virtual water trade:

> "For each nation of the world a 'water footprint' has been calculated (a term chosen on the analogy of the 'ecological footprint'). The [WF], equal to the sum of the domestic water use and net virtual water import, is proposed here as a measure of a nation's actual appropriation of the global water resources. It gives a more complete picture than if one looks at domestic water use only, as is being done until date."

Thus, the initial formulation of the WF concept integrates the debates on virtual water flows between countries and the traditional management of domestic resources. Here the WF of a country or region can be perceived as a complementary view to virtual water flows. An assessment of a national or regional WF (i.e. in the perspective of consumption) integrates the balance of virtual water flows of imports and exports.

The popularization of the concept of virtual water is, however, linked to a specific interpretation. The import of 'embodied water' contained in agricultural products could potentially 'save' water for the importing country, allowing a possible reduction in its water deficit (Allan, 1998, 2003). Allan (2003) clarifies its own use of the terminology:

> "The author (…) used the term to draw attention to the notion that serious local water shortages could be very effectively ameliorated by global economic processes."

Here, the focus is therefore not on assessing water resources appropriation and the potential environmental consequences in the country of origin but rather on assessing options for the country subject to water shortages.

- Evolution of the approach

Two main and parallel developments of the WF approach should be highlighted:

- *A progressive focus on products, supply chain and individual consumers*. The initial definition of the WF deals with the WF of countries. The first studies about a specific product contemplate a cup of coffee and a cup of tea consumed in the Netherlands (Chapagain & Hoekstra, 2003a, 2003b). However, they refer to the 'virtual water content' of one cup[33]. The term WF is still used under the perspective from the whole nation. This is the same in the analysis of the WF of cotton (Chapagain et al., 2005) and the different products assessed in the study on the WF of nations (Chapagain & Hoekstra, 2004). A view from the perspective of a product and the supply chain is formalized for the first time in the report on business WF accounting by Gerbens-Leenes & Hoekstra (2008), i.e. six years after the initial formulation of the WF concept.
- *Toward a view in terms of impacts*. The first definition of the WF concept by Hoekstra & Hung (2002) and the subsequent applications, e.g. Chapagain & Hoekstra (2003a, 2003b, 2004) do not focus on the consequences of the appropriation of water resources in and outside the country. They highlight the amount of freshwater needed for national consumption without questioning the implications for water management in other countries. The first study that explicitly makes reference to the issue of environmental impacts and implications in terms of policy and equity in the allocation of water is the assessment of the WF of cotton (Chapagain et al., 2005). It is also the first study to introduce the grey WF to also report the environmental impacts in terms of pollution.

- Water footprint of nations, regions or cities under the 'consumption perspective'

The initial perspective of the WF, the WF of nations, has been subsequently applied also to other geographical areas, such as regions or cities, i.e. considering the whole consumption of the population in these areas. Many WF assessments have been undertaken under this perspective:

[33]In fact, even if not referred to as a WF, the focus on the virtual water content of a cup of coffee or tea is innovative in these studies, as it relates directly to a consumption good. The virtual water content of products had already been considered and calculated (e.g. Zimmer & Renault, 2003) but only as a step to assess the virtual water flows between countries and not to highlight, for example, the elevated amount of water resources needed for the production of a product in the country of origin. It is noteworthy however that Zimmer & Renault (2003) proposed methods for the quantification of the 'virtual water value' of 'primary products', 'processed products', and 'by-products'. Renault (2003) also assessed the virtual water content of diets, in relation with the previously introduced concept of 'nutritional water productivity' (Renault & Wallender, 2000). This is a few years before the debate brought about by a view based on the aggregation of the water footprint of products (e.g. Vanham et al., 2013).

- *Referring directly to the WF concept*: Morocco and the Netherlands (Hoekstra & Chapagain, 2007a), China (Zhao et al., 2009), Spain (Garrido et al., 2010), Netherlands (Van Oel et al., 2009b), Indonesian Provinces (Bulsink et al., 2010), the United Kingdom (Feng et al., 2010), France (Ercin et al., 2013), the European Union (Steen-Olsen et al., 2012; Vanham & Bidoglio, 2013) or regions and cities such as the Madrid Autonomous Community (Naredo et al., 2009) or Beijing (Zhang et al., 2011; Huang et al., 2014). A series of studies have also assessed the WF of all countries in a global approach, successively refining the calculations or including new sectors (Hoekstra & Chapagain, 2007b; Fader et al., 2011; Hoekstra & Mekonnen, 2012a).
- *Referring to the virtual water concept, but focusing on regions or countries, like the WF approach*: China (Ma et al., 2006), India (Verma et al., 2009), Andalusia (Velázquez, 2007; Madrid & Velázquez, 2008) or the Yellow River Basin (Feng et al., 2012).

V.1.3 Water footprint of products, consumers and economic sectors

As explained in the previous section, one of the developments of the WF concept has been to contemplate products, supply chains and consumers and a series of WF assessments have adopted this perspective over the last few years[34]:

- A product:
    - *Produced or consumed in a specific area*: coffee and tea consumed in Netherlands (Chapagain & Hoekstra, 2007), Spanish tomatoes (Chapagain & Orr, 2009, Chico et al., 2010), pasta consumed in Italia (Aldaya & Hoekstra, 2010), dairy products from Australia (Ridoutt et al., 2010a), Italian bottled water (Niccolucci et al., 2011), Spanish olives and olive oil (Salmoral et al., 2011a), cut flowers from Kenya (Mekonnen et al., 2012), New-Zealand’s wine (Herath et al., 2013), a pair of jeans and cotton produced in Spain (Chico et al., 2013), or waste associated to fresh mango in Australia (Ridoutt et al., 2010b).
    - *At the world scale*: cotton (Chapagain et al., 2005), wheat (Mekonnen & Hoekstra, 2010a), rice (Chapagain & Hoekstra, 2011), paper (Van Oel & Hoekstra, 2011), or a list of crops and derived crop products (Mekonnen et al., 2011) and farm animal products (Mekonnen & Hoekstra, 2012a).
- A consumption pattern: different diets in China (Liu & Savenije, 2008) and in the EU (Vanham et al., 2013).

[34] All these studies do not introduce a definition of the WF in line with the framework of the Water Footprint Network (Hoekstra et al., 2011) and some aim at comparing different approaches (e.g. Ridoutt et al., 2010a, 2010b, 2012; Herath et al., 2011, 2013; Zonderland-Thomassen & Ledgard, 2012, see Section 3.3 on the comparison of volumetric and impact-weighted WFs). However, they all present a quantification of water resources consumption from the perspective of consumers or of a product.

- A whole sector or industrial process, such as bioenergy (Dominguez-Faus et al., 2009; Gerbens-Leenes et al., 2009a), hydropower (Herath et al., 2011; Mekonnen & Hoekstra, 2012b), biofuel-based transport (Gerbens-Leenes & Hoekstra, 2011), livestock production systems in Australia (Ridoutt et al., 2012), dairy farming in New-Zealand (Zonderland-Thomassen & Ledgard, 2012), a wastewater treatment plant in China (Shao & Chen, 2013) or tourism in Spain (Cazcarro et al., 2014).

In a view based on the aggregation of the consumption of resources along the supply chain, companies are a key link (Figure 5.2). It means that, in addition to the role of consumers or the issue of sustainability of the consumption pattern, the approach in terms of footprint has been especially propitious to underline the role of companies in providing sustainable products (Gerbens-Leenes & Hoekstra, 2008). Many businesses have engaged in footprint assessments for corporate environmental responsibility, faced with consumer demand for more sustainable products. Nevertheless, this may be only one of the reasons for undertaking a footprint assessment. For instance, WF assessments can help companies to identify water-related risks in their supply chain (SABMiller & WWF-UK, 2009).

The interest from this sector is shown by the studies dealing with corporate products, e.g. a pasta sauce and chocolate candy (Ridoutt et al., 2009; Ridoutt & Pfister, 2010), a carbonated beverage (Ercin et al., 2010), tea and margarine (Jefferies et al., 2012), pasta production (Ruini et al., 2013) or a soap bar (Francke & Castro, 2013). These examples are from the peer-reviewed literature; many other initiatives are reported by the Water Footprint Network (2013), particularly from the year 2009.

V.1.4 Water footprint assessment of production within a geographical area

The second type of WF assessment is based on aggregating footprints from productive processes occurring at the scale of a geographical area. It allows assessing sustainability of the use of internal resources of a region. Regarding water, this perspective is especially meaningful when the area considered is a river or aquifer basin, as this is the space where water resources are physically connected. Various studies have been published under this perspective: Aldaya & Llamas (2008) have considered the case of the Guadiana River Basin (Spain), de Miguel et al. (2012), the Duero River Basin (Spain), and Zeng et al. (2012), the Heihe River Basin (China).

The global studies by Mekonnen & Hoekstra (2011) and Hoekstra & Mekonnen (2012a), applying a global agro-hydrological model for the agricultural WF, completed by other sources for the other sectors, provide results of WFs at the scale of the river basin. These have been used to provide further analysis, e.g. water scarcity indexes at the river basin scale (Hoekstra et al., 2012), or as an input for other studies considering specifically this scale (Vanham, 2013; Ercin et al., 2013). These results should, however, be taken with care since world scale models may lack of accuracy and local data should be preferentially used.

By focusing on water resources having their origin within a river basin, these studies may seem to have a more conventional approach. Yet they differ from traditional water management since they contemplate a specific measure for blue water accounting and introduce green water in the analysis (see Sections 2.2. and 2.3 of this chapter).

V.1.5 Putting the water footprint into perspective: footprints at multiple scale

Approaches in terms of footprint are not limited to the field of water resources management. In particular, the ecological footprint took hold before the WF and the author and developer of the WF associates the formulation of the WF concept to the ecological footprint methodology (Hoekstra, 2009). The objective of this section is to present the concept of footprint as a measure of the impacts from human activities on the environment, especially from resource consumption and contamination, under multiple perspectives and scales, in order to put the debates around the WF approach into perspective.

First, the idea of reporting the consequences of human production and consumption of goods on the environment through the concept of a footprint implies a different perspective compared to previous approaches. The originality of the footprint approach is that it both goes beyond and integrates traditional perspectives. By tracing the impacts of the consumption of products all along the supply chain, these are no longer contemplated as a local issue and the equity of the appropriation of resources is not only related to the materials that are effectively traded but also to their ‘virtual’ content in terms of resources and contaminants. Impacts on natural resources and the environment from the consumption and production of goods can be aggregated according to different scales and perspectives (consumption or production) and potentially attributed to any entity in the supply chain or considered as embedded in the products. This view establishes a direct connection between the product and the impacts it has generated.

As shown on Figure 5.1 for the case of the WF, the method for calculating a footprint – at any scale and under the perspective of both consumption and production – consists in two steps (Figure 5.3). The first step is the quantification of the direct consumption of the resource or the emission of the contaminant in a basic process. What is essential in this step is the formulation of an indicator of the consumption or contamination.

The second step contemplates the aggregation of the material flows of the produced goods. In the consumption perspective, it relates to their destination, like an industry, another country or the final consumer. Under the production perspective, the flows of goods and services generated in a defined geographical area are aggregated. Data relative to this step can be for instance the Input-Output tables of the material flows in and out a region or a country (Duarte & Yang, 2011; Duarte et al., 2002). According to Yang et al. (2013):

> "An [Input-Output] table/model represents the monetary transactions of goods and services among different sectors of economic system. It provides a technique to specify how the substances flow among sectors through supplying inputs (including water) for the outputs (where [virtual water] is embedded) in the economic system."

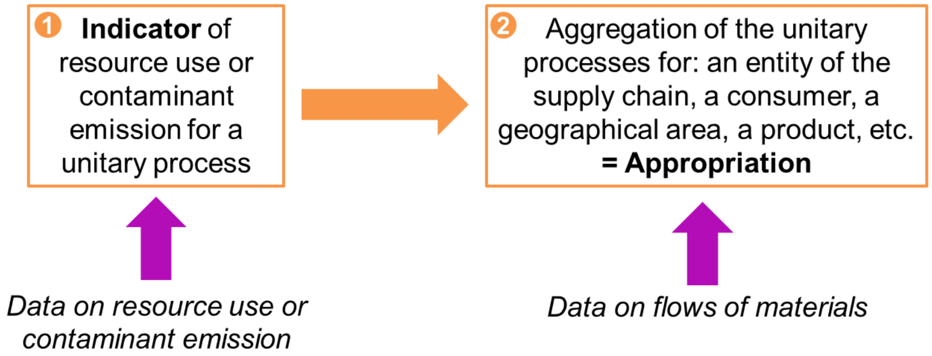

Figure 5.3. The two steps of a footprint assessment and related data.

Depending on the scale and perspective, the selection and aggregation of material flows can be based on a variety of data, according to the scope of the study, e.g. exports or imports of a product or sector, or material flows to or from a region.

The description of the flows of materials linked to the production of goods and services and their impacts is also essential in life cycle analysis (Guinée & Heijungs, 2005), industrial ecology (Jelinski et al., 1992) and ecological economics (Costanza et al., 1997)[35]. Social or Societal Metabolism (Giampietro & Mayumi, 2000) and Material Flows Analysis (Haberl et al., 2004) are also relevant. Thus, a WF approach can also be related to these fields. This integrated approach has been popularized under the term 'footprint' but other terminologies, some of them pre-existing, refer to a similar conception, e.g. 'embodied resource', 'virtual resource' or 'appropriation', in order to link production and consumption to the requirements in terms of materials or ecosystem services or consequences in terms of waste generated (Table 5.1)[36].

There is, however, no uniformity in the formulation, application and implications of these indicators. In particular, the direct comparison of footprints for different products depends on

[35] The number of papers published by the journal 'Ecological Economics (e.g. Chapagain & Hoekstra, 2007, 2011; Hoekstra, 2009; Aldaya et al., 2010a; Zhang et al., 2011; Ercin et al., 2013) is also an evidence for the interest of a WF approach in the field of ecological economics.

[36] The ecological footprint is expressed in units of area (ha, km^2). This spatial dimension reinforced the metaphor with a footprint. This view has been lost in the development of other types of footprint.

the local or global nature of the environmental impact. For instance, the carbon footprint of two products can be compared because the impacts of greenhouse gas emissions are assumed to be the same for the climate, wherever the emission takes place.

Recently the concept of a 'footprint family' has been mentioned in various publications (Galli et al., 2012; Fang et al., 2014). They specifically consider the ecological, water and carbon footprints – and energy footprint in the case of Fang et al. (2014). These are the three types that have gained major popularity in relation with the term 'footprint'. However, the list shown in Table 5.1, where indicators sometimes overlap, could be extended to take into account virtually all the contaminating and resource consuming processes.

Table 5.1. Examples of footprints and terms linked to a footprint approach.

Term	Definition
Embodied energy	"(...) the total (direct and indirect) energy required for the production of economic or environmental goods and services" (Costanza, 1980)
Ecological footprint	"(...) accounts for the flows of energy and matter to and from any defined economy and converts these into the corresponding land/water area required from nature to support these flows" (Wackernagel & Rees, 1996)
Human appropriation of net primary production	"The human appropriation of net primary production (NPP) (…) [is] defined as the difference between the NPP of the hypothetical undisturbed vegetation and the amount of biomass currently available in ecological cycles" (Haberl, 1997)
Carbon footprint	"The carbon footprint is a measure of the exclusive total amount of carbon dioxide emissions that is directly and indirectly caused by an activity or is accumulated over the life stages of a product" (Wiedmann & Minx, 2007)
Land (use) footprint	"A LCA-based land use footprint indicator could help in understanding the incremental pressure on land resources of agri-food production systems and consumption patterns, and enable the assessment of tradeoffs with greenhouse gas emissions, water use impacts and other relevant environmental burdens." (Ridoutt et al., 2014); see also Wilting & Vringer (2009), Yang et al. (2009) and Fader et al. (2011)
Nitrogen footprint	"(…) an N footprint [is defined] as the total amount of N_r [reactive nitrogen, all nitrogen species except N_2] that is lost to the environment due to individual's consumption of food and energy" (Leach et al., 2012)
Material footprint	"Because of its analogy to other footprint indicators (…), we suggest using the term "material footprint" (MF) (…) and define it as the global allocation of used raw material extraction to the final demand of an economy" (Wiedmann et al., 2013)

In addition to the issue of the environmental impacts of the production of goods, the specific debate in relation to the import of virtual water to alleviate water scarcity in the importing country is also found in relation to the other footprints. For instance, import of industrial goods can reduce the contamination in the country of destination to meet environmental standards.

V.2 Blue and green water accounting

V.2.1 Formulation of an indicator of resources consumption or contaminant emission

As presented in Figure 5.3, a basic input of a footprint approach is the quantification of a resource use or contaminants emission. In many cases, this is quite a direct step for two reasons:

- Only humans 'use' the resource or emit contaminants, which means there is no debate on the relative share attributable to humans.
- The whole amount of resource taken from the environment can be attributed, i.e. there is no possible re-use of the initial withdrawals.

In the case of water resources, the accounting of resource appropriation is more complex. For green water (see Section 2.3 of this chapter) and in-stream water, the 'use' is shared between human activities and the environment, for instance a forest is a natural ecosystem and a source of materials, a river is an ecosystem and can be navigated. In the case blue of water use, the issue of the reusability of return flows is essential to account for the 'appropriation' of water resources by humans (see next section).

V.2.2 Blue water accounting

With respect to blue water – i.e. the water available in rivers, lakes and aquifers – only the resources that are not available again for further use within the boundaries of the system under study to satisfy other users' demand or the environment are considered as appropriated by the considered use (Hoekstra et al., 2011). This relates with the traditional distinction between consumptive use and total use or withdrawals[37]. Water resources management has traditionally focused on the total use and demands are usually contemplated in terms of total withdrawals. In the WF approach, computing exclusively water that is not returned is viewed as more adequate in terms of 'appropriation'. The share that is used again downstream is not 'appropriated'. It has been claimed therefore that a WF is a more correct indicator than total use, which implies a new vision for water resources management (Hoekstra et al., 2011).

Consequently, the approach of the WF in terms of 'appropriation' is two-fold: first, it is linked to the concept of footprint, which considers multiple scales and perspectives of aggregation of the consumed water resources; second, it considers a specific physical indicator of water use that intends at accounting for the water really appropriated by humans (Figure 5.4).

[37] This point is specifically developed for groundwater in Chapter 6 on methods for the case studies.

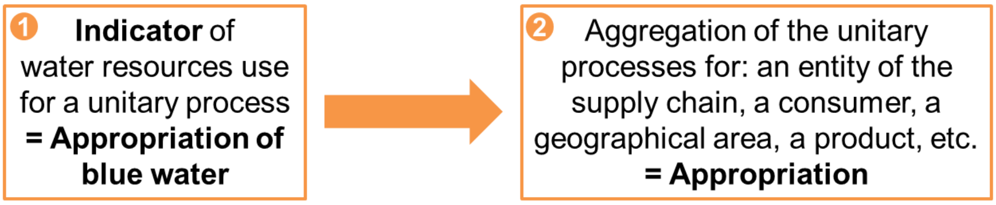

Figure 5.4. Two-fold dimension of the water footprint in terms of 'appropriation'.
(1) a specific indicator of water use and (2) the aggregation of virtual water flows according to multiple scales and perspectives.

The affirmation that the focus on total withdrawals in water resources planning can be misleading is not specific to the WF approach. Focusing on the consumptive use has been repeatedly advocated, particularly in relation with current discussions on efficiency of water resources use or demand management. A more efficient use or a reduction in demand can imply a reduction in total use or withdrawals, while the consumptive use (i.e. the WF) remains the same. This point was initially discussed in the field of irrigation sciences (Willardson et al., 1994; Burt et al., 1997). However, these questions and the associated methodologies of water accounting have been extended to all water uses (see Seckler, 1996; Molden et al., 2003; Perry, 2007, 2011, among others). For instance, Willardson et al. (1994) introduced a method based on the accounting of consumed, reusable and non-reusable fractions of the water use. In the framework developed by Molden (1997) and in subsequent publications (Molden & Sakthivadivel, 1999; Molden et al., 2001, 2003), the equivalent indicator to the WF is called 'water depletion' (Figure 5.5)[38].

These water accounting frameworks are based on the characterization of a series of fractions. The change in all these fractions should be assessed when a shift in water use is proposed (e.g. allocation to another user or a change in 'efficiency'). Contemplating the consumed or depleted fraction is a first step towards a more comprehensive view of the multiple indicators that are useful for water resources management. The approach of the WF contributes to this view by accounting only the water 'appropriated' for a specific use. A detailed water accounting based on these approaches has also recently been mentioned in a report by FAO (2012) and for the European Commission (Bio Intelligence Service, 2012). Various water resources assessments at basin scale have been published following these water accounting frameworks and focus on consumptive use (e.g. Karimov et al., 2012; Wu et al., 2014). Consequently, they can be considered as equivalent to WF assessments at basin scale.

[38] Perry (2011) refers to 'water consumption', even if it can be questionable since this is a more generic term that could be more easily used as a synonym for withdrawals, demand or use, which are terms that are often used interchangeably.

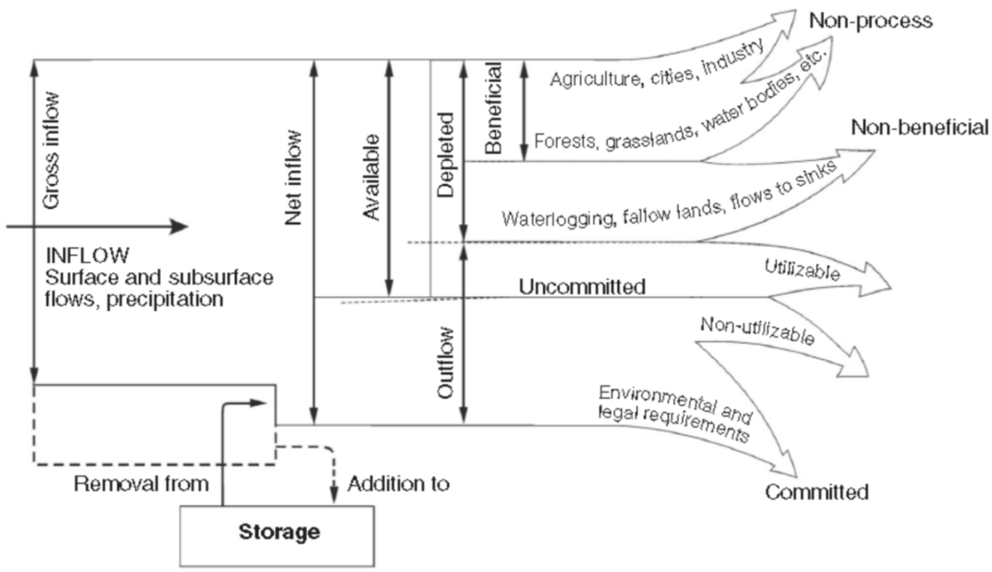

Figure 5.5. Water-accounting diagram, applicable to basin analysis and also at other scales. Source: Molden et al. (2003).
Process depletion is the water depleted in the intended direct use; non-process depletion is the depletion that simultaneously takes place, which can be beneficial (e.g. flow to a valuable wetland) or not (e.g. water logging); the share going back to the river basin integrates flows that are committed to downstream users and flows that are uncommitted. This framework was already presented by Molden (1997).

An indicator based on the consumptive use, i.e. the WF, may be in fact more relevant for a series of issues; however, in many situations it is important to address also total withdrawals. The WF and total withdrawals should be used together. In many cases it is desirable to reduce withdrawals in order to maintain sufficient flow in a river stretch in relation to specific issues – e.g. to reduce environmental damages, to dilute a local contamination, or to improve hydropower generation. However, this will not be accounted as a reduction of WF. Even if they were initially developed to warn on the importance to consider the ‘consumptive use’, the water accounting methods described above present the whole picture of the destination of the flows and this water accounting should be promoted to take informed decisions.

While the WF could contribute to the debate on the selection of the adequate indicator for water resources management, the majority of WF assessments do not focus on the water accounting phase and few studies include these debates (e.g. Jefferies et al., 2012). No direct link has been established with previously existing methods for water accounting in the debates on the WF formulation. Moreover, with few exceptions, the methods traditionally used in agricultural WF assessments are paradoxically not appropriate to quantify the actual consumptive use because of a mistaken assumption, which implies they quantify the theoretical crop evapotranspiration demand[39]. Additionally, many assessments that focus on the aggregation of virtual water flows in the economy, mainly through Input-Output analysis, do not consider adequately the

[39] For a more detailed discussion see Chapter 6 on methods and data for the case studies.

'appropriation from a water accounting perspective' (Figure 5.4) and introduce total withdrawals. On the other hand, the proponents of detailed water accounting have not identified the WF as an indicator that was similar to the 'depleted' or 'consumed fraction' and that could contribute to communicate the importance of undertaking a detailed water accounting.

V.2.3 The water footprint and the green water metaphor

Water resources management has traditionally focused on blue water. However, green water, the share of rainfall that can be used by plants to satisfy their evapotranspiration requirements directly, can bring out new perspectives, particularly in relation to ecosystem service provision, particularly food production, and land use (Rockström & Gordon, 2001).

First, the green WF allows debating the repartition of land use between the environment and human activities. An index of green water scarcity has been proposed by Hoekstra et al. (2011). This index would consider the share of green WF on the total green water 'available', in the same way a blue water scarcity index can be formulated based on the ratio between withdrawals and blue water availability. Nevertheless, these questions are linked to land use issues, which have already received much attention, and it has been argued that referring directly to land use may be clearer than the metaphor of green water (Ridoutt & Pfister, 2010; Yang et al., 2013).

Second, in the agricultural sector, yields could increase thanks to a better management of green water through improved agricultural practices that allow conserving soil humidity. This contrasts with the traditional approach of applying blue water (irrigation) as the solution to increase yield (Rockström et al., 2009). Hence, food production could be significantly increased by turning unproductive green and blue water flows to productive water (Falkenmark & Rockström, 2006; Rockström et al., 2009; Foley et al., 2011).

The quantification of green WFs has contributed to raise awareness on the role of green water. Green WF assessments often recommend improving green water productivity, i.e. turning more green water consumption towards the crop photosynthesis. Different techniques are available to this end, such as soil preparation (no tillage) or using supplementary irrigation in order to better benefit from rainfall (Rockström et al., 2009). Rainwater harvesting is also a way to avoid evaporation from soil. However, the availability of blue water can be affected.

WF assessments usually do not consider this level of detail. They call for an increase in green water productivity. According to Hoekstra & Mekonnen (2012b):

> "An important component of the solution to overexploitation of blue freshwater resources in water-stressed catchments is to increase water productivities (reduce product WFs) in water abundant areas (Hoekstra et al., 2011). Particularly the efficient use of the world's green water resources in rain-fed agriculture can help to reduce the need to consume blue water resources (Falkenmark & Rockström, 2004).

A mere focus on reducing WFs in water-stressed catchments displays a limited perspective on the question of what is globally sustainable and efficient water use."

V.3 Water footprint interpretation

V.3.1 A preliminary note on the interpretation of volumetric water footprints

A main critique of the WF expressed as a volume of green and blue water is that a volumetric approach cannot be used to directly assess the environmental impacts and guide decisions. A high WF in an area where water is abundant may be less worrying than a smaller WF exerted where water is scarce. Consuming green water or blue water also has different implications. It can be questionable therefore to add these two components. When the grey WF is considered, it can be even more problematic to present an aggregated WF.

Indeed it is clear that the final environmental impact is not known from a volumetric WF. It is recognized by Hoekstra et al. (2011) as they insist on the fact that a 'sustainability assessment' and a 'response formulation' are necessary. Yet the most common output of WF assessments is to present volumetric figures of green, blue and grey WFs. This situation should be analysed differently depending on whether the results are used to inform the general public or for effective decision based on a detailed technical assessment. In the first case, WFs have benefits in terms of raising awareness, as they show the implications from the consumption of products in terms of water resources appropriation and the fact that the amount of water to sustain our consumption pattern, particularly food, is high. Some other issues can be highlighted for awareness raising, such as the contribution of green water and its relation with land use. A main question is if the risk of misleading interpretation is not too high. On the other hand, the aim of this section is to review the interpretation and relevance of volumetric and impact-weighted WFs to guide effective decisions within a deeper analysis that would be aware of the 'traps' in their interpretation.

V.3.2 Blue water footprint versus green water footprint impacts

One of the main discussions in the interpretation of WFs is the relative environmental impacts of green and blue water use. It has been argued that green water consumption is mainly linked to land use issues, with usually little competitive uses (i.e. a limited opportunity cost, see Aldaya et al., 2010a), while blue water consumption can have high environmental impacts if ecological flows are infringed and is a competitive use in many cases. Nevertheless, the access to green water through the conversion of forests to fields to produce export-oriented crops, a phenomenon that has been happening over the last decades in many places illustrate that the environmental cost of green water consumption can be high (Willaarts et al., 2011).

There is no direct conclusion on whether green water or blue water consumption generates less environmental impact as a general rule. These two indicators should be considered separately. It has been proposed that environmental impacts associated with green water should be integrated under a land use footprint (Ridoutt et al., 2014). It would represent a change of focus, which can be justified depending on the scope of the assessment. Introducing a green WF can be useful and informative relative to a series of issues introduced in Section 2.3 of this chapter.

In fact, debates on the relative implications of blue and green water consumption mainly depend on the scope of the study. Assessing the environmental impacts of a product is not the same as considering the role of green and blue water relative to the virtual water imports. In this latter case, the view is that virtual water imports, usually coming from humid countries, i.e. green water, allow the countries lacking water resources to produce enough food for their population to solve this problem. While this can be an observed situation, the debate on whether this should be a strategy to relieve the pressure on blue water resources is unresolved, and beyond the scope of this thesis. It could be added, however, that this view usually focuses on the problem of blue water scarcity and masks other measures such as reducing food losses, changing the consumption pattern or improving green water productivity (Rockström et al., 2009).

V.3.3 Blue water impacts and response: impact-weighted and volumetric water footprints

- <u>Integrating the local damages: an impact-weighted blue water footprint?</u>

When considering solely blue water, a common recommendation is to include a scarcity indicator to weigh water consumption according to local conditions (Pfister et al., 2009; Ridoutt & Pfister, 2010, 2012; Berger & Finkbeiner, 2012; Bayart et al., 2014). This is based on the view that the same volume of water consumption causes less damage in a place where water resources are plentiful as compared to a place where these are scarce. The WF should therefore consist of a weighted indicator, and not in a volumetric indicator, in order to directly compare the consequences of product consumption on local water resources. According to Pfister & Hellweg (2009), the WF presented by Hoekstra and colleagues is a ‘water shoe size’ and not a ‘water footprint’. The water scarcity factor is usually expressed in terms of water consumption over total water availability and different procedures have been proposed to combine the volumetric and scarcity indicators (e.g. Pfister et al., 2009).

A response to these critiques by the proponents of a volumetric WF is that the presentation of a weighted WF is not as clear as a water volume that has been effectively consumed. Answering to Pfister & Hellweg (2009) and their ‘shoesize’ metaphor, Hoekstra et al. (2009) affirm that the value of the water consumed is a real data that can be used for water resources management.

- <u>Water footprint and life cycle assessment approaches</u>

Some authors who proposed impact-weighted indicators formulate their critique from the LCA approach. A detailed consideration of the quantitative aspects from water use is relatively new in LCA, which has focused traditionally on qualitative aspects (Kounina et al., 2012; Boulay et al., 2013). A series of papers have recently dealt with the inclusion of water use in LCA, sometimes including a comparison with the WF indicator (Koehler, 2008; Pfister et al., 2009; Jefferies et al., 2012; Berger & Finkbeiner, 2010; Bayart et al., 2010, among others).

The LCA approach focuses on the environmental impacts of the supply chain of a product 'from cradle to grave' on the basis of a series of indicators (Guinée & Heijungs, 2005). The impacts on 'human health', 'ecosystem quality' and 'resources' should be ultimately assessed ('end-point' LCA). The link between water resources consumption and these ultimate goals is not direct and a series of intermediary indicators can be elaborated. This 'mid-point' LCA can be sufficient depending on the scope of the study. Thus, main issues for the inclusion of water consumption in LCA are the formulation of 'mid-point' indicators, on the one hand, and their transfer to the 'end-point' approach, on the other hand (see Kounina et al., 2012).

The LCA framework is based on four phases: goal and scope definition, inventory analysis, impact assessment and interpretation (Figure 5.6).

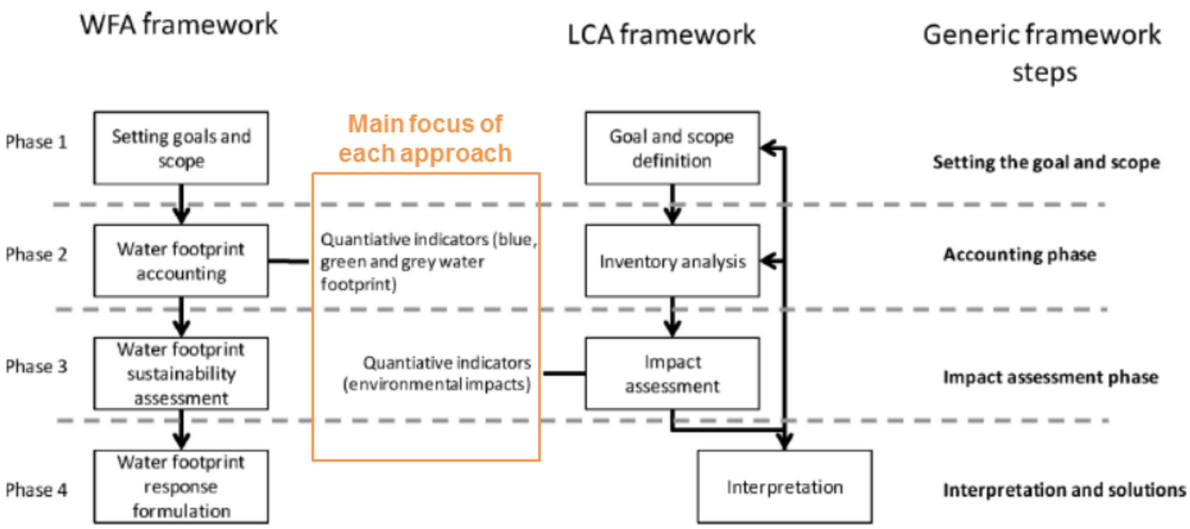

Figure 5.6. Comparison of the life cycle assessment and water footprint frameworks. Source: adapted from Boulay et al. (2013).

This structured approach aims at having a unified framework in order to be able to compare products and having a systematized approach for the verification and robustness of the assessment. It constitutes the framework for the standard ISO 14040:2006 'Environmental management – Life cycle assessment – Principles and framework' (ISO, 2006). The approach of the WF by the Water footprint network (Hoekstra et al., 2011) includes also a structured framework for WF assessments[40].

The WF and LCA frameworks are comparable as they include the same basic steps (Figure 5.6). The main difference is that the WF assessment focuses on the accounting of the water consumption, which is the main measure that should be reported (and interpreted), while the LCA formulates indicators in terms of impacts (Boulay et al., 2013). It should be noticed that the LCA is currently mainly applied to products and supply chains. It is on this aspect that it could be compared with the WF approach, which has itself other applications.

Under the LCA approach, these are the final impacts that should be interpreted. The main critique is that the WF in terms of volume is misleading as it does not consider the actual environmental impact, based on the observation that this impact is stronger when water is scarce than when it is plentiful. The question whether the issue of green water consumption should be included as such in the LCA (e.g. Núñez et al. 2013) or contemplated within the perspective of land use (Ridoutt et al., 2014) is not resolved.

- Is the debate between volumetric and impact-weighted water footprints relevant?

This section will show the main arguments presented under the two approaches to argue that in reality their specificity is questionable.

The WF approach by the Water Footprint Network includes a ‘sustainability assessment’ that integrates environmental, social and economic dimensions, particularly in terms of efficiency and equity (Hoekstra et al., 2011). A main argument for not formulating an indicator focused on the final environmental impacts is that a volumetric WF serves also for the economic and social assessment contrary to weighted WFs and the LCA that focus on the environmental dimension.

Nevertheless, the critique on the interpretation of volumetric WFs is justified in many cases: numerous WF assessments directly compare the WF in terms of volume to make consumption choices, particularly in the first studies (e.g. Chapagain & Hoekstra, 2007). The integration of social and economic dimensions often remains a declaration of intent as the sustainability assessment phase of the WF presented by Hoekstra et al. (2011) is little elaborated and very few WF assessments actually integrate these aspects. No detailed approach is proposed to combine

[40]This framework tends to be considered as the method for WF assessments, which should include explicitly these four successive steps. This kind of structured presentation can be useful to have an harmonized approach in certain contexts, with the aim to establish an official standard on the WF, like the four phases of LCA in the ISO 14040:2006 standard, or to insist on the fact that the WF quantification should not be considered alone but should be interpreted in a specific context. However, highlighting these different steps may appear as rigid. The WF accounting can be used in studies with their own goals, objectives, organization and presentation.

the different dimensions and to guide the response formulation of the calculated WFs. In spite of the claim that a WF assessment is not limited to WF accounting, the absence of sustainability assessments is the rule, including for the environmental aspects, where final environmental impacts are not discussed, and misleading interpretations of WFs are common.

In fact, the debate on whether the main result should be the volume or the final impact may be not essential. The assessment of the final environmental impacts is included in both approaches. It is the final indicator for a weighted WF, while it is theoretically part of a 'sustainability assessment' in the WF approach. For instance, Hoekstra et al. (2012) present an assessment of the environmental sustainability of river basins WFs, comparing them with the monthly water availability. Both approaches therefore contribute to the debate on water resources management associated with the supply chain of a product, and it could be expected that the conclusions and recommendations, the last step in both approaches, would produce similarly useful recommendations to improve the state of water resources.

The main critique from the proponents of a weighted WF is that it is more relevant to assess the real impacts of water consumption. Yet, there are challenges in considering a water stress index. For instance, water taken during low flow periods from a highly regulated river basin with dams to make water available during summer may have less impact than from an unregulated basin. The inclusion of ecological flows or the issue of scale represent also a challenge.

Thus, both the uncertainties and the difficulty for formulating local water stress indicators and integrating them in a weighted WF imply that doubts can also be raised on the direct interpretation of weighted WFs. A single indicator only represents part of the issues and it is problematic if it is interpreted too literally. The complexity of water use in terms of economic activity or environment means that it is questionable to favour a product based only on its weighted WF. Discarding a product based on a higher WF does not imply that water resources consumption will decrease since, as stressed by Yang et al. (2013), it is probable that another use will replace the use that is avoided. The main response could be working toward the reduction of the WF, which means decreasing the volume of water consumed[41]. It may be more meaningful to present a water stress indicator and water consumption separated, to understand more directly the reason for the environmental impacts.

The options to reduce a weighted WF are probably similar to a volumetric WF. Reducing a WF is the same as increasing the productivity of water. While the debate on water productivity are

[41] A very high weighted WF, i.e. a high water stress index, can raise awareness on the environmental impacts and consumers can have a role through a change in their choices. While this might be possible in extreme situation, it may be problematic to interpret a weighted WF in a general case.

often limited to blue water, one of the main messages in the response formulation of WF assessments is that green water productivity should also be improved (Hoekstra et al., 2011)[42]. Solutions include raising yields or reducing unproductive evapotranspiration. Agronomists and irrigation specialists have been working on these issues for decades (Molden et al., 2010), and the solutions proposed in WF assessments often appear general and theoretical.

Contemplating the economic and equity dimensions of a WF implies also that a WF sustainability assessment should include a number of issues relative to water resources management in the area of concern. Thus, in terms of response formulation, there is a need for integrating existing knowledge and methods in the WF approach. However, due to their different initial approach, the proponents of the WF sometimes reconsider traditional debates – e.g. formulation of water stress indicators, interpretation of water productivity, or definition of available resources – sometimes questionably, as if they were new issues.

V.3.4 A specific interpretation of the water footprint: equity

The equity aspects of WFs interpretation are often contemplated in WFs assessment. As explained by Hoekstra & Mekonnen (2012b):

> "WFs need to be seen from the perspective of equity as well. By comparing the WF of consumption for different nations, we showed that some people consume and pollute more freshwater than others (Hoekstra & Mekonnen, 2012a). The fact that US consumers have a WF per capita 2.6 times larger than people in China and India justifies a debate about equitable appropriation of freshwater resources. The world's spatially distributed freshwater resources are accessible from anywhere through trade in water-intensive commodities. The widespread inefficient use, overexploitation, and pollution of water must be a concern for all that have a water-intensive consumption pattern, not only for those that directly depend on the areas where environmental impact of water use is greatest."

This equity issue is directly linked to the approach of WFs from the perspective of consumers. One of the main advances of the WF indicator is to have contributed to reveal the link between the consumption of a product and the implications in terms of water use, potentially in another place of the world. The higher WF of some countries is linked to their consumption pattern, including their diet, and it is also one of the merits of the WF to have stressed the impacts the way of living have on water resources. Yet this remains a conceptual discussion, maybe useful only for awareness raising, since it is not evident, for instance, to establish caps on the WFs of individuals or countries. Focusing on the regulation of only one impact or footprint, among all the footprints, does not appear to be practical. Regarding the consumption level, other actions,

[42] An increase in green water productivity should allow using less blue water (Hoekstra et al., 2011). However, this is a very simple and direct interpretation which is questionable, since these productivity improvements can have a full range of consequences, for instance on prices or consumption, in a complex system of food production at the world scale.

such as food waste reduction, appear to be more realistic. A view in terms of footprints, particularly the WF, has helped raise awareness on the issue of food waste (FAO, 2013).

V.4 Economic value of water in water footprint assessments

According to Hoesktra et al. (2011), in addition to the environmental and social aspects, a sustainability assessment of a WF (see Figure 5.6) should contemplate an economic assessment. There is, however, no robust methodology to assess this dimension. Nevertheless, the economic aspect has been considered by some WF studies. They do not necessarily refer to an explicit 'sustainability assessment' as introduced by Hoekstra et al. (2011). It is ascribed to their own approach of the WF. The following studies complement the WF accounting of the agricultural sector with a water productivity indicator: Aldaya & Llamas (2008) (Guadiana River Basin), Aldaya et al. (2010b) (La Mancha Agricultural Region), Garrido et al. (2010) (Spain) and Salmoral et al. (2011a) (Spanish olives and olive oil). In these studies, the water productivity (or apparent water productivity) is considered as the market value of the production divided by the WF of the different crops. According to Garrido et al. (2010), this contributes to:

> "going one step further in virtual-water studies and adding a new economic dimension to the previous estimates."

The formulation of economic productivity indicators is not a specificity of a WF assessment as they are commonly used in water resources management (see Hellegers et al., 2009; Berbel et al., 2011; Perry et al., 2009). However, combining the WF approach highlights some important points relative to the water accounting as an input data for the economic productivity as will be described in Chapter 6 on methodology and data for the case studies.

V.5 Toward the 'Integral Water Footprint'

V.5.1 Waste and efficiency in the supply chain

Along the supply chain of a product many losses of materials occur. It might be inevitable in the processes of transforming raw materials into consumer goods. However, these processes may not always be fully efficient and other avoidable wastes are generated along the supply chain[43]. In the food sector, waste can be generated by final consumers or retailers or during transport and storage, if these are realized in bad conditions (Lundqvist et al., 2008; FAO, 2011; Zimmer,

[43] Even though the framework presented here is initially built on the WF, it is applicable for all footprints.

2013)[44]. Any stage of the supply chain virtually generates losses, which have also an associated WF (Figure 5.7).

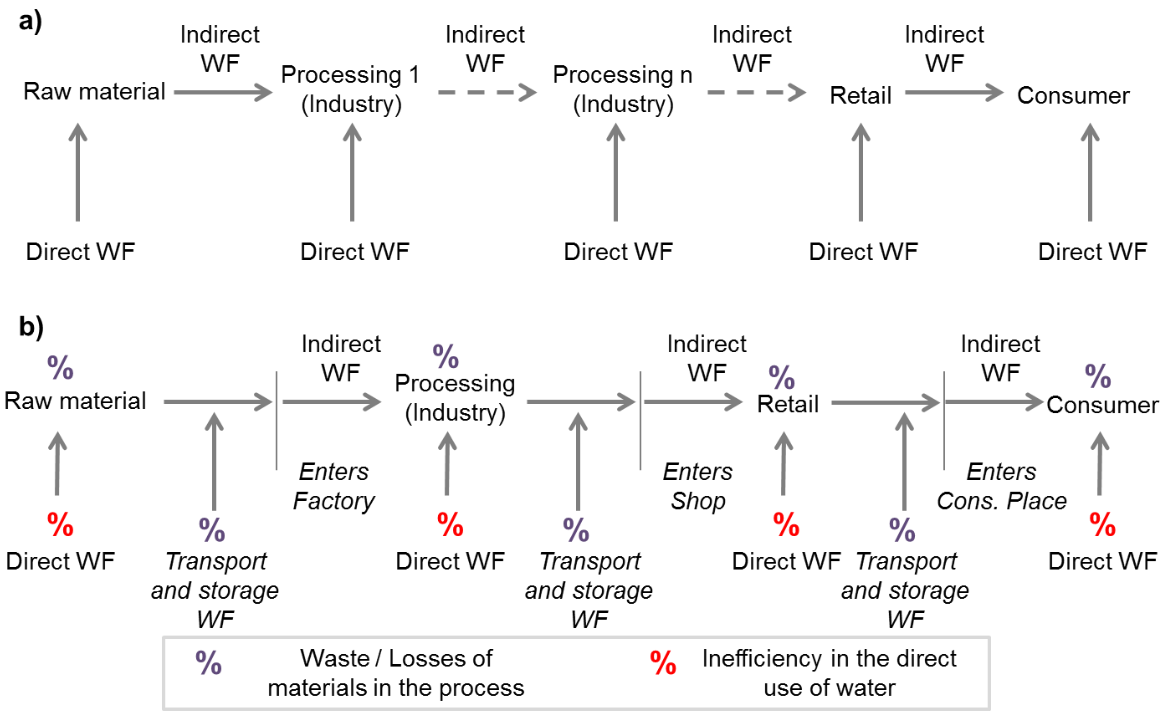

Figure 5.7. Direct and indirect water footprints along the supply chain: (a) traditional view; (b) with transport and storage and identifying losses and inefficiencies. Source: own elaboration based on Hoekstra et al. (2011) for a

Thus, a product delivered to the next step to be processed or consumed has implied a higher pressure on water resources (WF_{real}) as compared to the ideal situation without losses, which is usually considered (WF_{ideal})[45]. With *w* the percentage of losses, WF_{real} is given by:

[44] In a context of globalized chains, the footprint linked to the transport of goods and raw materials to the place of transformation and consumption should be contemplated, mainly because moving goods from a place to another requires energy. Thus, the footprint of a good is constantly changing: each additional kilometre adds the footprint of the consumed energy. In the case of the WF, this contribution is limited and may be considered irrelevant compared to the total product WF. This might, however, not always be the case and the rising contribution from biofuels for transport implies that it may become a major factor of water resources consumption in the coming years (Gerbens-Leenes et al., 2009; Velázquez et al., 2010). In the same way, to make a product available, it is often necessary to store it at some point of the supply chain. In the case of the food sector, energy is usually necessary to maintain good conservation conditions. Therefore the footprint associated to energy should be also taken into account.

[45] Losses can be particularly generated during transport and storage. This explains why it is important to take into account these two steps (more than for the additional WF they directly generate). It is also noteworthy that Zimmer & Renault (2003) had already introduced the concept of 'consumption efficiency' related to the fact that "production at the farm gate does not entirely convert into consumption because of various wastes before reaching domestic consumption, and also the process itself of food consumption generates its own waste. This is particularly true for fresh products (vegetables fruits) which are sensitive for conservation."

$$\mathrm{WF_{real}} = {}^{\mathrm{WF_{ideal}}}\!/_{(1 - \mathrm{w}/100)} \qquad (5.1)$$

Waste or loss of materials and the inefficiency of the processes that directly use water are different factors (Figure 5.7). In the latter case, the WF resulting from a direct water use is usually correctly computed since the inefficiency does not mean a corrective factor must be introduced. Nevertheless both factors highlight potential targets to reduce the actual WF of a product: reducing losses, through a higher efficiency in the transfer of materials to save their WF, and improving the efficiency of the processes using water directly from the environment.

V.5.2 'Downstream water footprint' and responsibilities in the supply chain

The WF approach identifies new responsibilities along the supply chain since the choices of an entity, such as a consumer or a company, have implications not only for its direct water use but also for the indirect water use. Then the focus is on the water that has been successively consumed to make available the goods used by this entity. An extension of this scope would consider also the water consumption generated by the use or further processing of products leaving the entity. Indeed subsequent, or 'downstream', uses of a product often imply additional water consumption, due to the successive direct water uses or the WF linked to waste during further processing or transport and storage, as explained in the previous section (Figure 5.7).

Thus, this 'downstream WF' could be added to the traditional WF, which focuses on the direct and indirect water consumption before the product reaches an entity, i.e. the 'upstream WF' (Figure 5.8). It would constitute the 'Integral Water Footprint'. This indicator would account for the whole water consumption in the life cycle of the product:

$$\textit{Integral Water Footprint} = \textit{upstream WF} + \textit{downstream WF} \qquad (5.2)$$

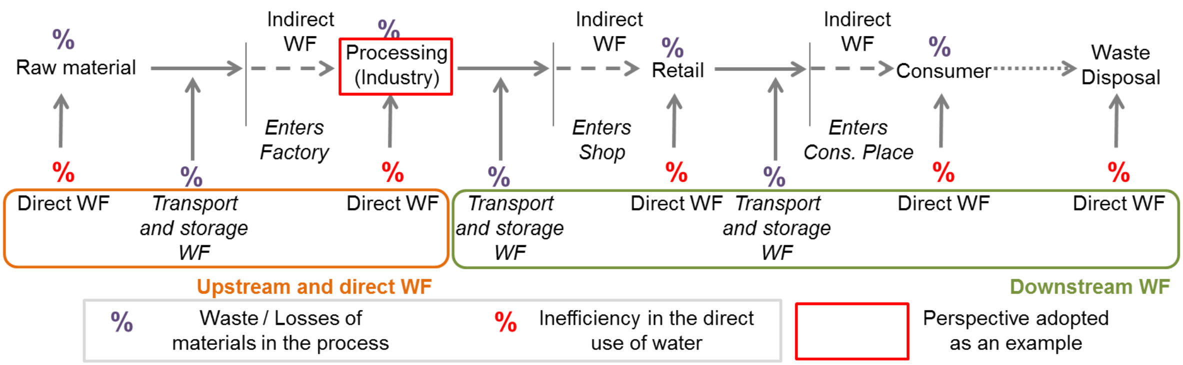

Figure 5.8. Upstream and downstream water footprints from the perspective of an entity of the supply chain (here, an industry).
Inefficiencies and losses are potential source of reduction of the WF.

The rationale for integrating the two perspectives in a single indicator is that an entity can take action to reduce its impacts on water resources by looking at both sides. For example, a food company could take measures to reduce food waste by improving packaging or the transport and storage conditions of products. These measures would potentially change the behaviour of consumers and retailers, which are entities situated downstream in the supply chain. As a result, the 'Integral WF' of this company would be reduced.

In the case of final consumers, the novelty in the consideration of a 'downstream WF' would correspond mainly to taking into account the WF of waste disposal (water treatment plant, waste collection and processing, etc.) (Figure 5.8). This final stage has not been included in the WF assessment so far (see the traditionally considered supply chain model on Figure 5.7, a). It may especially be relevant for the grey WF.

In this thesis, the issue of waste in the supply chain and the implication for the 'Integral WF' of products is illustrated in the case of the Campo de Dalías aquifer. The agricultural production is composed of fresh fruits and vegetables that are usually directly delivered to customers, i.e. unprocessed in the supply chain. Thus, this case is convenient for direct illustration of the concepts introduced in this section. The detailed methodology is presented in Section 2.4 of Chapter 9 on Campo de Dalías Aquifer.

V.6 Synthesis

- The WF measures human appropriation of freshwater resources. It constitutes a flexible indicator that can be applied at any scale, for any entity or group (consumption perspective) or a geographical area (production perspective) to measure the water consumption that is generated by or within the object of study, both directly and indirectly.
- The concept of WF has been introduced initially in relation with the debates on virtual water flows between countries, referring to the WF of 'Nations' as the sum of the internal and external water resources consumed to sustain the domestic consumption. The idea of assessing the environmental impacts in the country of origin and the focus on products and supply chain are subsequent developments of the concept.
- A footprint, or 'human appropriation' of resources, associates the direct and indirect resources consumption with an entity of the economy. Thus, the WF can be related to perspectives of physical flows accounting in the economy that have been previously contemplated under 'embodied' resource or energy, or the ecological footprint.
- The idea of 'human appropriation' also implies to contemplate the adequate indicator for the human pressure. By taking into account water that is not available again in the

river basin after withdrawal, the WF appears as a specific indicator that is related to other water accounting frameworks introduced earlier, e.g. in irrigation sciences. A reflection on the indicator for water use and consumption is also important in relation to the significance of economic indicators such as water productivity.

- The consideration of the green water allows introducing a series of debates relative to a better benefit from rainfall in agriculture or land use pressures. However, these subjects involve many issues. Decisions based on green WF are complex and cannot be interpreted only on the basis of a WF accounting. The general call to improve green water productivity to ease pressure on global blue water resources is simplistic and does not reflect the complexity of the global economy.
- The main debate between a volumetric WF and an impact-weighed WF, related to the LCA approach, comes from the vision that a direct interpretation of a volumetric WF is not possible. Yet the formulation of an indicator able to report the final environmental impact is complex. It is harder to interpret the final result and potential uncertainties and errors, as an additional factor is introduced. Moreover, the options to reduce a weighted WF are certainly the same as to reduce a volumetric WF and equally complex, since they involve a better efficiency or productivity of water, a much debated issue by agronomists, or a change in consumption pattern.
- Water stress indicators are also part of the environmental sustainability assessment of a traditional WF. However, there is no consensus so far on the content of this ‘sustainability assessment’ of a WF. In the same way, the ‘economic sustainability’ and ‘social sustainability’ have still not been formalized and these aspects are not often taken into account, despite a claim it is fully part of the traditional WF approach.
- The Integral WF is proposed to account for the whole water resource consumption on the life cycle of a product. Compared to the traditional WF, which focuses on the ‘upstream’ water consumption, it integrates also ‘downstream’ water consumption, which allows identifying new options for WF reduction, particularly regarding waste and losses in the supply chain. These potentially take place at every stage, and especially during transport and storage, which are often overlooked.
- The example of the issue of waste and losses is one of the realistic advances that a view in terms of WF in a consumption perspective has allowed to identify, together with, for instance, a reflection on the equity and the impact of the consumption pattern, especially diets.

Chapter VI

CASE STUDY METHODS AND DATA

<u>List of symbols and abbreviations of this chapter</u>

AIW	applied irrigation water (m^3/ha)
‘allowance’	(m^3/ha)
CWC_b	crop water consumption of blue water (m^3/ha)
CWC_g	crop water consumption of green water (m^3/ha)
‘drought’	(dimensionless)
eff	irrigation application efficiency (dimensionless)
DEM	direct employment generated by the use of water (number of jobs/m^3)
DEV	direct economic value of water (€/m^3)
ET	crop evapotranspiration (m^3/ha)
ET_0	crop reference evapotranspiration (m^3/ha)
K_c	crop coefficient for ETR calculation (dimensionless)
MP	market price of production (€/kg)
P	rainfall (m^3/ha)
P_{eff}	effective rainfall (m^3/ha)
PWP	physical water productivity (kg/m^3)
r	share of return flows (dimensionless)
W_{agri}	total groundwater withdrawal for agriculture (m^3)
$W_{urb/ind}$	total groundwater withdrawal for urban or industrial use (m^3)
Y	crop yield (kg/ha)
WF_{b_ground}	total water footprint from groundwater for the agricultural sector (m^3)

VI. CASE STUDY METHODS AND DATA

This chapter begins with the presentation of the methodology for quantifying water resource use under the water footprint approach and the data used in the different case studies. Afterwards, the methodology and data for the economic and social indicators are also detailed. Finally, a discussion of some points relative to the methods and data are analysed in relation to: a) the issue of scarce availability of data associated specifically with groundwater and b) some points that differentiate the methodology of this thesis from the mainstream approach of water accounting under the WF approach.

VI.1 Water footprint quantification

As this thesis is dedicated to the characterization of groundwater use, the methodology of WF calculation is oriented towards the determination of the blue WF. The green WF has only been integrated in the Guadalquivir River case study, which considers the role of groundwater within a balance of water resource flows at the scale of a river basin as a specific objective. The grey WF, which integrates the effect of pollution of water resources in the WF indicator (Hoekstra et al., 2011), is not considered in this thesis. In this section, the methods that are specifically used in the different case studies are presented together with more general considerations for water accounting. These considerations contribute to justifying the assumptions adopted in the case studies and to introducing a series of issues regarding the relevance of the WF indicator, which is one of the main objectives of the thesis.

VI.1.1 Green water footprint of agriculture (for Guadalquivir River basin case study)

Crop evapotranspitation (ET) is obtained from rainfall water stored in the soil (‘green water’) and, for irrigated areas, irrigation application (‘blue water’):

$$ET = CWC_g + CWC_b \tag{6.1}$$

with CWC_g the crop water consumption[46] of green water per hectare and CWC_b the crop water consumption of blue water per hectare.

If the amount of rainfall is sufficient to satisfy crop ET requirements (ETR), then there is no irrigation and the CWC_g is equal to ETR. In the absence of direct ET measurement in the field, ETR can be determined on the basis of formulae or computer programs, such as FAO’s CROPWAT (Allen et al., 1998; FAO, 2010), which multiply the reference ET (ET_o) by a crop coefficient (K_c, usually obtained from literature review) on the growing period of the crop:

[46]The expression ‘crop water consumption’ is preferred to ‘crop water use’ (usually abbreviated as CWU), which is commonly introduced in WF assessments (e.g. Aldaya et al., 2010b; Gerbens-Leenes et al., 2009; Hoekstra & Mekonnen, 2012a) since the term ‘use’ is vague and is usually indistinctly introduced to refer to total or consumptive use.

$$ETR = K_c \times ET_0 \tag{6.2}$$

ET_o depends on the local climatic conditions. The value of K_c changes according to the stage of development of the plant. It rises progressively (usually a linear evolution is considered), up to a value where the plant attains its full development, i.e. its maximum ETR. The value of K_c depends also on parameters, such as the nature and thickness of the soil layer and the irrigation method. It is noteworthy that the value of ET_0, and consequently of ETR, corresponds to a plant growing in optimal conditions, i.e. without lacking water or nutrients or suffering from plagues. To take into account the reduction of ETR that is due to lack of water, an additional coefficient can be introduced (K_s), which depends on the water availability in the soil (Allen et al., 1998).

In case of insufficient rainfall to satisfy the ETR, the share of crop evapotranspiration obtained from green water (i.e. CWC_g) is obtained from the effective rainfall (P_{eff}), which is the rain water that is available for the plant in the soil (based on FAO/AGLW method, FAO, 2010):

$$P_{eff} = 0{,}6 \times P - 10 \text{ for } P \leq 70\text{mm} \tag{6.3}$$

$$P_{eff} = 0{,}8 \times P - 24 \text{ for } P > 70\text{mm} \tag{6.4}$$

where P comprises the monthly rainfall.

In other terms, CWC_g is calculated as the minimum between P_{eff} and ETR:

$$CWC_g = \min(P_{eff}, ETR) \tag{6.5}$$

This calculation is done at a monthly step and CWC_g is summed up over the crop growing period or over the whole year for perennial trees, following the recommendations of Hoekstra et al. (2011). Thus, ET from agricultural land out of the growing season is not computed in the main calculation of green WF of agriculture for annual crops.

However, a specific objective of the Guadalquivir River basin case study is to present the results of the WF quantification within a water balance at the scale of the whole river basin. This requires also taking into account green water consumption on agricultural land out of the growing season. Based on the results of the hydrologic model Balance MED, which has been applied to an area comprised within the Guadalquivir River basin (Sierra Norte de Sevilla) by Willaarts et al. (2012), green water consumption during the non-cropping season is assumed to amount to 40 % of green water consumed by agriculture for the whole year. Even if the model was applied to a small area of the river basin, this value is introduced as a first approximation.

The crop coefficients are obtained from Allen et al. (1998), AQUAVIR (2005) and Orgaz et al. (2005) and plant and harvesting dates are obtained from MAPA (2002). ET_o is provided by the

Spanish meteorological agency (AEMET, 2012). One meteorological station is selected for each of the ten provinces covering the basin. K_s has not been considered.

VI.1.2 Blue water footprint accounting: general considerations

The blue WF is an indicator of the appropriation of freshwater by humanity. That is to say, it contemplates the water volume that is not delivered back in the same water body for further use after an intended use took place (Hoekstra et al., 2011). This can be due to different reasons:

- water is consumed (evaporated) during the intended use;
- water is consumed on the way from the point of withdrawal to the intended use (the share of conveyance losses that is evaporated or turned unusable);
- water is consumed on the way from the intended use to the point it is available again for further use (the share of return flows that is evaporated or turned unusable);
- water is delivered back in a period where it has no value for the users downstream (e.g. high flows period), whereas withdrawals have impacted on other users (e.g. during low flows period);
- water is incorporated into the product (usually a very small fraction).

In general, only the first component is contemplated in WF assessments. The integration of water evaporated during the transfer from its source and back to the natural environment in fact depends on the level of detail and objective of the study[47]. For instance, a detailed description of the destination of all the return flows is out of the scope for a study covering a whole basin, while it is recommended at the scale of an irrigation project[48]. When conveyance losses and return flows are never delivered back to the system of interest, the blue WF is constituted by the whole amount of withdrawals. Thus, the components of the WF depend on the reusability of the return flows within the boundaries of the system (Figure 6.1).

- *Case of a confined aquifer (e.g. Campo de Dalías and La Loma de Úbeda Aquifers).* The aquifer is isolated from the surface by impervious layers and return flows cannot percolate directly to the aquifer. The WF equals the whole amount of withdrawals.

[47] Zimmer & Renault (2003) already noted that “In most studies on virtual water for food, the basic value of virtual water only considers the water evapotranspired at field level. However for irrigated agriculture, water losses either for the field application or during the distribution must be considered if there is no possibility of recycling these losses at basin level. It might be useful to introduce a correction coefficient to include them as proposed by Haddadin (2002). Furthermore water leaching sometimes required in arid areas to deal with saline water must also be considered as water consumption.”

[48] Omitting the potential consumption of conveyance losses and return flows implies the WF is underestimated, i.e. the whole pressure on water resources is not accounted for. In fact, depending on the level of detail, the inclusion of these fractions may not be relevant, compared to the error in the estimation of the main fraction (water consumed during intended use).

- *Case of an unconfined aquifer (e.g. Western Mancha Aquifer)*. A distinction should be made between agricultural use, on the one hand, and urban and industrial uses, on the other hand. Return flows of irrigated areas located above the aquifer can be assumed to percolate in the aquifer and should not be accounted in the WF (Foster & Perry, 2010). In the case of urban and industrial uses, treated return flows are usually delivered back to surface water bodies, and the whole amount of withdrawals can be considered as a WF from the aquifer perspective, even if a part can infiltrate in losing streams.
- *Case of a river basin (e.g. Guadalquivir River basin)*. Depending on the local situation, conveyance losses and return flows can contribute to resource availability downstream, whether they are delivered back to a surface water body or percolate in an aquifer.

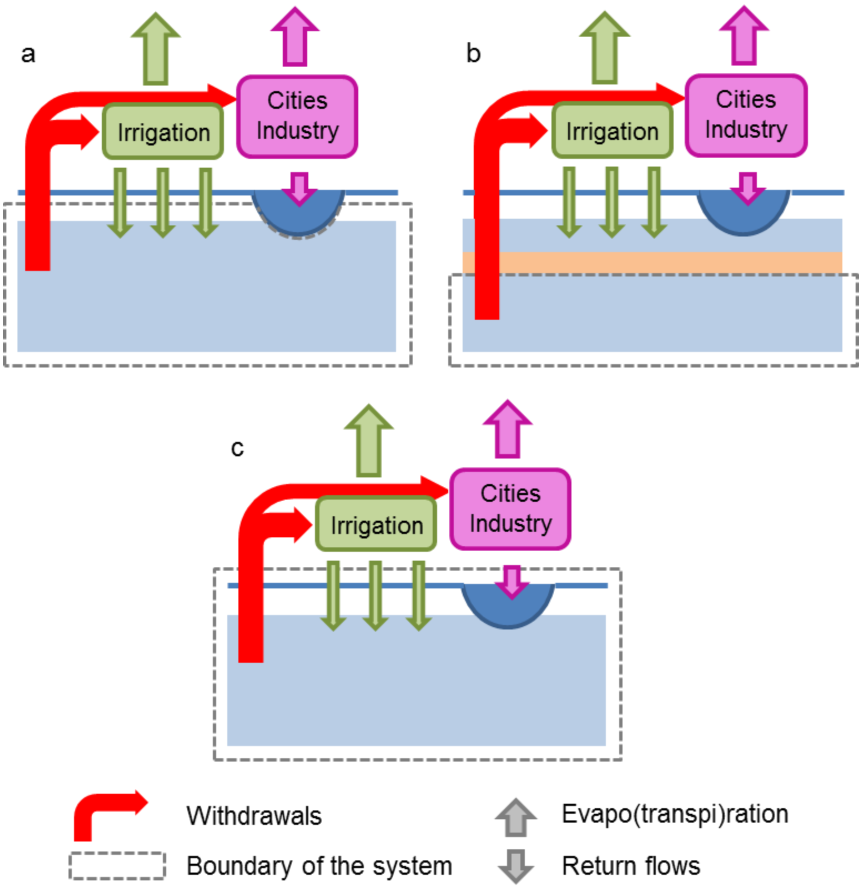

Figure 6.1. Blue water footprint accounting according to the boundaries of the system. (a) an unconfined aquifer, (b) a confined aquifer, (c) a river basin.

VI.1.3 Blue water footprint of agriculture

- <u>General method</u>

Depending on the reusability of the return flows, the blue WF is equal to the whole applied irrigation water (AIW) or this value multiplied by the total irrigation application efficiency (eff):

- when return flows are not reusable:

$$\mathrm{WF} = \mathrm{AIW} \quad (6.6)$$

- when return flows are reusable:

$$\mathrm{WF} = \mathrm{eff} \times \mathrm{AIW} \quad (6.7)$$

Quantifying the WF from the estimation of withdrawals might be viewed as not very accurate when compared to other more common approaches for WF accounting, all the more so since the lack of data on the actual amount of withdrawals can lead to make rough estimations. Nevertheless, usual methods for agricultural WF accounting contemplate theoretical irrigation water requirements that can be estimated with accuracy but do not correspond to the real water consumption (see Section 3 of this chapter on discussion of the methods for the full detail of the reasons and implications of this difference in the approach). It should be noted, however, that the WF value can be estimated from the irrigation water requirements when the farmers' strategy is to apply the required amount of water.

- Western Mancha Aquifer case study

The CWC_b values for the year 2000 and the period 2007-2008 have been selected from various sources (Table 6.1). The approach is to consider the satisfaction of the irrigation requirements presented by SIAR (2011) for the crop that are more sensitive to water stress and mainly cultivated during summer, where there is low rainfall (maize, melon, pepper, tomato and onion).

Table 6.1. Crop blue water consumptions for the Western Mancha Aquifer case (m^3/ha).

	2000[a]	2007	2008	2009	Source and comments
Cereals	1,500	1,500	1,500	1,500	CH Guadiana (2010)
Vine	2,000	2,000	2,000	2,000	Ruiz-Pulpón (2010): applied volume 2000-2500 m^3/ha - confirmed by farmers during field interviews
Maize	6,500	6,100	6,600	6,820	SIAR (2011)
Melon	4,600	5,400	4,300	4,100	SIAR (2011)
Pepper	5,050	5,610	4,500	5,050	SIAR (2011)
Tomato	5,680	5,460	5,670	5,900	SIAR (2011)
Garlic	2,250	2,250	2,250	2,250	CH Guadiana (2010)
Onion	6,050	6,600	6,200	5,350	SIAR (2011)

[a] For the year 2000, few data are available. As the WF for this year is only indicative (for comparison), the value for crop water consumption is taken as the mean for the period 2007-2009.

Ruiz-Pulpón (2010) and farmer interviews allowed estimating the CWC_b of vines at 2,000 m^3/ha. Regarding cereals and garlic, the values have been obtained from the 'Withdrawals Regulation Plan' (CH Guadiana, 2010), which introduces the values used to control the amount of water consumed by the farmers[49]. The lower water application as compared to full satisfaction of crops needs can be explained from a series of reasons, e.g. deficit irrigation practices, economics of groundwater use or the institutional context with the definition of water quotas (that are more or less complied with).

- Guadalquivir River Basin case study

In the Guadalquivir basin, 77% of the agricultural demand is satisfied as an average (Junta de Andalucía, 2010b) and water delivered to farmers can be restricted by the Guadalquivir River Basin Authority during drought periods. CWC_b is calculated by multiplying the irrigation allowances (factor 'allowance' in m^3/ha) for each crop group within each management district (given by AQUAVIR, 2005) by the application efficiency ('eff'). As a general rule, a 0.8 application efficiency is considered (Junta de Andalucía, 2010b) with the exception of paddy fields, with 0.5 (Strosser et al., 2007). Additionally, irrigation allowances are reduced depending on the level of drought in each management district, a measure that is imposed by the Special Plan for droughts (CH Guadalquivir, 2007). According to the instructions of this plan, allowances are reduced by 5, 30 or 70% (factor 'drought' given as a fraction of unit), depending on the drought level in the management district presented by MARM (2010) (Appendix 4). The final expression of CWC_b is given by:

$$CWC_b = \text{allowance} \times \text{eff} \times \text{drought} \tag{6.8}$$

In the case of groundwater, access to aquifer storage means a constant availability of groundwater, independently of climatic variations. For the two years (1997 and 2002) where data availability on groundwater irrigated area allows calculating the WF from groundwater based on CWC_b, ETR is considered to be satisfied and CWC_b is obtained as the difference between CWR (given by AQUIVIR, 2005) and CWC_g (calculated according to the method presented in Section 1.1 of this chapter). The case of olives is addressed specifically, as the strategy of deficit irrigation implies that only around half of the CWR is met (Fereres and Soriano, 2007), and a constant CWC_b of 2,000 m^3/ha is introduced.

For the year 2008, only data on total withdrawals by groundwater body is available (CH Guadalquivir, 2010a), without detail on the nature of the crops that are irrigated. Thus, the WF from groundwater for the agricultural sector (WF_{b_ground}) is obtained by multiplying the abstracted volumes (W_{agri} in m^3) by an application efficiency (eff) of 0.85.

[49] For the specific case of cereals, this has been confirmed during farmers' interviews, who explained that, during the period of study, the cost of pumping and of other inputs (particularly fertilizers, which should be added in higher quantity when more water is applied), in a context of low market prices, implied that a minimum of water was applied. Nevertheless, this situation might be only temporal as many factors, e.g. aquifer level and market prices, have changed since then.

$$WF_{b\_ground} = eff \times W_{agri} \tag{6.9}$$

- La Loma de Úbeda Aquifer case study

The main aquifer is a confined aquifer and return flows do not percolate. It means that the WF is constituted by the totality of the withdrawals. Values of withdrawals in the year 2002 have been obtained from Gollonet et al. (2002) and in the year 2008 from the hydrological plan of the Guadalquivir River basin (CH Guadalquivir, 2010a). The value of CWC_b is 1,600 m^3/ha.

- Campo de Dalías case study

The main aquifer is also a confined aquifer and the WF is constituted by the whole amount of withdrawals. The amount of irrigation water applied by crop is obtained from Fernández et al. (2007) and Cajamar (2005).

VI.1.4 Blue water footprint of urban and industrial uses

- Method

Data on these two sectors are usually presented in terms of total withdrawals. For a study at aquifer or river basin scale, an approximation of the water not available downstream can be obtained considering a fixed percentage of return flows[50]. With r, the ratio of return flows on total withdrawals (W):

$$WF = (1 - r) \times W_{urb/ind} \tag{6.10}$$

- Case studies

In the case of the Guadalquivir basin, the share of return flows is 72% and 44% for urban and industrial uses, respectively (CH Guadalquivir, 2010a). Regarding the Campo de Dalías case study, the whole amount of withdrawals is considered for the WF (no return flows to the aquifer). For the Western Mancha and the La Loma de Úbeda Aquifers, no industrial and urban uses have been considered since these are negligible as compared to agricultural withdrawals.

VI.1.5 Blue water footprint linked to dams, conveyance losses and return flows

- Dams water footprint

Reservoirs are essential to make water available during the whole year. Although the use of water in dams (particularly for hydropower) is often said to be non-consumptive, as the water is not diverted from the river, evaporation from the water surface can represent a significant share of the water consumption in regulated river basins (Martínez-Granados et al., 2011; Mekonnen and Hoekstra, 2012b) and should be integrated in the WF balance. It is often disregarded in WF

[50] In a detailed assessment of the WF of a specific industry or a city, details should be obtained on the different process that evaporate water (industrial process, heating for washing and cooking, lawns irrigations, etc.) and the potential losses in the distribution network that might recharge an aquifer.

studies, even if the calculated WFs would not take place without dam regulation and the associated evaporation.

In the case of the Guadalquivir River Basin, the volume of water evaporated from reservoirs is calculated according to Hardy and Garrido (2010), who established a linear relation between the evaporated volume and reservoir capacity on the basis of a survey of 44 Spanish dams. All reservoirs are considered as artificial lakes, i.e. the evaporation is human appropriation of water and constitutes a WF[51]. The WF of dams is not attributed to a specific use as they have numerous functions: electricity generation, satisfaction of demand, flood mitigation, etc.

- Conveyance losses

When water is delivered to the intended use, it can leak from pipes or non-impervious canals. These flows should be accounted as a component of the WF depending on their destination. For instance, riparian vegetation can develop along irrigation canals and the associated evapotranspiration should be accounted in the WF as it is not delivered back to the river basin[52].

This potential consumption has not been included in the case studies because groundwater is usually used in the proximity of the well in the agricultural sector. For the Guadalquivir River Basin case study, where the majority of irrigation water is surface water, the scale of the study does not allow considering such a level of detail.

- Return flows

All the return flows or water that is not evaporated during the intended use will not necessarily be available for further use. For example, in the case of urban and industrial use, (treated) wastewater can be delivered back in another catchment or directly to the sea.

In the case of agriculture, two types of return flows can be distinguished: water percolating into the ground and run-off. The re-use of run-off can be assessed based on the configuration of the drainage system of the irrigation project, identifying whether return flows are delivered back to an irrigation canal or the river, end up ‘irrigating’ plants outside the farm plots, or contribute to waterlogging (Lankford, 2012). Part of the water that percolates in shallow aquifers, can show up rapidly, near the plot and identically feed the vegetation and contribute to waterlogging.

[51] If a natural lake is used for hydropower, for instance, the evaporation should not be considered as a WF since the water consumption can be associated to the lake ecosystems.

[52] These ‘green corridors’ are sometimes old for traditional irrigation systems and can have a high ecological value. They also contribute to the landscape of irrigated areas. This could question the inclusion of their water consumption as a WF (human appropriation), i.e. question the repartition between ecosystems and human consumption. It may be problematic to modernize these traditional irrigation systems since the new impervious canals would make an end to the flows towards these ecosystems.

Water flows that percolate more profoundly contribute to the recharge of the aquifer and the spatial and temporal potential of their re-use is linked to the destiny of the groundwater outflows (natural outflows or pumping). Even if any generalization should be avoided, it can be observed that the majority of irrigation projects take place on alluvial aquifers, because this is where the more fertile areas are located[53]. Alluvial aquifers are characterized by a mid- to high permeability, which means that the outflows to the river from irrigation return flows can occur quite 'rapidly' (from a few days to a few years depending on the distance to the river, see Kendy and Bredehoeft, 2006). Part of the return flows reaches the river stream during high flow periods, where they may not be useful (Figure 6.2).

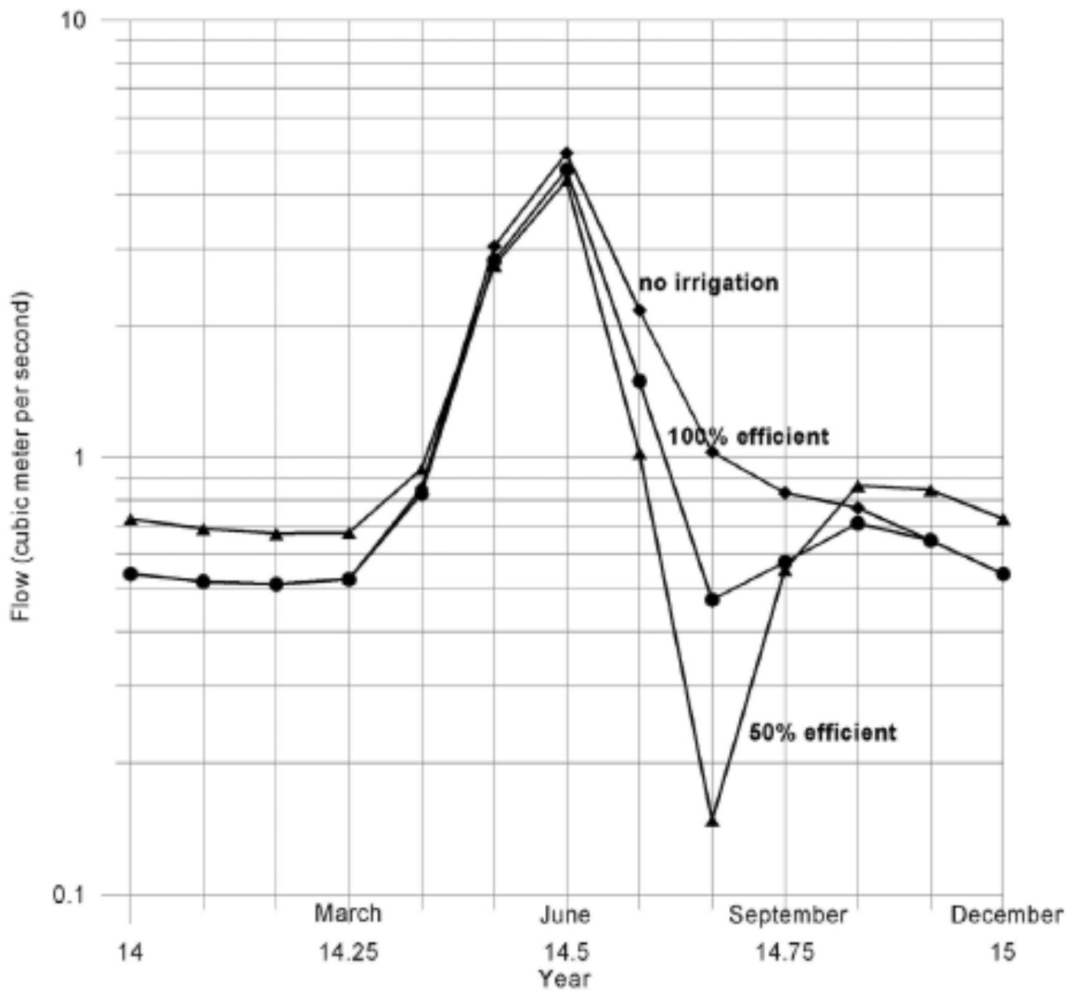

Figure 6.2. Change in a river flow as a consequence of irrigation (Gallatin River, USA). Source: Kendy and Bredehoeft (2006).

The 100% efficiency curve is the same as the non-irrigation curve except during the May through October irrigation season. This figure shows the influence of return flows on river flows: from end of September to May, flows are higher for the 50% efficiency scenario as compared to the no irrigation scenario since return flows (absent in the case of 100% efficiency) go back to the river during the whole year.

[53] More details on the possibility of re-use of the return flows according the local configuration and the situation in the river basin (e.g. head waters, environmentally sensitive area, seashore) can be obtained from Molden et al. (2001), who defined 'hydronomic zones' in order to target areas where irrigation efficiency should be promoted.

However, these return flows are not necessarily 'lost' for downstream users. If a reservoir is located downstream, they can be stored and used for the following irrigation campaign. In river basins that are extremely regulated, the allocation of flows over the year is altered as compared to the natural situation. More water can flow during summer than winter in order to satisfy irrigation demand. Thus, the role of return flows in the system, and whether they are 're-usable' or not, should be assessed carefully.

To sum-up the assessment of the reusability of return flows is not straightforward: many factors contribute to return flows consumption. Even if it can be affirmed that the total withdrawals are not the most suitable indicator to report the human 'appropriation' of water resources (Hoekstra et al., 2011), an approach that would only focus on the consumptive use for the intended process (e.g. in the field, for agriculture) would necessarily underestimate the real consumption of water resources (the real WF). Thus, if a balance of water resources consumption based on the WF is done at basin scale, more water would be accounted as available than there is in reality.

Nevertheless, the opportunity and necessity to identify all these potential additional factors of water consumption depend on the scope and objective of the study. In the case of the Guadalquivir River Basin, due to the scale, all destinations of return flows cannot be assessed and the whole amount of return flows is considered to be reused. This is the same for the Western Mancha Aquifer as there is almost no run-off and return flows percolate. In the two other case studies, no return flows can reach the aquifers since these are confined aquifers.

VI.1.6 Water footprint of livestock and pastures (Guadalquivir River case study)

The WF of livestock includes both direct and indirect water consumption. Direct WF refers to the water consumption for animal drinking and for farm management. Indirect WF refers to the virtual water embedded into animal feed coming from the internal agrarian production (already accounted in the agricultural WF), pastures and feed imports (WF exerted out of the basin).

The WF of livestock and pastures is only contemplated for the Guadalquivir River Basin. The additional WF associated to livestock includes the direct consumption (based on Rodríguez-Casado et al., 2009) and the WF of pastures. The WF of animal feed is already accounted in the crops WF when it comes from the area and is a WF exerted outside when feed is imported. The WF of pastures (green water) is estimated multiplying an ET of 1,930 m^3/ha (Corominas, 2011) by the pasture area in the basin (6,100 km^2 according to CH Guadalquivir, 2010a).

VI.1.7 The water footprint of a geographical area (aquifer or river basin)

In order to estimate the total WF within the case study areas, the WF of all water uses occurring within their boundaries are added up. The green and blue agricultural WFs are obtained as:

$$WF_{green} = \sum (S_i \times (CWC_g)_i) \quad (6.11)$$

$$WF_{blue} = \sum ((S_{irr})_i \times (CWC_b)_i) \quad (6.12)$$

with S_i and $(S_{irr})_i$ the total and irrigated surface relative to the crop i (ha) and $(CWC_g)_i$ and $(CWC_b)_i$, the green and blue crop water consumption for the crop i (m^3/ha).

In the case of the Western Mancha Aquifer, estimation of the WF from non-authorized groundwater use is based on specific data on non-authorized irrigated areas.

VI.1.8 Virtual water content and physical water productivity

Water productivity is defined as a ratio between an output and an input and its significance depends on the selected factor for output and (water) input (Molden and Sakthivadivel, 1999; Playán and Mateos, 2005; Hellegers et al., 2009; Berbel et al., 2011). CWC_b is specifically considered for the input (green water is not included here). In the case of physical water productivity (PWP in kg/m^3), output is defined as the crop yield (Y in kg/ha).

$$PWP = Y / CWC_b \quad (6.13)$$

The PWP corresponds to the inverse of the blue WF of crops expressed in unit of production (m^3/kg or m^3/item), which can also be referred to as virtual water content. The virtual water content can be expressed for irrigated crops only, or referring to a generic crop coming from the area, whether it is irrigated or not. This ‘overall virtual water content’ is obtained from the ratio of the whole production of the area (rain-fed and irrigated) over the total blue WF of the considered crop. It is noteworthy that it is this value that is presented in the world database elaborated by Mekonnen & Hoekstra (2010b), which is the input for many WF assessments.

The main source for yields is the annual yearbook of the Agriculture Ministry (MAGRAMA, 2013), which shows the data for each province. The province of Ciudad Real is used for the Western Mancha Aquifer. The province of Seville, Cordoba, Jaén and Granada are considered for the Guadalquivir River Basin, according to the location of the irrigated areas. In the case of the Campo de Dalías case study, yields have been obtained from Junta de Andalucía (2013).

VI.2 Economic indicators of water use

VI.2.1 Direct economic value

The direct economic value of water (DEV in €/m^3) corresponds to the ratio between the market value of the production (MV in €/ha), or gross revenue of the farmer, obtained as the yield multiplied by the market price (MP, in €/kg), and CWC_b:

$$\mathrm{DEV} = \mathrm{MV} / \mathrm{CWC_b} = (\mathrm{Y} \cdot \mathrm{MP}) / \mathrm{CWC_b} \tag{6.14}$$

DEV is commonly called ‘water productivity’ or ‘apparent water productivity’. However, since the definition of this kind of indicators is often imprecise and since it aims at showing the economic value generated by the use of water, the term ‘economic value’ is directly included in its name in this thesis.

In the case of the Western Mancha Aquifer, another ‘economic output’ introduced is the supplemental gross revenue obtained from irrigation as compared to rain-fed production. It consists in subtracting the market value of rain-fed production that would have taken place instead of the irrigated production to the value of the real irrigated production:

$$\mathrm{DEV} = \frac{\mathrm{MV_{irr}} - \mathrm{MV_{rain}}}{\mathrm{CWC_b}} \tag{6.15}$$

When a crop cannot be cultivated under rain-fed conditions, the gross revenue of the most profitable rain-fed production that can potentially be grown by the farmer (barley in the case of maize and vine in the case of vegetables) is subtracted from the gross revenue generated by the irrigated crop. This approach aims at representing more precisely the value of water since the agricultural sector generates value without blue water (rain-fed) and it is questionable to attribute all the value of the production to water[54]. A same kind of indicator is used in the La Loma de Úbeda case study, with the subtraction of the market value of olive production under rain-fed conditions to the value of irrigated production.

Another economic indicator is the profit obtained by farmers. It consists in subtracting the production costs to the market price of the production. It is presented for the Campo de Dalías case study, where data on production costs are obtained from Junta de Andalucía (2013).

For the Guadalquivir River Basin, production market prices received by farmers at national scale have been obtained from MAGRAMA (2013). However, at the local scale, the national average prices presented may not be relevant and local data sources have been given priority when available. In the case of the Campo de Dalías, these were obtained from Cabrera and Uclés (2012). For the Western Mancha Aquifer the market prices and EU CAP subsidies for cereals have been obtained from DGAACM (2010). For the La Loma Aquifer, olive oil market prices have been obtained from MAGRAMA (2013), which is acceptable in this case since more than 90 % of olive oil production originates from Andalusia (in 2010).

[54]The value of the alternative rain-fed production can be viewed as the opportunity cost of land (see Berbel et al., 2011).

VI.2.2 Employment generated from the use of water

The social dimension has been included in terms of direct employment generated by the use of water (DEM in number of jobs/m^3), and it is defined as the number of direct jobs necessary for one hectare production in a year divided by CWC_b. In the Western Mancha Aquifer, jobs that would have been required under normal rain-fed conditions are subtracted in order to obtain the part of employment directly attributable to irrigation.

For the Western La Mancha case study, labour units per crop and year (differentiating rain-fed and irrigated crops) have been obtained from DGAACM (2010). For the Campo de Dalías case study, labour costs and family working hours were obtained from Junta de Andalucía (2013).

VI.2.3 Aggregation for the whole area of interest

The economic and employment indicators are also aggregated to obtain the overall economic value and jobs generated in the whole study area, by crop or in total. This is done by multiplying the indicators relative to one hectare by the crop areas. In the case of the Western Mancha Aquifer, the legal and non-authorized activities are distinguished.

VI.2.4 Synthesis of the indicators and their calculation

The main indicators and their relation with input data are shown on the Figure 6.3.

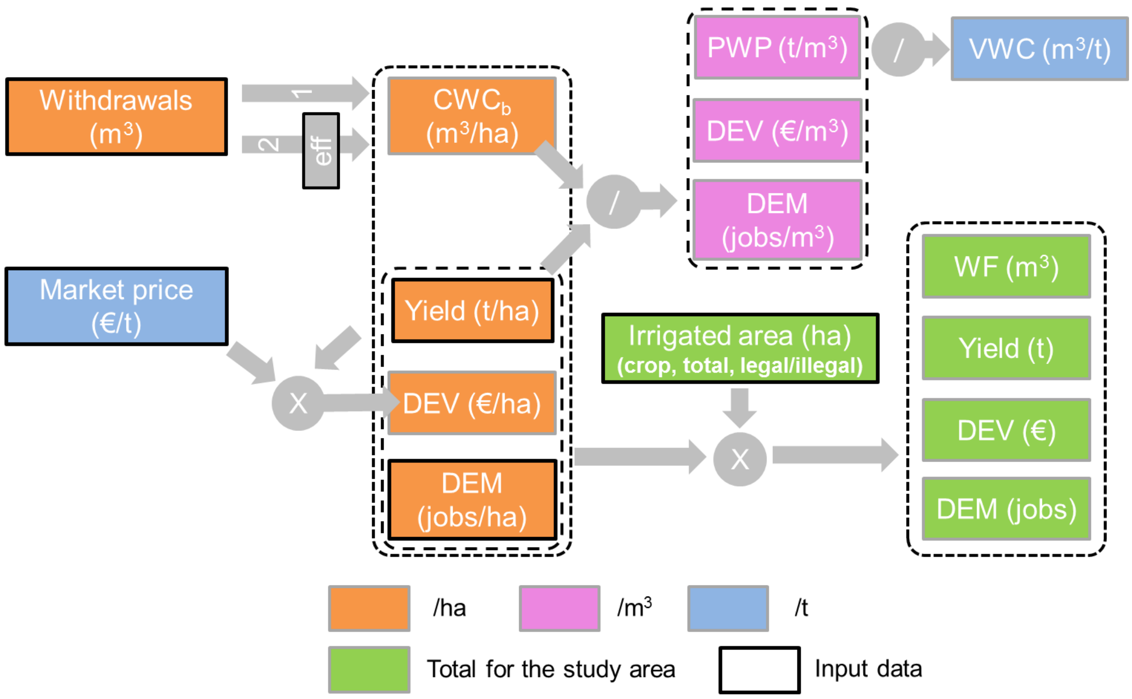

Figure 6.3. Main indicators and their calculation from data sources.
1: when return flows are not re-used; 2: when return flows are available again.
eff: efficiency; DEV: direct economic value; DEM: direct employment; WF: water footprint; PWP: physical water productivity; VWC: virtual water content. Only the main method to obtain the DEV is presented. For the description of the other monetary indicators see Section 2.1 of this chapter.

VI.3 Discussion of methods

VI.3.1 Data availability relative to groundwater

The availability and accuracy of data is an essential point for a robust and detailed assessment regarding water use. A traditional problem is linked to the area at which the data are generated. The scale (e.g. regional, national or a whole river basin) may not be adapted for a local study. Moreover, aquifer, river basin or irrigation district areas do not match with political or administrative boundaries. Different categories of data are also presented by the different organizations or administrations according to their needs and objectives. It is usually necessary to combine the information from different sources to obtain the data that is needed.

This issue is particularly relevant for groundwater. The lack of specific data on the origin of water usually restricts the possibility to undertake a detailed assessment of the WF from groundwater. The issue is not limited to the access to basic data, such as the areas specifically irrigated by groundwater. Ideally, much of the data, such as yield or costs, should be differentiated from surface water. For instance, the constant availability of groundwater allows farmers to prevent water stress for the crops, which has a positive influence on yields. Since yield data do not differentiate value according to the origin of water, a higher CWC_b implies that the calculated virtual water content of the crops (CWC_b / Yield) is higher (i.e. the water productivity is lower) for groundwater irrigation. This is not the case in reality thanks to the higher yield due to a better satisfaction of irrigation needs.

VI.3.2 An unfortunate hypothesis to quantify water footprints

- <u>A common but unrealistic assumption</u>

When rainfall has not been able to meet the crop water requirements (i.e. $CWC_g = P_{eff}$ in Equation 6.5), if the land is equipped for irrigation, the farmer can provide blue water to the crop to meet partially or totally the ET requirements (ETR). The crop water consumption of blue water (CWC_b) depends on the amount of water that is applied by the farmer. On the basis of Equation 6.1:

- if enough water is applied to satisfy ETR:

$$ET = CWC_g + CWC_b = ETR \qquad (6.16)$$

- if applied water is not enough to satisfy ETR:

$$ET = CWC_g + CWC_b < ETR \qquad (6.17)$$

In the first case, CWC_b can be directly obtained from the values of ETR and CWC_g that have been both previously determined (see Section 1.1 of this chapter):

$$CWC_b = ETR - CWC_g \qquad (6.18)$$

As ETR and CWC_g are calculated on the basis of bio-physical data that are quite easily available (e.g. rainfall, soil characteristics, climatic factors), Equation 6.18 can be an appealing way to determine CWC_b. Yet this expression is only valid when the farmer applied enough water to meet ETR. However, this is an assumption that is made in most of the WF assessments in order to calculate the blue WF from Equation 6.18 or based on the evaporative demand of the crop as the 'crop water use' (e.g. Chapagain and Hoekstra, 2007; Hoekstra and Chapagain, 2007a; Aldaya et al., 2010b; Bulsink et al., 2010; Garrido et al., 2010, Hoekstra and Mekonnen, 2012a; among others).

Nevertheless, this hypothesis may be unrealistic in many situations. In semi-arid regions – and virtually in any region where irrigation takes place, since it remediates the limited availability of water – farmers have to cope with restrictions in irrigation water availability. It means that the real ET, even in irrigated areas, can only be a fraction of the total crop water requirements, particularly during the driest years, when water allocation can be drastically reduced. In addition, agronomic practices such as supplemental and deficit irrigation are based on applying less water than full crop water requirements in order to improve water productivity (Fereres and Soriano, 2007)[55].

For groundwater, physical limits can be partially avoided thanks to the continuous availability of the stock of the aquifer. However, as shown by Ortega et al. (2004). pumping costs impact directly on profitability and decision of farmers to irrigate or not. This adds to the deficit irrigation practices mentioned above. The institutional context is also important, for instance with the definition of withdrawal quotas. Thus, whenever possible, the value of CWC_b should be obtained based on real irrigation water application values and not the assumed satisfaction of entire crop water requirements. The calculation of the crop irrigation water demand can serve as a proxy for CWC_b only in case there is evidence that farmers had access to sufficient water to satisfy these requirements and that this is the strategy they followed.

The assumption that ETR are met thanks to blue water application implies that studies that adopt this hypothesis overestimate the amount of blue WF (Thaler et al., 2012). In fact, the obtained WFs are an estimate of irrigation water demand to meet ideal conditions that do not correspond with the reality and the management by farmers, instead of being a quantification of real water consumption. Very few WF assessments recognize this caveat, with the notable exception of Gerbens-Leenes et al. (2009a) who note:

[55] These practices are not only due to the lack of water that requires saving water and using it efficiently. They are also determinant for the quality of the production (e.g. a limited water stress allows concentrating the sugar in grapes or melons).

> "Similar to earlier studies (Chapagain & Hoekstra, 2004; Hoekstra & Chapagain, 2007, 2008), the calculations have been based on the assumption that crop water use is equal to crop water requirements. When actual water availability is lower and water stress occurs, this study overestimates the crop water use. With respect to agricultural yields, we have taken actual yields (...)"

The previous citation illustrates also the paradox of introducing real yield data, whilst the crop water consumption is contemplated for an ideal situation, as observed in most WF studies. Thus, theoretically 'ideal' or 'optimum' yield should be also applied. This means that virtual water contents are overestimated[56]. Furthermore, while it is sometimes argued that yield is proportional to evapotranspiration in order to justify that WF quantification can be based on optimum conditions, it should be reminded that the linear relation is not valid for very low yields, which can be found on extensive areas of developing countries (Zimmer, 2013).

Finally, having an estimation of real water consumption is critical for 'environmental sustainability assessments'. The adequacy of such assessments, based on theoretical demand, can be questioned, such as in the case of Hoekstra et al., 2012[57].

- An illusory accuracy

The CWC_b should preferentially be obtained from data on water application by farmers for each crop. Real irrigation water application data might not be easily available and may lack relevance and accuracy, with a high level of uncertainty. However, in many cases, once irrigation efficiency has been contemplated, it is an estimation closer to the value of the real ET than the value obtained through the hypothesis of compliance with the irrigation requirements calculated through a formula, although this common method seems to be accurate (Figure 6.4). Indeed, the accuracy is related to the value of the crop irrigation requirements, not the real crop water consumption. Efforts to refine the value of crop irrigation requirements, e.g. considering a

[56] Unsurprisingly, Gerbens-Leenes et al. (2009a) note that "In the most extreme cases, [their] study found crop water requirements that were a factor of 2 different from earlier studies (…), whereas at other times the results were similar". The study was object of a reply due to unrealistic figures (Maes et al., 2009). However, in their response, Gerbens-Leenes et al. (2009b) affirm that "[they] compared WFs of bioenergy from different crops, using best estimates of actual yields under actual water use, rather than potential yields under optimized water use". This is in contradiction with the method used in their main paper and a table in the response showing 'Crop water requirement' and 'Irrigation requirement'. They discard Maes et al. (2009) based on other arguments. It would be interesting to analyze this situation, encountered in most WF assessments, under a 'sociology of sciences' lens (see Latour, 1988), since it represents a methodology that is self-justified, for instance based on earlier studies using it. Moreover, in spite of continuously using terms such as 'requirements' or 'crop water use', which have a clear meaning and reflect the methods effectively used, the results are said to represent real water use.

[57] In this specific study, the issue adds to the questionable introduction of 'monthly water availability' contemplated in regulated big river basins as if they were not regulated (no dams, i.e. natural flows). This is an example where a supposed higher accuracy, in this case a monthly time step instead of an annual step, in fact lacks relevance, as the assumptions are unrealistic.

shorter time frame for calculating the ETR or a detailed water balance in the soil, are in vain relative to the determination of the real CWC_b. It represents a quest for an illusory accuracy.

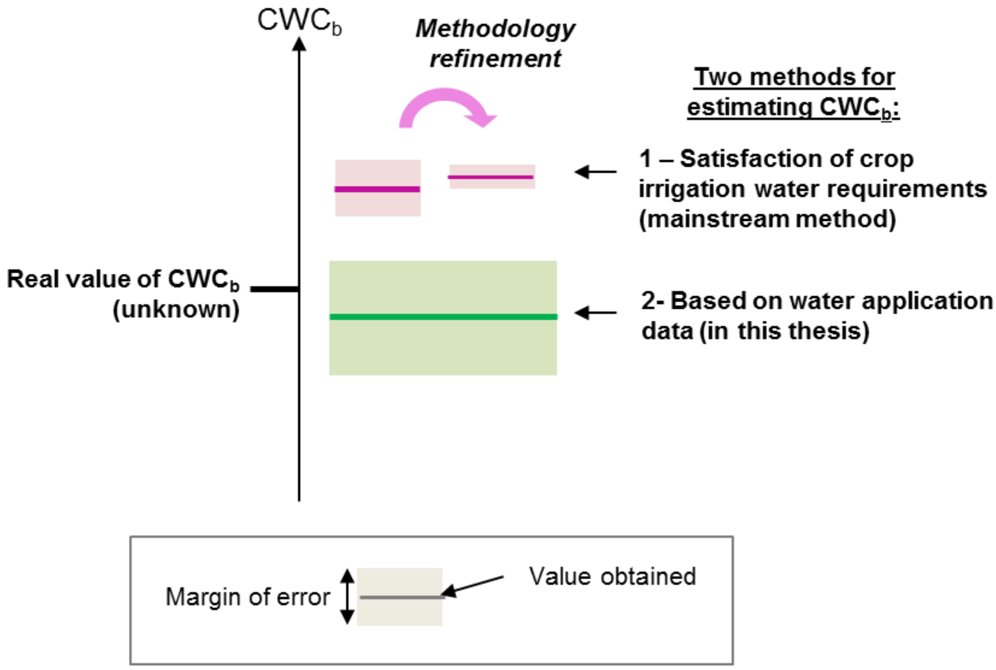

Figure 6.4. An illusory accuracy: common method to quantify crop blue water consumption versus the method used in this thesis.

While it can be recognized that the hypothesis on the satisfaction of irrigation requirements may be the most appropriate method for systematic studies at continental or world scale (e.g. Rost et al., 2008; Liu et al., 2009), downscaling these results for local assessment might be problematic. This issue is particularly relevant in relation with the WF quantifications led by Mekonnen & Hoekstra (2010b), which is a data base that often serves as an input for other studies (e.g. Ercin et al., 2013; Vanham and Bidoglio, 2013). The assumption of compliance with blue water requirements, with the use of CROPWAT, tends also to be perceived as the standard methodology for WFs quantification, whereas local data often would be more robust[58].

- Illustration for the Western Mancha case study

In order to visualize the implications of quantifying CWC_b from data on real use as compared to an evaluation based on ETR, the values adopted for the present study are compared with the ones presented by Aldaya et al. (2010b) for the same study area, distinguishing a dry, an average and a humid year (Table 6.2). In the case of maize and tomato, the satisfaction of crop water requirements is assumed in this thesis and the values are more or less the same (2008 is an average to dry year). However, Aldaya et al. (2010b) overestimate the CWC_b of vines and

[58] See the issue of ‘a self-justifying methodology’ in footnote 56, p.126.

cereals by around 50% (in the 'average year' scenario), which can be problematic since these are the two crops that occupy most of the irrigated area in the Western Mancha Aquifer (65% for vine and 20% for cereals in 2008, see Chapter 7)[59].

Table 6.2. Crop water consumptions given by Aldaya et al. (2010b) and for the present study in the Western Mancha Aquifer (m^3/ha).

	Dry[a]	Average[a]	Humid[a]	Present study (2008)
Vines	3,619	2,890	1,670	2,000
Barley	3,743	2,200	2,079	1,500
Wheat	3,759	2,277	2,058	1,500
Maize	7,460	6,534	4,445	6,600
Tomato	6,510	5,779	3,845	5,670

[a] Aldaya et al. (2010b) / Only the crops common to the two studies are presented.

VI.3.3 Modelling irrigation recommendations and optimal growth

Traditional models of ET requirements are based on optimal growth of the plant. As described in Section 1.1 of this chapter on the method for green water accounting, this can be corrected thanks to the integration of a coefficient that takes into account water availability restrictions in the soil (K_s). Yet, the optimal development of the plant may not correspond to the optimal conditions for a valuable production, in quantity and quality. This refers particularly to deficit irrigation techniques that consist in reducing irrigation water at some stage of the development of the plant, to obtain some benefits (e.g. more sugar in the fruits). A model that would not integrate this technique would provide wrong irrigation requirements that do not correspond to the farmer practices and that do not help in obtaining the real value of blue water consumption.

[59]This example shows also that when CWC_b is based on the satisfaction of the ETR, its value is higher for drier years (Table 6.2). This is paradoxical since the water availability is usually reduced in dry years, even if this should be relativized by the availability of groundwater and the regularization of dams.

Chapter VII

WESTERN MANCHA AQUIFER

VII. WESTERN MANCHA AQUIFER

Objective of the case study: The main objective is to assess groundwater use, from a physical, economic and social perspective, in a context where a protected area of high ecological value (Tablas de Daimiel wetland) depends directly on high groundwater levels. As a specific approach for the case study, non-authorized activity is characterized, which allows the identification of key drivers for informal water use. Based on this assessment, some aspects of a recent public plan, the Special Upper Guadiana Plan, are discussed.

VII.1 Introduction

VII.1.1 Location and main features of the study area

The study area, the Western Mancha Aquifer (WMA) hydrogeological unit covers an area of around 5,000 km^2 and is located in the Upper Guadiana basin (Figure 7.1).

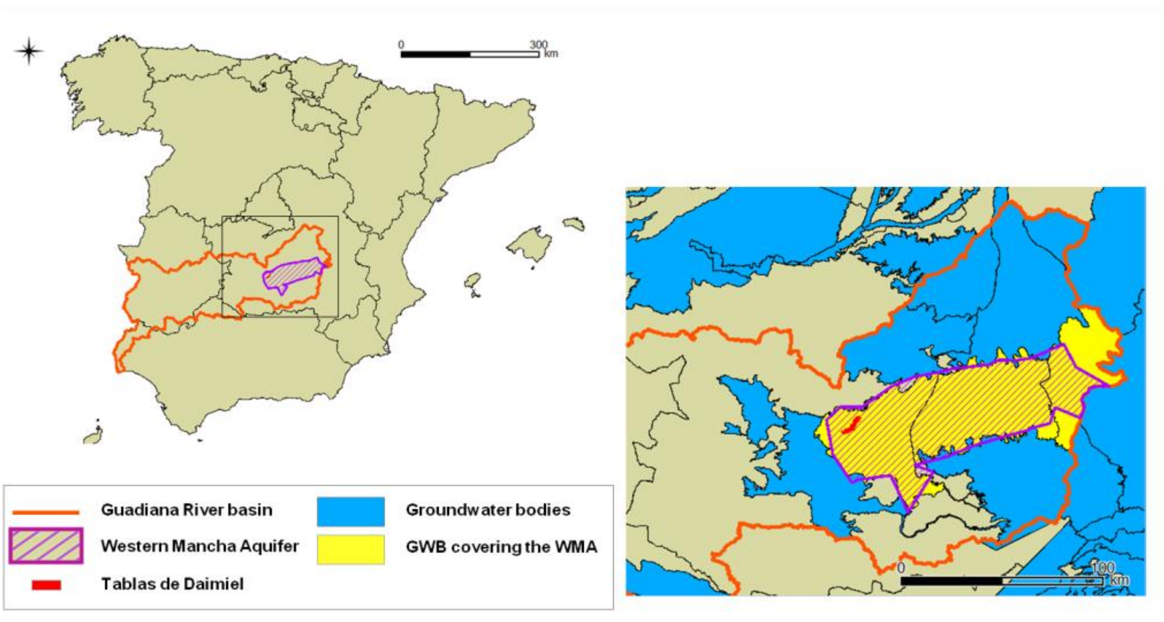

Figure 7.1. Location of the Western Mancha Aquifer and Tablas de Daimiel wetland.
Hydrogeological units will be replaced by the groundwater bodies (GWB) as the management scale of groundwater once the new Water Plan is in force.

The entire Guadiana river basin presents two clearly contrasted domains. The middle and lower stretches of the basin are characterized by the predominance of surface water use, regulated through big dams. The Upper Guadiana Basin is marked by the constant interaction between a complex system of interconnected aquifers and the surface water bodies and ecosystems (Fornés et al., 2000). The semi-arid climate presents a high seasonal and annual variability of precipitations (415 mm/year in average on the period 1941-1991, Bromley et al., 2001).

Agriculture plays a major role in the local economy. 'La Mancha' region is globally significant as the largest vineyard area in the world with 400,000 ha. That is around 2% of the global vineyard area. Vines have been almost exclusively rain-fed until the ban on vine irrigation was lifted in the 1990s. Regarding vegetables, melon, garlic, and onion located in the WMA represent, respectively, 34%, 20% and 14% of the national outdoor cultivated areas for the year 2008 (MAGRAMA, 2013). Cereals, the third most important crop in the WMA (mainly barley), still constitutes a large proportion of both rain-fed and irrigated land, whereas water intensive crops such as alfalfa, maize or beetroot have reduced substantially over the last few years. Only groundwater use from agriculture is considered since it represents 95% of the withdrawals from the aquifer (CH Guadiana, 2007).

Intensive groundwater withdrawals, almost entirely for irrigation purposes, have produced deep socio-economic changes, but also substantial environmental impacts by lowering the water table and disrupting the flows towards the Tablas de Daimiel wetland. This has generated a conflictive situation between groundwater use and the conservation of a valuable ecosystem. It became an emblematic case, which has been analysed in numerous studies (Llamas, 1988; Fornés et al., 2000; Bromley et al., 2001; Custodio, 2002; López-Gunn, 2003; Blanco-Gutiérrez et al., 2010; Martínez-Santos et al., 2010; Varela-Ortega et al., 2011, among others).

VII.1.2 Evolution of the use of groundwater since the 1970s

Groundwater-based irrigation boomed in the 1970s. Until then, agriculture was mainly rain-fed, with an amount of groundwater withdrawals of 50 hm^3, which were withdrawn thanks to traditional water wheels (*norias*) that could access groundwater thanks to the high water table (Bromley et al., 2001). The introduction of pumped boreholes led to a dramatic rise in the irrigated area, which reached 600 hm^3 in the year 1988 (Figure 7.2). Some trends on the evolution of the conditions of groundwater use can be distinguished:

- The irrigated area and withdrawals have risen proportionally over the period 1974-1989.
- A steep fall in the groundwater level is observed from 1990 to 1995. It is associated to a major drought and implied higher pumping costs, which may explain the reduction and stabilization of the irrigated area from 1990 to 1995. A public plan to reduce withdrawals (see Section 1.4 of this chapter) may have contributed also to this reduction.
- Meanwhile, a progressive decoupling of the irrigated area and the withdrawals took place from 1993 to 1997. This can be linked to various reasons, such as the change in irrigation techniques (mainly from surface to sprinkler irrigation), the shift to less water consuming crops, or the introduction of supplementary and deficit irrigation as a

response to the rising pumping costs and a changing institutional setting with a public plan paying for withdrawals reduction.

- A rise in water levels took place from 1996, which corresponds to a period with more intense rainfall. It has been followed by an increase in the irrigated area and the amount of withdrawals, which confirms the potential influence of pumping costs. The authorization of the irrigation of vines from 1995 should also be taken into account.
- The decoupling of withdrawals from the irrigated area implies that, although the irrigated area reached its maximum value in the 2000's, with 140,000 ha, the amount of withdrawals is slightly higher than half of the maximum withdrawals reached in 1988.

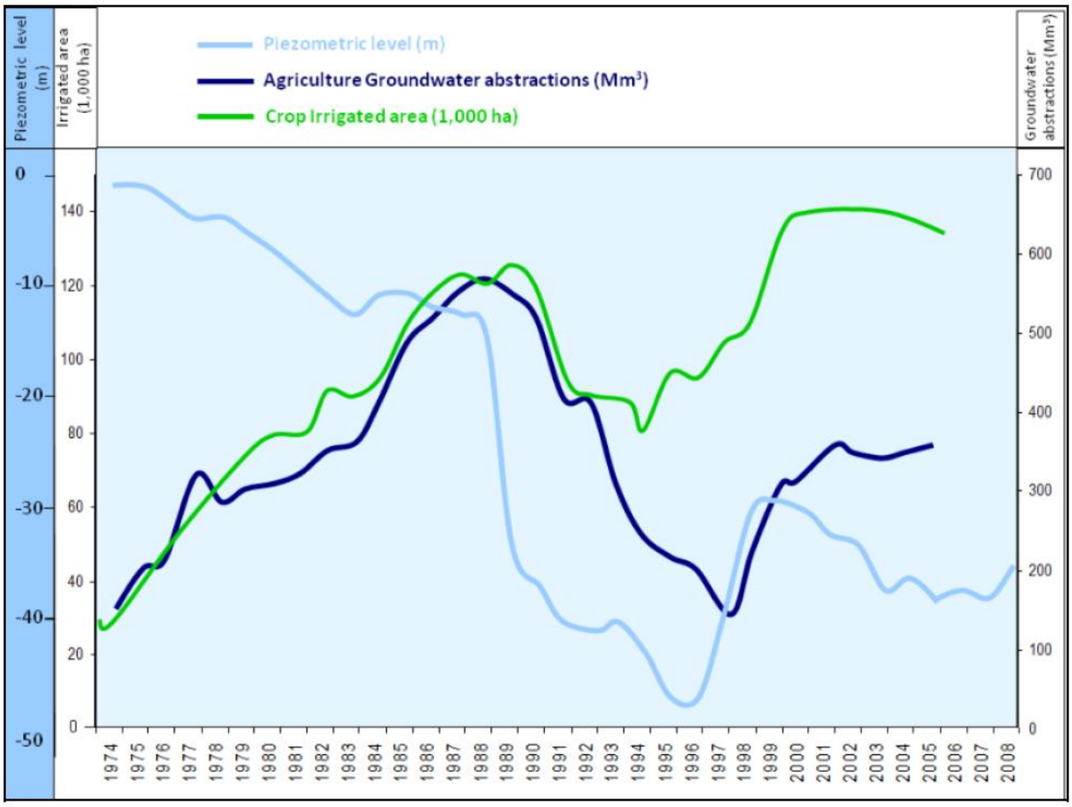

Figure 7.2. Evolution of piezometric level, irrigated areas and groundwater withdrawals in the Western Mancha Aquifer (1974-2008). Source: Zorilla (2009)
1 Mm^3 = 1 hm^3

The more recent evolution of the groundwater level is presented on Figure 7.3 (from De la Hera & Villarroya, 2013). It can be observed that another rise in the piezometric levels took place in the period 2010-2013, which has been exceptionally humid, with the groundwater table reaching almost its natural elevation and various groundwater resurgences in the area have been reported.

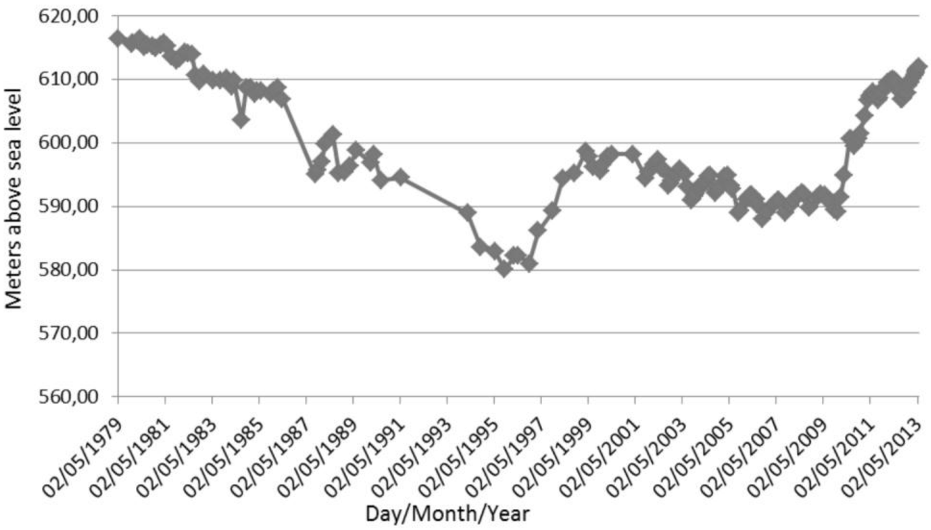

Figure 7.3. Evolution of piezometric level close to the Tablas de Daimiel (1979-2013). Source: De la Hera & Villarroya (2013)

VII.1.3 Hydrogeological setting and impact of pumping on Tablas de Daimiel

- <u>Hydrogeological setting</u>

The basement of the area is constituted by Jurassic and Cretaceous pervious carbonate materials in the East and of almost impervious Palaeozoic and Triassic materials in the West. The WMA is made of limestone and marl materials and presents an average thickness of 200 m (Figure 7.4). It is an unconfined aquifer with a hydraulic conductivity ranging from below 1 m/day near the system boundaries to nearly 100 m/day in its central area (Martínez-Santos et al., 2008).

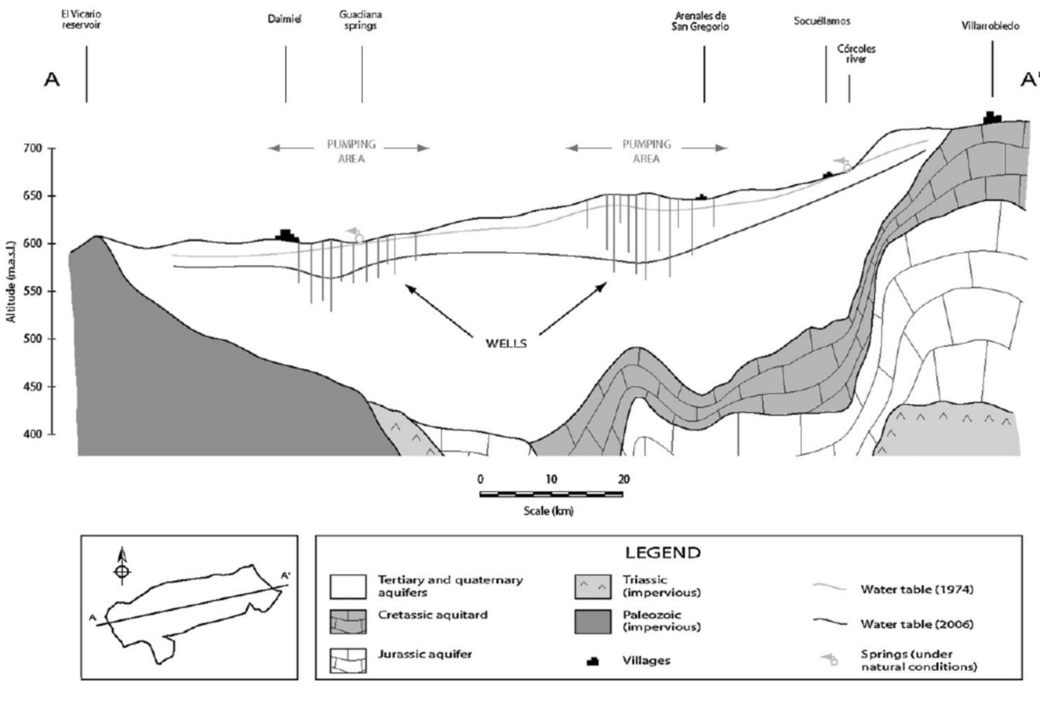

Figure 7.4. Hydrogeological section of the Mancha Occidental system. Source: Martínez-Santos & Martínez-Alfaro (2010)

- <u>Change in inflows and outflows and withdrawal limit objectives</u>

The average value of the natural inflows from rainfall has been estimated at 210 hm^3 per year (Table 7.1). Other sources of inflows are river streams and lateral transfers from other aquifers. In the natural state, groundwater outflows were going to rivers, wetlands, and evapotranspiration. Nevertheless, when groundwater levels fall up to the point the flows to the rivers and wetlands are cancelled, withdrawals are the only outflows. Low levels trigger also an increase in inflows from lateral transfers (10 hm^3) and direct evapotranspiration from the groundwater table do not occur anymore, which results in an additional 128 hm^3 'appropriated' by withdrawals as compared to the natural situation. On the other hand, the infiltration from rivers is reduced, potentially because of the effect of the neighbouring groundwater bodies. Finally, the flows changed from going toward the natural area of discharge on the western border of the aquifer, to the central area where the majority of the withdrawals take place.

Table 7.1. Modification of the water balance of the groundwater bodies 'Mancha Occidental I', 'Mancha Occidental II' and 'Rus-Valdelobos'. Source: adapted from Martínez-Cortina et al. (2011).

	Natural state		1950s		2009	
	In	Out	In	Out	In	Out
Inflows from rainfall	210	0	210	0	210	0
Evapotranspiration	0	128	0	116	0	0
Rivers	105	249	104	200	66	0
Lateral transfers[a]	62	0	62	0	72	0
Withdrawals	0	0	0	60	0	348
Total	377	377	376	376	348	348

[a] Without the internal lateral transfer between the three groundwater bodies that covers the area of the WMA (Mancha Occidental I', 'Mancha Occidental II' and 'Rus-Valdelobos'). Even if the borders of the three groundwater bodies do not match exactly with the WMA, the figures can be considered valid for the WMA. See Appendix 5 for a detailed balanced of the three groundwater bodies.

This evolution of the water balance illustrates why a dynamic approach must be introduced when considering the definition of availability of resources as explained in Chapter 3 on the groundwater allocation. According to Martínez-Cortina et al. (2011), the good functioning of wetlands and other associated water bodies would require reducing the withdrawals to 60 hm^3, i.e. the situation of the 1950's. IGME (2010) defined a threshold of 101 hm^3 and recognizes that higher withdrawals may not produce a significant improvement of the state of the Tablas de Daimiel wetland (Table 7.2). Nevertheless, the final Guadiana Water Plan recommends annual withdrawals no higher than 222 hm^3, including 13% of return flows, i.e. a WF of 200 hm^3 (CH Guadiana, 2013). This value is higher compared to the initial value of 194 hm^3 presented in the draft version of the Plan (CH Guadiana, 2011). This has been mainly justified by the use of a more accurate hydrogeological model and the integration of the more intense rainfall of the period 2010-2013. However, this value is questionable if the objective is to maintain enough flows to the wetlands (Table 7.2). More considerations of these different objectives are presented in the discussion section of this chapter (Section 4.5).

Table 7.2. Value of pumping and related environmental state. Source: adapted from IGME (2010).

Pumping for the WMA	Pumping for the whole Upper Guadiana	Implication for environmental state
230 hm^3	275 hm^3	From the current situation, it would mean a noticeable increase in groundwater levels, with, however, a limit if this pumping intensity is maintained. From the point of view of the resurgence of natural outflows in certain areas and wetlands recovery, this first step would not be sufficient, and a later more restrictive objective should be established for environmental and hydrological improvement.
175 hm^3	200 hm^3	Discharge could start in areas with the lowest topography such as *los Ojos del Guadiana*, <u>at least seasonally during series of humid years</u>.
101 hm^3	125 hm^3	Discharge and wetland areas would increase and the system would be <u>less sensitive to the seasonality of the discharge</u>. This figure can be considered as a reference for a notable environmental and hydrological recovery.

Highlights by the author of the thesis.

- <u>Impact of pumping on the wetland ecological functions and services</u>

Regarding the area of the Tablas de Daimiel, a direct consequence of the disconnection with groundwater is the reduction in flooded area (Figure 7.5). It presents also more intense seasonal variations. As groundwater flows are essential during the driest months, it potentially affects the ecological state of the wetland. Moreover, it means that the hydrological regime is now entirely governed by surface water flows to the wetland. A consequence of this situation is that during dry periods, the wetland receives almost no flows, and the flooded area can be almost zero for several years (e.g. 1993-96 or 2006-10).

These conditions have triggered an intense desiccation process on much of the area of the wetland, which have deeply affected its biodiversity and ecological functions and the associated services (e.g. provision of raw material or medicinal plants, fisheries, CO_2 storage capacity) and value (e.g. aesthetic or existence values) (De la Hera & Villarroya, 2013).

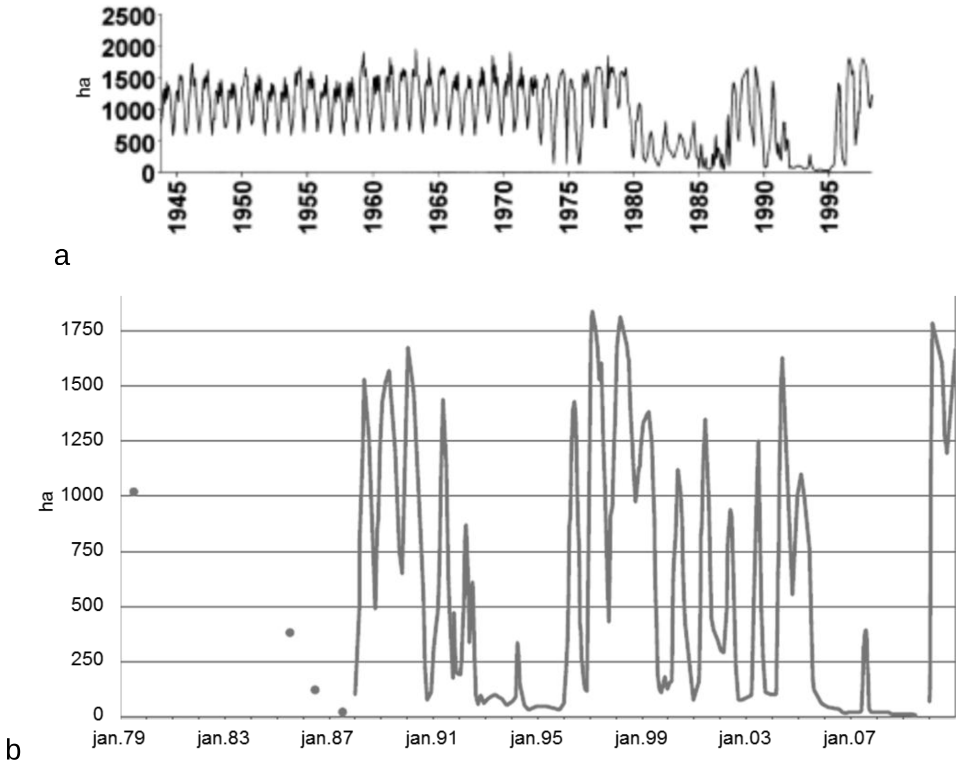

Figure 7.5. Flooded area of the Tablas de Daimiel. Source: (a) Alvarez-Cobelas et al. (2001) (modeled before 1995) (b) Castaño-Castaño (2013).

This situation has also generated underground peat combustion (Llamas, 1988; Fornés et al., 2000), which implies problems regarding water quality, in addition to the more direct hazard of fire propagations (Moreno et al., 2010). These ecological damages justified the decision of transferring water from the Tagus-Segura transfer to the area (Table 7.3). However, not all this water arrived at the Tablas de Daimiel as it infiltrated or was 'stolen' in its way to the wetland.

Table 7.3. Annual transfer to the Tablas de Daimiel wetland from the Tagus-Segura transfer. Source: De la Hera & Villarroya (2013).

Year	Amount of water transferred (hm^3)	Year	Amount of water transferred (hm^3)
1988 - 1989	12.1	2001 - 2002	20
1989 - 1990	13.3	2002 - 2003	25
1990 - 1991	15.8	2003 - 2004	9.4
1991 - 1992	17.8	2004 - 2005	0
1992 - 1993	6.5	2005 - 2006	0
1993 - 1994	0	2006 - 2007	10
1994 - 1995	15	2007 - 2008	0
1995 - 1996	0	2008 - 2009	20
1996 - 1997	30	2009 - 2010	1.5
1997 - 1998	0	2010 - 2011	0
1998 - 1999	0	2011 - 2012	0
1999 - 2000	26	2012 - 2013	0
2000 - 2001	20		

Furthermore, the decrease in the groundwater level has generated an unsaturated layer under the Tablas de Daimiel (Aguilera et al., 2013). It means that the hydrological functioning of the area has been inversed, from receiving groundwater flows to being an area of recharge. This resulted in the potential contamination of the aquifer (Moreno et al., 2010).

Even if this chapter focuses on problems linked to the quantitative situation, it is important to notice that not only groundwater withdrawals have detrimental impacts on the ecological quality of the wetland. For instance, discharges from industrial and non-treated urban effluents have also been reported (De la Hera & Villarroya, 2013).

VII.1.4 Issue of illegal groundwater use and public authorities successive plans

The clear drop in groundwater levels and the related impacts on Tablas de Daimiel led to the provisional administrative declaration of overuse in the year 1989, finally confirmed in 1994. This declaration included, in addition to the creation of groundwater user groups, the closure of the groundwater resource to new users after 1994, and the establishment of an annual cap on withdrawals through the reduction in existing water rights (Table 7.4). This adds to other implications of the Water Act of 1985, such as the prohibition to modify the characteristics of the well (deepen or change the location).

Notwithstanding, the consequences of these measures were softened by the introduction of a program popularly known as the ‘Wetlands Plan’ (*Plan de Humedales*), for the period 1993-2006. The plan consisted of compensatory payments (partially coming from EU funds) in lieu for the temporal reduction in groundwater withdrawals (Fornés et al., 2000; Varela-Ortega et al., 2011). In addition to its cost (250 million euro in total), one of the major drawbacks of this plan is not having tackled directly the issue of non-authorized use. During this period, controls and sanctions were not effective and the few attempts to close illegal wells triggered a strong opposition from local lobbies (Martínez-Santos et al., 2008; Blanco-Gutiérrez et al., 2010).

Table 7.4. Restrictions from the Withdrawal Regulation Plan for the years 2007 to 2009. Source: CH Guadiana (2010).

Crop	Farm size	Initial Right	Withdrawal Regulation Plan 2007 – 2008	Withdrawal Regulation Plan 2009
General	<30 ha	4,280 m^3/ha	2,640 m^3/ha	2,000 m^3/ha
	30-80 ha		2,000 m^3/ha	
	>80 ha		1,200 m^3/ha	
Vine			800 -1,000 m^3/ha	1,500 m^3/ha

[a] Depending on the annual rainfall.

Both legal and illegal uses for irrigation coexist in this area. Therefore, as reflected by Blanco-Gutiérrez et al. (2011), any policy intended to solve environmental problems in this aquifer should tackle both authorized and non-authorized water uses. For this purpose, getting new insights on the actual extent of the informal economy in the WMA appears essential.

In response to this situation, a new plan, effective from 2008 to 2012, the Special Upper Guadiana Plan (SUGP) (CH Guadiana, 2007), included the problem of non-authorized use in the core of its actions[60]. The main measure is the purchase of groundwater rights to be reallocated for the recovery of the aquifer and associated wetlands and for the regularisation of non-authorized users on social grounds (small professional farms) irrigating vine and vegetables. The price of the purchase has been fixed to a maximum of 10,000 €/ha, with a general rule that 70% of rights purchased to go to environmental restoration and 30% to the 'social' regularization of illegal wells. The plan intended to regularize 17,500 ha for vine and 1,700 ha for vegetables with, for vine, the condition that for regularized users the allowed volume of extraction is 700 m^3/ha rather than 1,500 m^3/ha (existing legal vine rights) (López-Gunn et al., 2012). Another important measure in the SUGP is the improvement in the control of extractions through remote sensing technologies and a network of metering devices[61].

VII.2 Specific methodological considerations

VII.2.1 Sources for data on irrigated area

The analysis focuses on the following crops: vine, wheat, barley, melon, garlic, onion, and pepper, which represent around 95% of withdrawals (Consorcio del Alto Guadiana, 2011). The remaining irrigated crops are considered as a single group.

Irrigated areas by crop were obtained from two main data sets:

- Data at municipal scale from the Castilla-La-Mancha regional government for the years 2000 to 2009 (JCCM, 2010);
- Data from remote sensing generated by the Consorcio del Alto Guadiana for the period 2007 to 2009 (from 2004 in the case of cereals) (Bea et al., 2009; Consorcio del Alto Guadiana, 2011).

[60] The SUGP has been abrogated in 2012, for a variety of reasons, including its huge cost but also for political reasons. The new measure is a water market directly between the farmers (Ley 11/2012; BOE, 2012). In the discussion section of this chapter, the focus is on the SUGP, because it was the plan in force, when the major part of this chapter was researched. However, many of the issues that are commented relatively to the SUGP are also important points to take into account for water markets.

[61] It is also the main measure of the draft river basin Water Plan (CH Guadiana, 2011) to achieve the objectives required under the EU Water Framework Directive.

The reliability of data sets is essential and the reason for using two sources is because the data from JCCM (2010) was generated from varied sources (farmer declarations for subsidies, cooperative statistics, direct inventories, etc.) with potential high uncertainty. This can be appreciated when compared with more recent remote sensing data for the period 2007-2009, which can be considered more accurate (as shown in the Section 3.1 of this chapter). This was also confirmed in expert interviews, which recommended the use of JCCM irrigation data for the early part of the decade but recognized that these were not really accurate in the later years.

VII.2.2 Estimation of the non-authorized area

Regarding data on non-authorized irrigated area, the SUGP is the first official document that includes an evaluation of non-authorized areas (CH Guadiana, 2007). In this document, non-authorized irrigated area is divided into three main categories: 'herbaceous crops', which refer to vegetable crops; 'woody plants', which correspond mainly to vine, and 'other', which refers to cereals (Table 7.5)

Table 7.5. Non-authorized areas for the main type of crops and their share.

Category of crop (designation by the SUGP)	Non-authorized area (ha) (CH Guadiana, 2007)	Average irrigated area 2000-2006 (ha) (JCCM, 2010).	% Non-authorized / Total irrigated area.
Vine ('woody plants')	27,203	40,982	66.4
Vegetables ('herbaceous crops')	7,456	15,834	47.1
Barley, wheat, maize, sunflower, beetroot, alfalfa and rapeseed ('other')	17,723	69,516	25.5
Total	52,382	126,332	41.5

The text of the SUGP (CH Guadiana, 2007) indicates that the estimation of non-authorized irrigated areas that are presented has been based on a variety of data sets like an inventory for the period of 2001-2005 and remote sensing data for the years 2005 and 2006[62]. Thus, we calculate the percentage of non-authorized area on the basis of the average irrigated areas of the crops included for the period 2000-2006 based on JCCM (2010) (Table 7.5). This percentage is supposed the same for the whole period of study.

[62] Non-authorized activity included in the SUGP only corresponds to areas with no irrigation permits. Other types of infractions, related to compliance with the restrictions imposed after the overexploitation declaration like pumping more water than quota, or deepening the well, were not included.

VII.3 Results

VII.3.1 The water footprint of the Western Mancha Aquifer

- <u>A first overview based on the irrigated areas</u>

A first glance on the evolution of the WF per type of crop can be obtained by analysing the share and evolution of irrigated areas[63]. Between the years 2000 and 2009, the total cultivated area is fairly stable in the WMA (around 42,000 ha). However, there is an important increase in the proportion of the irrigated area, at the expense of the rain-fed area. The drop in rain-fed area is mainly ascribed to the reduction in vine rain-fed area by 60,000 ha (Figure 7.6).

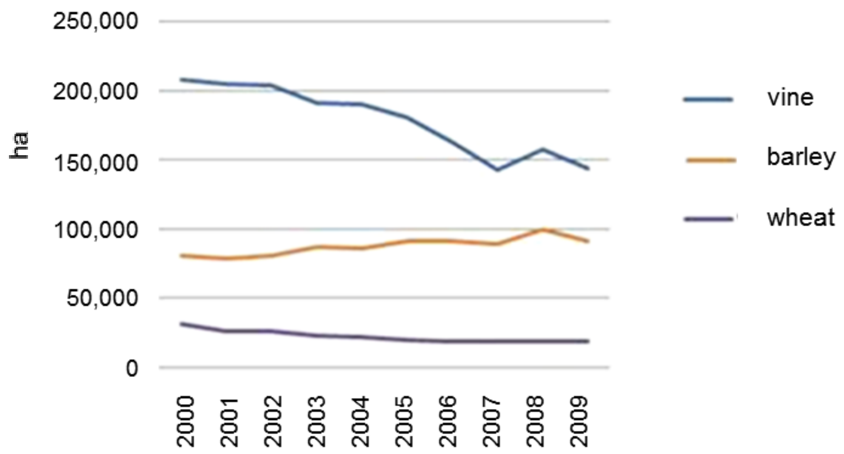

Figure 7.6. Rain-fed area by crop (2000-2009). Source: own elaboration based on JCCM (2010).

The irrigated area of vegetables is stable, around 15,000 ha, and data from JCCM (2010) and remote sensing are in concordance (Figure 7.7). For cereals, values obtained based on data from remote sensing are much lower than when using JCCM (2010), particularly for years with more intense spring rainfall since the irrigated cereal area is correlated with spring rainfall intensity (Bea et al., 2009) (Figure 7.7). For instance, the irrigated area in the year 2007 (average to humid year) is around 8 times lower than in 2009 (a dry year).

Thus, in addition to the variation in crop water consumption, which is usually contemplated in WF assessments, e.g. in Aldaya et al. (2010b), the variation in irrigated area is another important factor that influences the value of the WF in relation with the climatic conditions. However, even for the driest years, the value given by JCCM (2010) is much higher than the value obtained by remote sensing. It means that using the data from JCCM (2010) would result in an overestimation of the WF for cereals. This difference may be related to the way data are produced by JCCM (2010), which implies a difficulty to distinguish between irrigable (i.e. the

[63] The WF is obtained multiplying the crop areas by the crop water consumption by ha (CWC). It means that the uncertainties regarding CWC are reported on the value of the WF. Not including CWC allows disregarding the variation of the annual climatic factor, and other uncertainties regarding CWC data. Additionally it allows distinguishing the evolution of the WF that is linked to the evolution of crop areas.

area equipped for irrigation but that may not be irrigated because of the climatic and economic context) and irrigated area (area effectively irrigated).

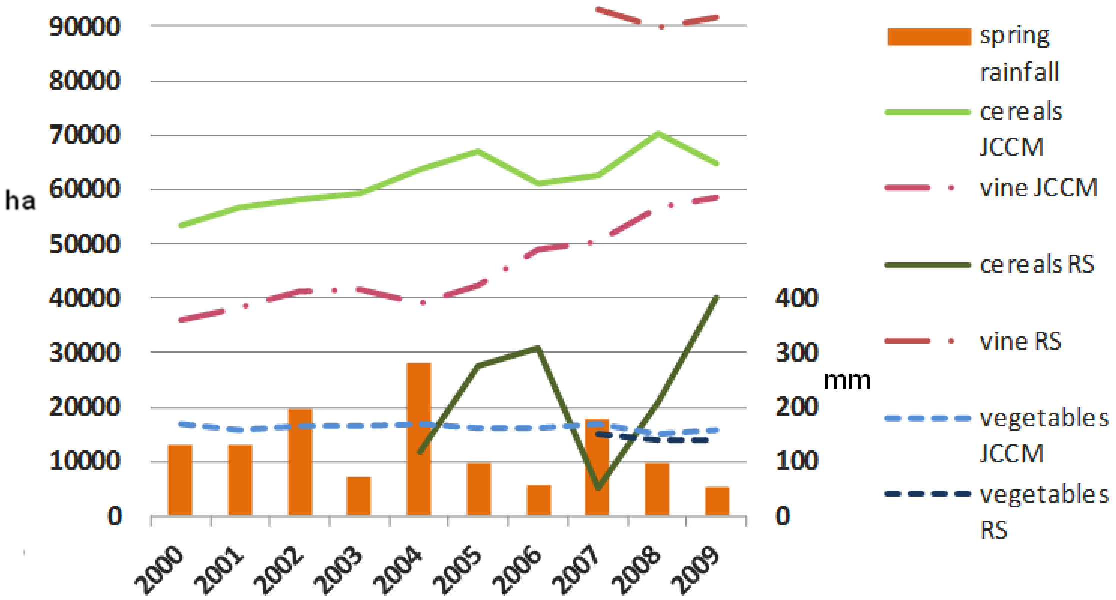

Figure 7.7. Irrigated area by crop according to the two data source and spring rainfall for the period 2000-2009.
JCCM: based on JCCM (2010); RS: based on Bea et al. (2009) and Consorcio del Alto Guadiana (2011)

In the case of vine, there is also a substantial difference between the two data sets: remote sensing data presents an area of around 90,000 ha for the period 2007- 2009, meanwhile for data from JCCM (2010) this area only reaches 58,000 ha during the same period. It means that the rise in irrigated vine area since the year 2000 is steeper than the official data shows (JCCM, 2010), with an increase of around 55,000 ha. Most of this increase is non-authorized, which may explain why it is not reflected by JCCM (2010).

- Evolution of the water footprint

The total WF of the WMA reached a value of 332 hm^3 in 2009 (Figure 7.8), meaning that the objective set by the draft Water Plan of the Guadiana (200 hm^3) is exceeded by around 130 hm^3. Between the year 2000 and the years 2007 and 2008, the total WF is fairly consistent, around 300 hm^3. The rise in irrigated vine area seems compensated by the fall in the irrigated areas of the other categories of crops. Significantly, the WF of vine grew by more than 110 hm^3 during the 2000's. It represented more than 60% of the total WF in 2008. However, farmers who grow cereals maybe did not fully use their rights in 2007 and 2008, which contributed to the WF reaching 330 hm^3 in 2009, because a drier spring made it necessary to irrigate more area[64].

[64] This 'flexibility' of cereals (blue water being applied when green water is not sufficient) is not observed for vine and vegetables, which must be irrigated during summer with a more or less fixed quantity each year, because it hardly rains in this season.

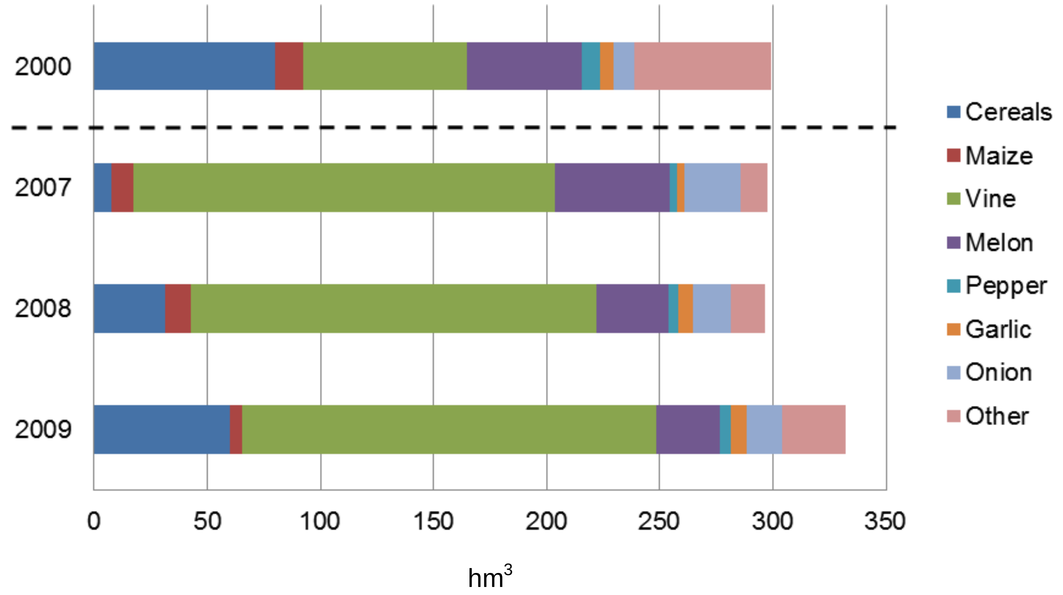

Figure 7.8. The water footprint of the Western Mancha Aquifer for the year 2000 and for the period 2007-2009 (hm^3).
The data for the year 2000 is obtained from JCCM (2010). The other years are based on remote sensing.

The virtual water content of the main crops of the WMA is presented Table 7.6.

Table 7.6. Virtual water content and economic and social water productivity by crop (2008).

	Barley	Maize	Vine	Melon	Pepper	Garlic	Onion	Average
Crop water consumption (m^3/ha)	1,500	6,600	2,000	4,300	4,500	2,250	6,200	-
Rain-fed yield (kg/ha)	2,667	-	5,700	-	-	-	-	-
Irrigated yield (kg/ha)	3,667	11,000	15,000	32,800	40,000	7,600	34,000	-
Market value (€/kg)[a]	0.13	0.155	0.19	0.20	0.22	1.05	0.12	-
Market value (€/kg)[b]	0.17	0.18	0.20	0.42	0.84	1.26	0.16	-
Virtual water content (overall) (L/kg)	60	-	87	-	-	-	-	-
Virtual water content (irrigated) (L/kg)	409	509	133	131	113	296	182	
Direct economic value I (€/m^3)[c]	0.09	0.24	0.88	1.26	1.70	3.04	0.47	0.85
Direct economic value II (€/m^3)[d]	0.32	0.30	1.43	1.53	1.96	3.55	0.66	1.28
Rain-fed labour (jobs/ha)	0.013	-	0.065	-	-	-	-	-
Irrigation labour (jobs/ha)	0.018	0.029	0.09	0.125	0.35	0.2	0.2	-
Direct employment I (jobs/hm^3)[c]	3.3	2.9	12.5	14.0	63.3	60.0	21.8	13.6
Direct employment II (jobs/hm^3)[d]	12.0	5.2	45.0	29.1	77.8	88.9	32.3	38.8

[a] Source: DGAACM (2010) / [b] Source: MAGRAMA (2013) (i.e. national value, presented to illustrate the difference between local data and national data) / [c] Considering the value generated by the alternative rain-fed activity / [d] Ratio between market value and crop water consumption.

The values range from 113 L/kg (pepper) to 509 L/kg (maize). 'Overall virtual water content'[65] can be calculated for grapes and cereals (barley and wheat) as they are also rain-fed crops: 87 L/kg and 60 L/kg respectively.

VII.3.2 Socio-economic indicators linked to the water footprint

The main socio-economic aspects associated with water consumption by irrigation in the WMA are analysed for one year (2008), which represents an average to low rainfall year (Table 7.6). It is important to take into account that the following analysis is based on the direct economic value and employment considering the surplus generated by agriculture compared to the potential rain-fed production that would have taken place if no water was available. That is rain-fed barley instead of irrigated cereals and rain-fed vine instead of irrigated vine or vegetables (see Chapter 6 on the case studies methodology).

The results show that garlic and pepper have the highest average economic and social returns per cubic meter consumed. They are followed by melon, vine and onion, whereas cereals present the lowest returns with 20 times less jobs associated and 30 times less gross economic return by unit of water consumed, compared to the most productive vegetables. The first value (Direct employment) can be explained since the difference in labour requirements between irrigated and rain-fed cereals is small, as compared to the shift from the baseline rain-fed crops (vine and barley) to vegetables, which are much more labour-intensive. The second value (Direct economic value) is also exceptionally low for cereals because, in the reference year, the difference in yield of the irrigated and rain-fed cereals is low compared to other years (Figure 7.9). Therefore this underestimates the value of water in the irrigation of cereals. However, the maximum difference is only 2.5 times, which does not have a great influence on the final results.

When comparing the WF with the jobs and gross economic returns generated for the whole study area (Table 7.7), it appears that irrigated cereals, despite representing 11% of the WF, only contribute to 2.5% of employment and 1% of the gross revenue in the area. Meanwhile, vegetables which represent 20% of the WF contribute to 38% of the employment and 30% of the gross revenue. This is similar to vine; with a share of 60.5%, it represents 59% in terms of labour and 63% for gross revenue. However, when compared to the whole agricultural economy of the area (both irrigated and rain-fed), the share of these crops is lower, with the sum of irrigated vine and vegetables reaching 26% of the employment and 41% of the gross revenue.

[65] It refers to the average virtual water content of a crop coming from the area, whether it is irrigated or not. It is obtained by the ratio of the whole production of the area (rain-fed and irrigated) on the total blue WF of the crop (see Chapter 6 on methods).

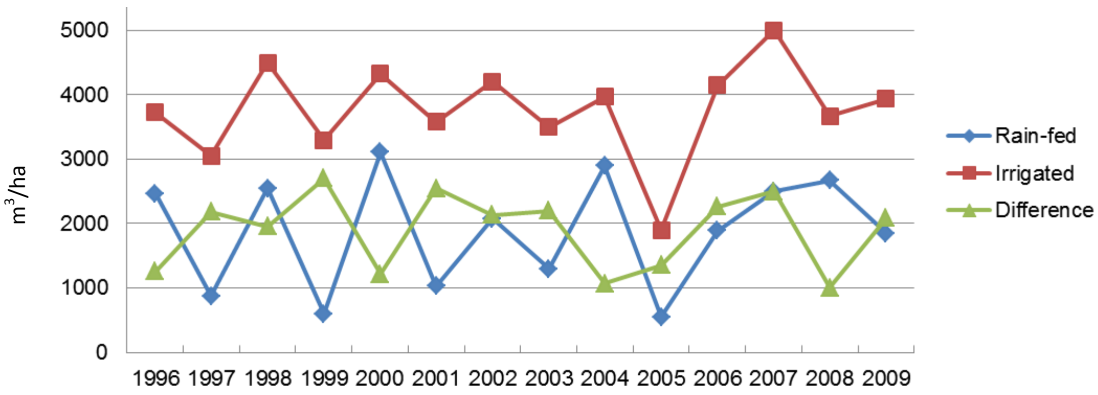

Figure 7.9. Yield of barley on the period 1996-2009 for rain-fed and irrigated areas. Source: own elaboration based on MAGRAMA (2013)

Table 7.7. The water footprint and contribution in terms of jobs and gross revenue (2008).

	Cereals	Maize	Vine	Melon	Pepper	Garlic	Onion	Total vegetables	Other	Total
Total value										
Water footprint (hm^3)	31.4	11.1	179.3	32.0	4.3	6.3	16.8	59.4	15.1	296.3
Employment I (number of jobs)[a]	105	27	2,241	446	238	380	366	1,430	206	4,010
Employment II (number of jobs)[b]	377	49	8,069	929	293	563	543	2,327	586	11,408
Gross revenue I (million euro)[a]	2.7	2.3	158.4	40.3	6.4	19.3	8.0	73.9	12.8	250.2
Gross revenue II (million euro)[b]	10.0	2.9	255.5	48.8	7.4	22.5	11.1	89.7	19.3	377.4
% / Irrigated agriculture										
Water footprint	10.6	3.8	60.5	10.8	1.4	2.1	5.7	20.0	5.1	100
Employment I[a]	2.6	0.7	58.9	11.7	6.3	10.0	9.6	37.6	5.4	100
Employment II[b]	3.3	0.4	70.7	8.1	2.6	4.9	4.8	20	5.1	100
Gross revenue I[a]	1.1	0.9	63.3	16.1	2.6	7.7	3.2	29.5	5.1	100
Gross revenue II[b]	2.6	0.8	67.7	12.9	1.9	6.0	2.9	23.8	5.1	100
% / Total agriculture (irrigated and rain-fed)										
Employment I[a]	0.7	0.2	15.9	3.2	1.7	2.7	2.6	10.2	1.5	28.5
Employment II[b]	1.7	0.2	37.2	4.3	1.3	2.6	2.5	11	2.7	52.6
Gross revenue I[a]	0.5	0.4	27.8	7.1	1.1	3.4	1.4	13.0	2.2	43.9
Gross revenue II[b]	1.8	0.5	44.8	8.5	1.3	3.9	1.9	15.7	3.4	66.1

Employment and gross revenue for the category ‘other’ are obtained considering the average value of the direct economic value and direct employment / [a] Contemplating the value generated by the alternative rain-fed activity / [b] Ratio between market value and crop water consumption.

VII.3.3 Non-authorized use: water footprint and economic valuation

Based on the data from the SUGP (CH Guadiana, 2007), the percentage of non-authorized use by type of crop has been estimated (Table 7.5). For the year 2008, it means that 57.4% of the irrigated area have no water permits, corresponding to a WF of 155 hm^3, whereas 115 hm^3 corresponds to authorized use (Figure 7.10). In terms of economic value and direct employment, non-authorized use generated 215 million euros and 2.460 jobs, while only 143 million euros and 1,800 jobs were associated with authorized use.

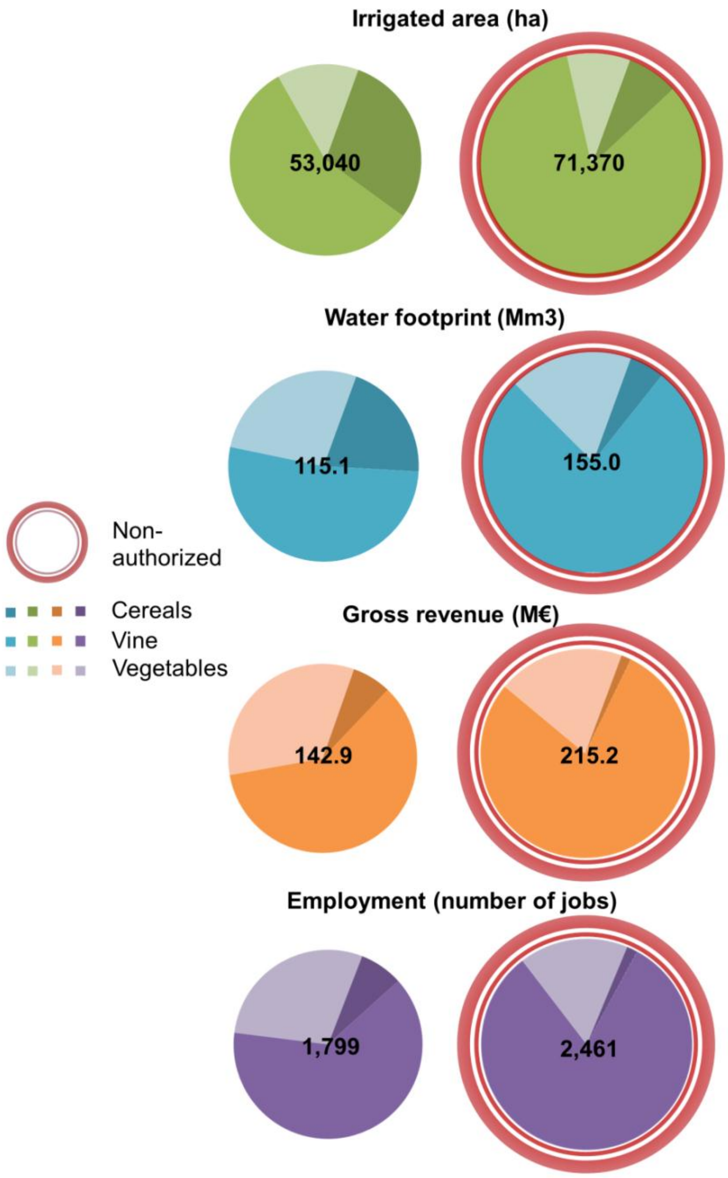

Figure 7.10. Irrigated area, water footprint, gross revenue and employment for authorized and non-authorized groundwater use in the Western Mancha Aquifer (year 2008).
These figures only include cereals, vine and vegetables (melon, pepper, garlic, onion). 1 Mm^3 = 1 hm^3

If we zoom into the non-authorized water consumption, it appears that 77% of the non-authorized WF is related to vine, which implies the highest percentage of jobs and gross revenue related to non-authorized activity (73% and 79% respectively). This is followed by vegetables (18% of the non-authorized WF) and finally cereals (5%). It means that non-authorized water consumption in two crops, vine and vegetables together, generates around 56% of gross revenue and 57% of employment related to irrigated agriculture in the area.

With regards to non-authorized use in relation to the Direct Economic Value (DEV) and Direct Employment (DEM), it appears that the share of non-authorized activity is higher for the crops that present a higher DEV and DEM (vine and vegetables as compared to cereals). Interestingly, the average economic value of a ‘legal drop’ is higher than for an ‘illegal drop’ (0.87 €/m^3 versus 0.83 €/m^3). This is due to the fact that the share of vegetables, which present a higher DEV, is higher in authorized activity than in non-authorized activity. For the DEM, the figure is similar for both authorized and non-authorized use (14 jobs/hm^3).

Finally, following the WF under the ‘perspective of consumption’, the content in ‘virtual illegal water’ of the main agricultural products of the WMA can be estimated (Figure 7.11). On the basis of these results, the ‘virtual illegal water content’ of a generic bottle of vine of 75cL from La Mancha is 65L (on an estimated total virtual water content of 98L).

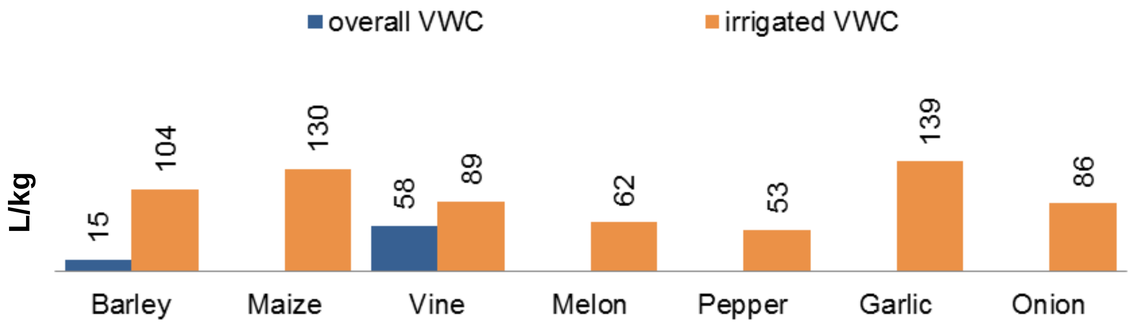

Figure 7.11. Virtual illegal water content of the Western Mancha Aquifer main productions.

VII.4 Discussion

VII.4.1 Extent of non-authorized activity

The share of non-authorized activity is derived from the values for three categories of crops presented by the SUGP (CH Guadiana, 2007). Since these values are not referred to a particular year, the percentage of non-authorized use considers the average irrigated area for the period 2000-2006, based on JCCM data. As the results of the period 2007-2009 show, the irrigated cereal area is overestimated and the vine area underestimated in data from JCCM (2010), particularly based on the fact that non-authorized use might not be reported. This means that the percentages may in reality be higher for cereals and lower for vine. This has to be taken into account when comparing our results to previous estimates.

VII.4.2 Informal economy and the logic of the Special Upper Guadiana Plan

Faced by the anarchy of illegal irrigation, there are repeated claims for the enforcement of existing irrigation rights by the Administration and the closure of illegal wells. However, in the short term, it would result in a large socio-economic impact for the area, as shown by the results of the non-authorized agricultural activity (Figure 7.10). In this context, the application of the existing regulatory system carried a high economic, social and political cost, explaining both its ineffectiveness and the exacerbation of the conflictive situation (Aragón, 2003). Particularly, vine and vegetable irrigation are key for the profitability of small farms (Blanco-Gutiérrez et al., 2011) and is socially equitable as compared to the prevailing repartition of water rights. This could be one of the main factors explaining the conflictive situation. Indeed, the repartition derives mainly from the recognition of groundwater use prior to the Water Act of 1985, a time where irrigation of vine was prohibited. Thus, farmers that are specialized in vine growing did not have sufficient rights, when they were finally authorized to irrigate.

This point may explain why many farmers in the area have been reluctant to engage in a water market, a solution proposed before the implementation of the SUGP: the initial allocation of rights is perceived as unfair and many farmers would not accept to buy water from farmers who had a privileged starting position at the time of rights allocation.

Meanwhile, on the environmental side, in addition to the obligation for the Administration to satisfy the requirements of the EU WFD, environmental concerns from society continued to escalate. These opposite demands explain the logic of the SUGP, as a compromise to regularize part of the non-authorized use of groundwater, while also trying to ensure a general reduction in withdrawals to let more groundwater flow to the wetlands.

VII.4.3 Economic value of groundwater

The SUGP favoured vine and vegetables on the basis of the productivity of these crops. However, when considering water (or land) productivity indicators, some precautions must be taken. This constitutes also a limitation for the indicators used in this thesis.

First, the possibility of irrigating is particularly critical in the driest periods when yields of rain-fed areas fall down. This means that the farmer can maintain an activity in the rain-fed area only thanks to the surplus of revenues allowed thanks to irrigated agriculture. The value of groundwater for agriculture could be higher than what has been considered in this chapter.

Second, regarding the use of water productivity indicators (such as DEV) in order to promote the allocation of water to the more productive uses, a pivotal point is to take into account the marginal value of water and not the average value (Hanemann, 2006). For cereals, the cultivation or irrigation of one supplemental hectare can be considered to produce the same profit as the preceding one, because the market price is determined by world market conditions. Thus, marginal value can be assumed to equal average value (as noticed by Berbel et al., 2011). However, in the case of vine, one supplemental hectare may contribute to reinforce the imbalance between supply and demand and, thus, to lower the market price. Indeed, as shown earlier, the irrigated area rose significantly in the 2000s and the wine sector in the Mancha area is facing a structural crisis, with an increasing over-production over the last few years (Ruiz-Pulpón, 2010). It represents a symptomatic example that water productivity indicators can be misleading. A simplistic interpretation based on economic water productivity would encourage farmers to switch to vine since it generates more jobs and money per drop than other crops, as done by public plans favouring vines irrigation. However, as the market for wine is local, the price is sensitive to over-production, which reduces the profitability of this crop.

In reality, the price received by farmers was subsidized through indirect EU CAP payments destined for the distillation of stock to deal with the problem of over-production, which has permitted to maintain the selling price for farmers. This has now been converted to direct payments for farmers. In the mid-term, the marginal value of water could be null or even negative, since continued over-production makes prices fall and each supplemental cubic meter used to produce more grapes will contribute to this phenomenon. In the short-term however, the crisis attracts subsidies, which represent a financial resource for farmers and the region. Maintaining the profitability of grape through subsidies is therefore a driver for WMA groundwater overuse. Retrospectively, the development of vine irrigation may have also resulted in that rain-fed vine is no longer competitive, making the shift to irrigation necessary (a ‘snowball effect’). Moreover, the vine varieties that are irrigated are different from the varieties traditionally grown rain-fed in the area. They are grown on ‘*espaldera*’ (trellis), in rows that allows mechanised harvest (Plate 7.1), which potentially implies fewer jobs.

The case of vegetables is more difficult to assess, since there is a high variability in the market price for these products from one year to another and even within the same year, depending on the amount of production. If prices are too low, production is sometimes not harvested and this would mean ‘wasting’ the associated WF. Among other factors, capital availability (high production costs) and risk shape the conditions of vegetable production (see Holtz and Pahl-Wolst, 2012). Markets already give strong incentives, as can be seen in the calculated productivities, and any push to reallocate water to vegetable production should take these limiting factors into account and consider the level of demand.

The indicators of DEV and DEM remain valid, when they are not used to promote water reallocation. They present the average value generated by water use for each crop.

Plate 7.1. Irrigated vines grown in '*espaldera*'. Source: The author

VII.4.4 Implications of the Special Upper Guadiana Plan on the water footprint

The main cornerstone of the SUGP is the purchase of water rights to be re-allocated to small, professional farmers (30% of bought rights) and also for environmental purposes (70% of purchased rights). The water rights would be bought from farmers irrigating cereals and redistributed for the irrigation of economically more productive crops like vine and vegetables. The defined criteria have been based on plot size, age and professional status, to grant formal rights to small farms (10 ha on average) belonging to young farmers (up to 40 years old), and which have agriculture as a main source of income.

If not analysed in more depth, the rationale of the SUGP could end up being counterproductive for the WMA, mainly in the case of vine irrigation. In addition, the plan has been criticized for the high costs from buying rights (10,000 €/ha on public funds). In the context of the current economic crisis, this can explain why only part of the reallocation included in the plan (only related to vine) has been effective and why it was finally abrogated in 2012 (Requena, 2011). Figures 7.12 and 7.13 present the effect of the implementation of the SUGP on the WF of the WMA, if fully implemented, or as implemented in June 2011, respectively[66].

[66] The 'June 2011' situation corresponds in fact to the final implementation of the SUGP due to a lack of funding and its subsequent abrogation.

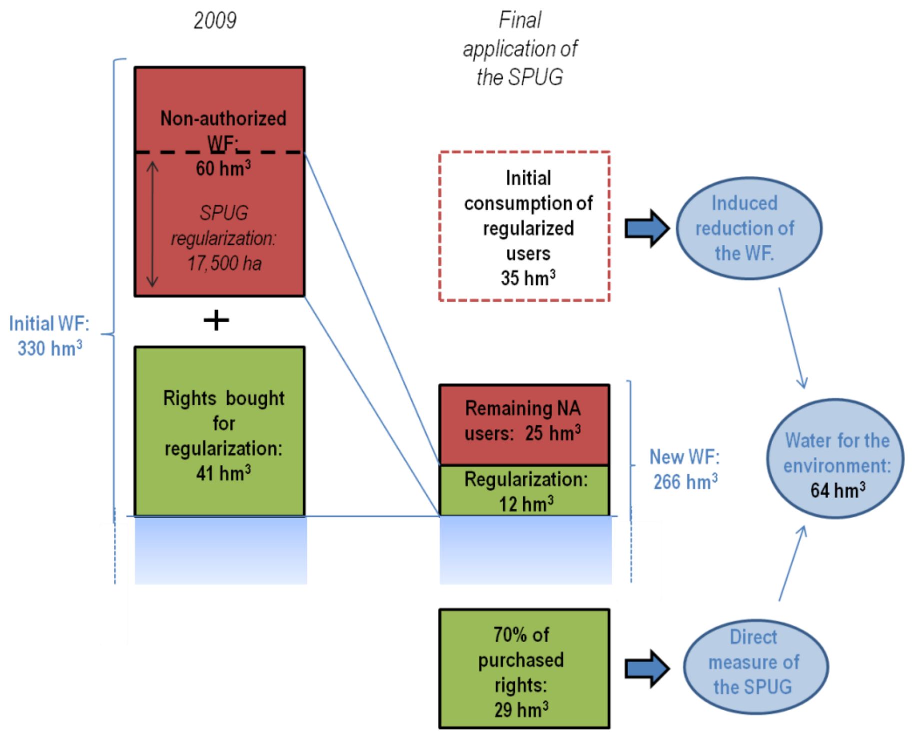

Figure 7.12. Effect of the SUGP implementation (for vine) on the water footprint. Source: López-Gunn et al. (2012)

NA: Non-authorized

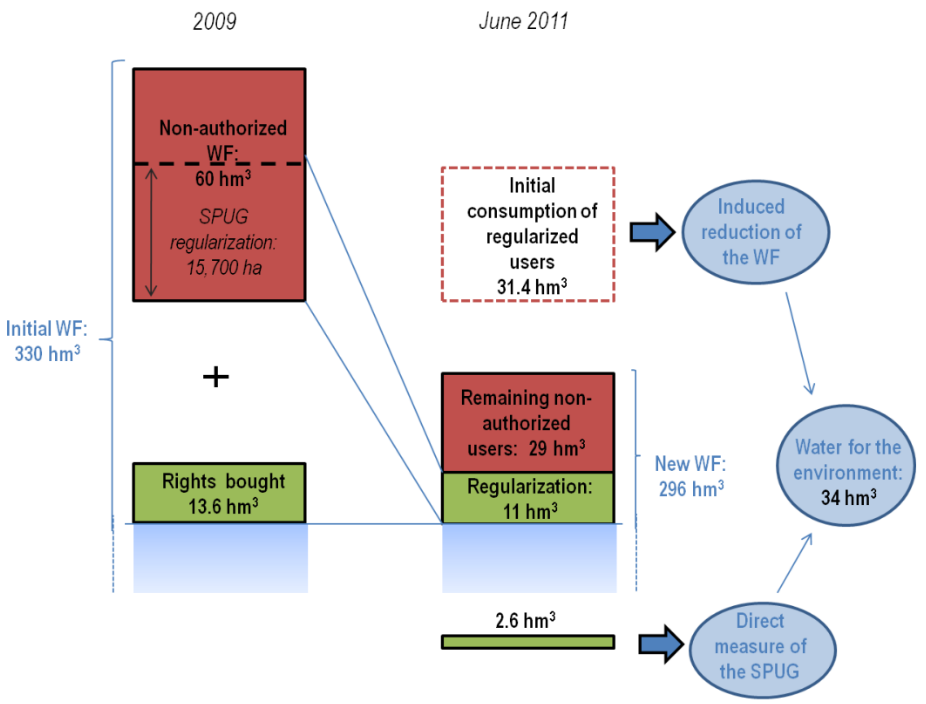

Figure 7.13. Implementation of the SUGP in June 2011 and its effect on the water footprint. Source: López-Gunn et al. (2012).

In the case of vine, the regularization almost reached the planned area. However, the allocation of 70% to the environment as planned in the SUGP was not respected since the big majority of the purchased rights (81%) has been allocated for the regularization of non-authorized users (Requena, 2011). As a consequence, only 2.6 hm^3 have been dedicated directly (i.e. coming directly from the purchased rights) to the environment and this represents less than 1% of the total WF of the WMA, which could lead to question the effectiveness of the SUGP.

However, an important point to consider, as illustrated by the detailed accounting shown in Figure 7.12, is that regularized users – if monitored effectively – will no longer consume their initial use, since their new WF will correspond to the part of the purchased rights that they received. Therefore this initial WF also goes to the environment. Furthermore, vine regularization only allows an extraction volume of 700 m^3/ha (i.e. half the right of legal users and up to three times less than the estimated illegal water use). This volume would ensure the basic water necessities of the plant rather than boosting yield.

This indirect water saving, which has been almost systematically forgotten when the effectiveness of the SUGP has been discussed, represents more water than the direct application of the measure of the SUGP and amounts to 31.4 hm^3 in June 2011. The total lowering of the WF is therefore of 34 hm^3 (10% of the WF of 2009), giving a completely different perspective on the effectiveness of the SUGP[67]. The purchase of the rights to respect the initial distribution between regularization and environment would imply a final reduction of the WF of 64 hm^3 (20% of the initial WF).

In fact, it would be possible to distinguish two independent measures that were applied together:

- buying rights to reduce directly the WF;
- regularising the non-authorized users on the condition they reduce their use of water.

In the first case, the water would be bought at a price of around 5 €/m^3 (10,000 €/ha and 2,000 m^3/ha). And a reduction of the WF of 40 hm^3 (Figure 7.12), would cost around 200 million euros. The second measure could have been realized without costs (apart from the monitoring costs) and could have allowed a reduction in the WF of 40 hm^3 (based on the initial non-authorized WF of 60 hm^3). Nevertheless, the acceptability of a plan based only on this measure is questionable, since it would consist in regularizing illegal use without reducing the total amount of rights.

[67] Relatively to this comment, the definition of the reference situation appears as essential. Here, it has been considered the effect of the SUGP on the estimated WF before its application (the WF in 2009), i.e. a situation where legal and illegal users coexist. If the reference situation is taken to be only the initial legal WF, the effect of the SUGP is to rise the legal WF more than it should be, since the percentage of water that should have gone directly for the environment is much lower (19% instead of 70%). However, it may be better to consider policy implications on real initial use.

Combining both effects, in June 2011, on the basis of a drop in the WF of 34 hm^3, the recovery of water for the environment was obtained for less than 2 €/m^3. Moreover, the final objective of the SUGP is not only quantifiable in terms of extraction reduction, but is also to build the basis for a new local governance of the groundwater bodies, thanks to the incorporation of the informal irrigators and a possible new cooperation between the stakeholders. The cost of the SUGP would be then relative because of possible steps in the social consensus achieved and the possibility of a new start.

However, in practice, a series of problems have emerged in the application of the plan, which could raise some doubts on the opportunity of spending this amount of money:

- There is no evidence so far that this restriction of the regularized vines (700 m^3/ha) has been respected. There are doubts on the real possibility for famers to irrigate vines with such a reduced amount of water. The question of effective control remains, even if meters have theoretically been installed.
- The legality of the purchased rights is questionable if they correspond to groundwater that was effectively used before the Water Law of 1985.
- It has been reported that the majority of the purchased rights correspond to areas that had not been irrigated during the five years prior to the purchase and some have been obtained outside the areas of priority for the purchase, or even from areas located in the water public domain (WWF, 2012)[68].
- In addition, some irrigation still took place in the areas that correspond to purchased rights (WWF, 2012). This issue is related again to monitoring withdrawals.
- Some farmers that were not irrigating vine before the plan may have obtained rights.

This adds to some more 'intrinsic' issues relative to the SUGP and its conception:

- Fairness of buying rights from users who were granted rights, on the basis they used groundwater prior to 1985.
- Fairness of granting rights to farmers who were using groundwater illegally.

[68] However, it does not mean that these rights were going to remain unused in the future. The conditions prevailing in the years before the application of the SUGP (high price of fertilizers, low market price of production and high cost of water because of a deep groundwater level) implied the irrigation of cereals was not very attractive. These conditions have changed in the last few years, e.g. with higher market prices for cereals and reduced pumping costs thanks to the recovered groundwater levels. Moreover, it reveals a major drawback for measures that aim at purchasing water from the existing rights (whether by public authorities or between particular through water markets): the buyers more willing to sell their rights are probably the ones who were not using them or who were the least productive. While this is the objective of a water market (transferring the rights to the more productive use), it can be problematic when the least productive users were not using their rights fully (e.g. because farmers need the full amount of water only for the driest years, especially for cereals). The new user will certainly use it fully, due to the high productivity. A transfer of rights should therefore be based on the estimation of real water use, not the total amount of withdrawal right granted.

- The final WF obtained after the application of the plan and the closing of the remaining illegal wells (240 hm^3, Figure 7.12) remains high relatively to the maintenance of the Tablas de Daimiel wetland (see next section).

The full implementation of the SUGP could have constituted an opportunity for a new start, but in the end may only have added a layer of complexity. It should be also kept in mind that the switch from cereals to vine means a change in the type of demand from one dependent on climatic conditions (spring rainfall), which vary each year, to a fixed one (vine areas are irrigated in summer and require a fixed amount of water), potentially exacerbating the pressure on the aquifer and the wetland.

VII.4.5 Withdrawal targets and cost-effectiveness of measures for wetland recovery

In the context of a wetland that is sustained thanks to a high groundwater table (see Section 1.3 of this chapter), a small drop in groundwater level can change drastically the flows received by the wetland. Under a climate with frequent drought periods of several years followed by more humid periods, the critical issue is the conservation of flows to the wetland during the driest years. The recharge is then much lower than its average value and withdrawals should be reduced even more. Even a substantial reduction of withdrawals (with loss of revenue and jobs for the area) may induce a limited improvement in the state of the wetland since there is no linear relation between pumping reduction and wetland conservation. Wetlands are representative of systems presenting thresholds and tipping points. Figure 7.14 illustrates that, despite significant reductions in withdrawals of 20 % or 40 % compared to the baseline scenario, the period with groundwater flows to the wetland hardly improves since, below a certain groundwater level, no flow goes to the wetland[69].

The level will probably fall anyway during the driest years, which will cancel the outflows to the wetland. Thus, the improvement of the state of the wetland may be limited. As presented earlier in the Table 7.2, IGME (2010) has established that the maximum pumping for a notable ecological recovery is 100 hm^3 (i.e. a WF of 90 hm^3). Below this threshold, flows to the wetland would be disrupted during the dry periods.

[69] A more detailed study could allow the definition of a varying acceptable amount of withdrawals, depending on the inflows to the aquifer during the preceding years. This should be based on a model of groundwater flows that goes beyond the scope of this thesis.

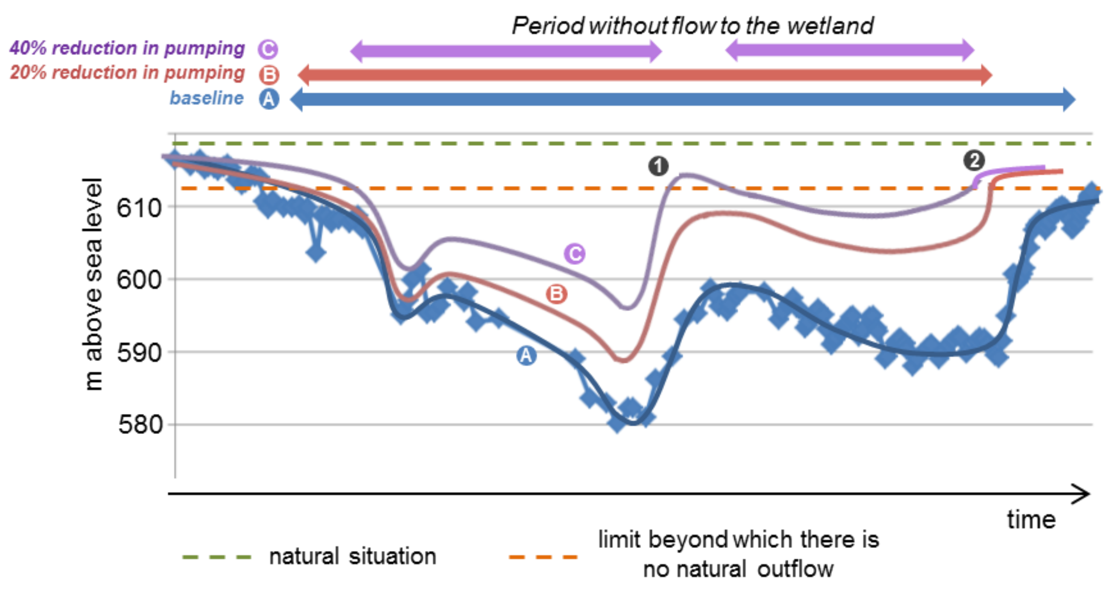

Figure 7.14. Implications of reduced pumping on levels and flows to the wetlands. Source: own elaboration based on the withdrawals data presented by De la Hera & Villarroya (2013)
The A (baseline) curve corresponds to the observed evolution of groundwater levels for the period 1979-2013. It is used here mainly to illustrate the influence of changing groundwater levels in a more realistic way than taking a random curve. B (20% pumping reduction) and C (40% pumping reduction) curves are obtained basically by rotation of the A curve. However, when the groundwater table attains the level where outflows to the wetland and evapotranspiration occur (points 1 and 2), these outflows must be added to the withdrawals and levels drop more rapidly. The lower pumping costs could also imply higher withdrawals when the level is high.

Following this approach, knowing the value of recharge during humid years and its average value has few implications on the strategy for wetland conservation. Even if the magnitude and frequency of extreme events influence the possibility of a high water table, the recovery of the aquifer level may be only temporary if the pressure (withdrawals) is not drastically reduced during drought periods (Figure 7.14). Moreover, a more rapid decrease in levels takes place with a high water table, because of the additional outflows to the wetlands and the potential rise in withdrawals due to lower pumping costs. Thus, more than the events that are able to replenish the aquifer, these are future drought periods that should be characterized (Table 7.8).

The stability of the groundwater level (out of the 'safe' pumping domain for the wetland) or the conservation of the aquifer stock, are not indicators that reflect wetland conservation (Table 7.8). Speaking in terms of depleted stock or water level recovery is a discourse associated with the objective of conserving the stock of groundwater, which implicitly disregards the issue of wetland conservation, i.e. a view in terms of 'stock' that overlooks 'flow impacts' as explained in Chapter 3 on groundwater resource allocation.

Table 7.8. Characterization of two potential objectives of management.

	Objective	Indicator	Time horizon	Main concern	Evidence on Figure 7.14
A	Conserving the stock of the aquifer for a continuous use.	Level, groundwater stock.	A few years or decades.	Return of a high rainfall period in order to replenish the aquifer.	The level is recovered in all the scenarios.
B	Conserving sufficient flow to the wetland for its good functioning.	A level above the threshold for enough flows to the wetland.	Duration of the drought.	Conserving enough outflows to the wetlands during drought periods.	The wetland receives more or less water depending on withdrawals.

It should be observed that the objective introduced by the Water Plan – 220 hm^3 (CH Guadiana, 2013) – is probably too high to allow for the conservation of Tablas de Daimiel wetland. This value may contribute only to extend for some months the period where groundwater outflows take place during humid periods. In fact, this may be a compromise that includes the necessity to maintain the economic activity linked to groundwater use[70].

In relation with the previous considerations, the question of the real possibility of recovering the Tablas de Daimiel wetland should be raised. Indeed, the objectives set by the Water Plan or the SUGP, or the consequences in terms of economic, social and political cost of a high withdrawals reduction, show that the probability of ecosystem recovery appears very limited (Figure 7.15). This was already discussed by Llamas (1988):

> "(…) [the Tablas de Daimiel] regeneration is virtually impossible because of the economic and social implications involved."

If the measures that aim at recovering the wetland or 'obtaining water for the environment' (e.g. the SUGP) will not mean a tangible recovery of the wetland, a first consequence is that it is not necessary to have costly measures (from the economic or political point of view), if there is a high uncertainty on their success.

The focus of the public authorities should therefore be questioned and objectives should change to a management where the ecological impacts are not the central objective but that focus instead on the aquifer 'as a stock' ('scenario A' in the Table 7.8). It is the 'classical' situation considered by the economic-modeling of groundwater literature (see Chapter 3 on the definition of groundwater availability), which could be informative in this case. It should be also kept in

[70] The approach of the EU WFD, however, would have required defining the maximum pumping that does not jeopardize the wetland, and only in second place, justifying the exemption of attaining this level of pumping because of the high economic cost.

mind that the proper system has some margin for ‘self-regulation’ due to the potential reduction in pumping, when levels drop and pumping costs rise, as explained in particular for cereals earlier in this case study.

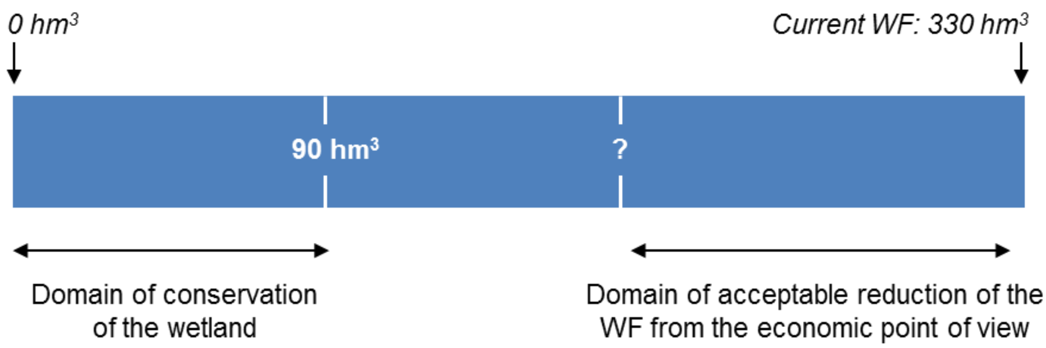

Figure 7.15. Acceptable withdrawals reduction and policy domains.

VII.5 Synthesis

- The WF of the Western Mancha Aquifer reached 330 hm^3 in 2009, with vines constituting 60 % of the WF, followed by cereals and vegetables (20 % and 15 % respectively). However, cereals have varying water requirements according to annual climatic conditions. The change in WF composition in the period 2000-2009 is mainly due to vine irrigation development.
- The WF triples the limit for conserving the Tablas de Daimiel wetland on the basis of a continuous groundwater flow to the wetland. With higher values, flows could be maintained only during humid periods. Thus, it is questionable if the limit set by the Guadiana Water Plan (CH Guadiana, 2013) at 200 hm^3 will maintain groundwater outflows out of exceptionally humid periods.
- Crops with higher direct economic value and direct employment generation show higher illegal activity.
- Illegal use has generated more than half of income and employment linked to irrigated land in 2008 (215 million euros and 2,500 jobs). This shows the economic, social and, thus, political implications of acting directly on illegal use. It is all the more difficult since the initial allocation of water rights is perceived as unfair by vine irrigators who were not entitled rights because vines irrigation was prohibited until 1995.
- The Special Upper Guadiana Plan, in force in the period 2007-2013, has been an attempt to change rights attribution. Rights were bought definitively from farmers irrigating cereals and allocated to the environment and non-authorized vines irrigators.

In fact, the measure that has allowed for the most important decrease in WF is the reduced entitlement of rights for regularized vines irrigators. The cost of buying rights was, however, too high which meant an end to the plan. It has allowed reintegrating part of illegal users in the regulation sphere, which may contribute to a better acceptance of rules in the future.

- Productivity indicators should be interpreted carefully as reallocation should be based on marginal productivity. Thus, the vision of vines as a productive crop that has motivated its development is questionable, all the more so since subsidies have maintained prices artificially in a context of over-production. Reallocation of rights based on productivity may also contribute to consolidate the WF, which was variable annually (when rights were used for cereals), toward a fixed consumption (for vines).
- It is questionable if a groundwater based economy in the current order of magnitude and the conservation of the Tablas de Daimiel wetland are compatible. It should be recognized that one of the objectives must be prioritized, which would have implications for public policies design. However, this is a politically difficult decision.

Chapter VIII

GUADALQUIVIR RIVER BASIN

VIII. GUADALQUIVIR RIVER BASIN

Objective: The Guadalquivir River Basin case study is the only one where the scale is at the river basin level. It aims at applying the WF methodology to show the situation on water resources use and its evolution over the period 1997-2008, thanks to: first, a detailed accounting, and, second, an assessment, where the role of groundwater and its dynamics is integrated, particularly applying the concepts of 'current' and 'future capture'. The case of the La Loma de Úbeda Aquifer (Jaén Province) serves as an illustration of the dynamics of the development of olive groves irrigation, which has supposed an intensification in the use of groundwater in the headwater over the last twenty years. Even though the focus is on the particular role of groundwater, the general approach is based on the evaluation of the whole WF of the Guadalquivir River Basin, integrating green water and blue water[71].

VIII.1 Introduction

The Guadalquivir River basin is located in the south of Spain (Figure 8.1). It is a semiarid region where rainfall amounts to 535 mm/year and which covers an area of 57,530 km^2, 90% of which is included in the Autonomous Community of Andalusia[72]. The population of approximately 4.1 million is concentrated in the lower stretch of the basin. The Seville Province has around 2 million inhabitants, with 1.5 million in the Seville urban area. The secondary and tertiary sectors are also mainly located in this region.

Agriculture constitutes an economic activity that extends over much of the territory. The combination of climatic conditions and topography in the upper and middle parts of the basin makes these areas particularly suitable for olive groves. The lower part of the basin is characterized by higher crop diversification (rice, cotton, olive, cereals, sunflower and fruits). In the whole basin, of the total cultivated area of about 2.6 million ha, nearly 1.5 million ha was dedicated to olive growing in 2008. The irrigated olive groves, with 470,000 ha, constitute approximately 60% of the total irrigated area. A main driver for this situation has been the EU Common Agricultural Policy (CAP) subsidies that have incentivized the expansion of irrigated olives in the 1990s. The major droughts experienced in this period have also meant that

[71] Some parts of this chapter are a revised and more detailed version of the paper 'The water footprint of a river basin with a special focus on groundwater: The case of Guadalquivir River basin (Spain)' (Dumont et al., 2013a), which is itself based on Salmoral et al. (2011b) and Dumont et al. (2012). A co-author (G. Salmoral) analysed the green and surface water WF accounting, based a jointly developed methodology.

[72] The other Autonomous Communities covered by the Guadalquivir River Basin are Extremadura, Castilla-La-Mancha and Murcia. It is therefore a 'shared' river basin and its management depends on the Spanish central State. However, because more than 90% is included in its territory and all the other Autonomous Communities are located upstream, in 2007 Andalusia claimed to have the full control on the part covering its territory, a decision canceled by a ruling from the *Tribunal Constitucional* in 2011.

irrigating was a kind of insurance against the reduction of revenues. The last section of this chapter, dedicated to the La Loma de Úbeda Aquifer, deals more extensively with the issue of olive irrigation.

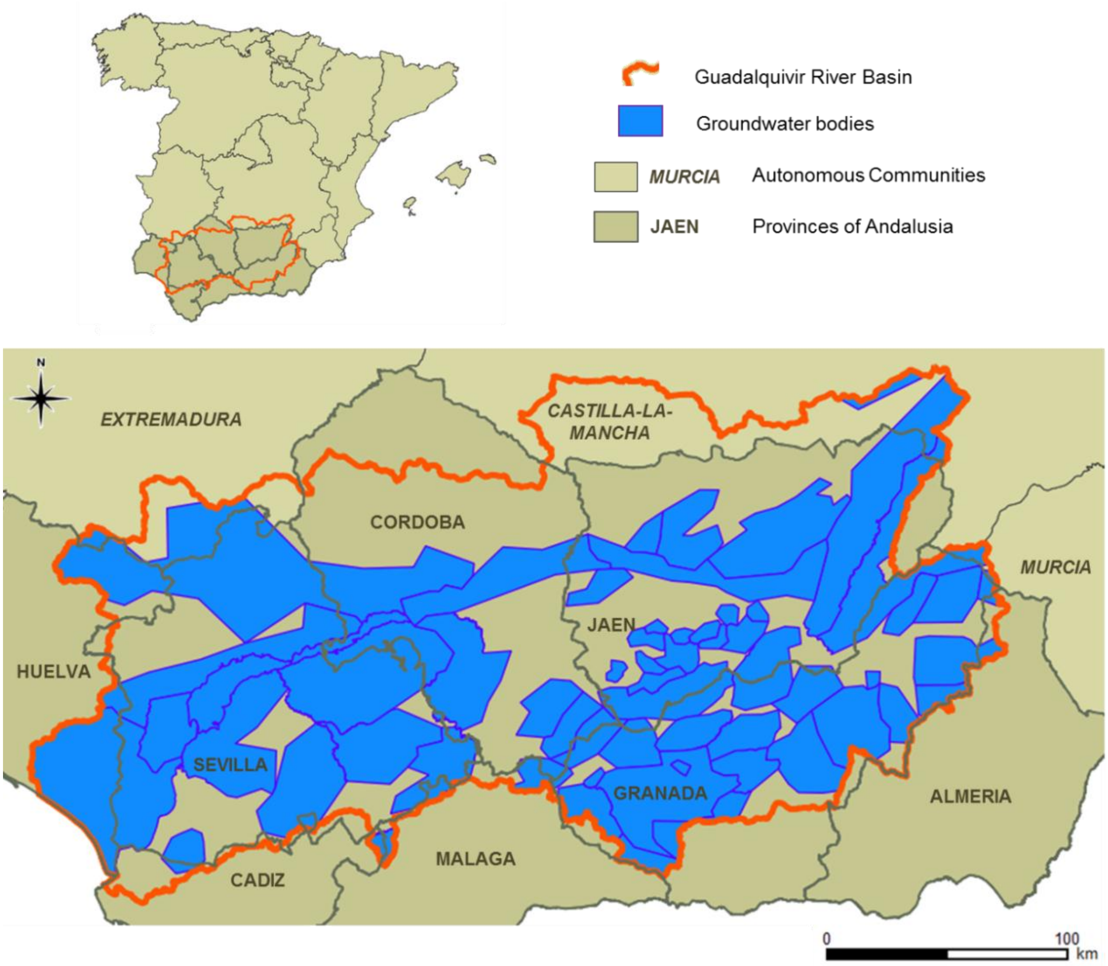

Figure 8.1. Localization of the Guadalquivir River Basin and groundwater bodies.

The hydrological network is organized around the 655 km long axis of the Guadalquivir River. The organization in charge of water resource management and planning in the basin is the Guadalquivir River Basin Authority, which depends on the Spanish central government.

The Guadalquivir River is deeply affected by human activity, especially due to the necessity to deliver water to farmers during the summer months for irrigation. It is a highly regulated watershed and during the low flow season (May-August), river flows principally originate from dams discharges for irrigation. Characteristic of semi-arid areas, rainfall presents a high intra- and inter-annual variability (Table 8.1 and Figure 8.2). These variations are mitigated by the numerous reservoirs. During the driest years, however, the agricultural demand cannot be entirely satisfied.

Table 8.1. Rainfall and water flows generation for the periods 1940/41-2005/06 and 1980/81-2005/06. Source: CH Guadalquivir (2010a).

	Period 1940/41-2005/06	Period 1980/81-2005/06
Rainfall (average) (mm)	573	536
Rainfall (max.) (mm)	983	849
Rainfall (min.) (mm)	260	280
Water flows (average) (hm^3)	7,043	5,754
Water flows (max.) (hm^3)	21,530	15,180
Water flows (min.) (hm^3)	372	372

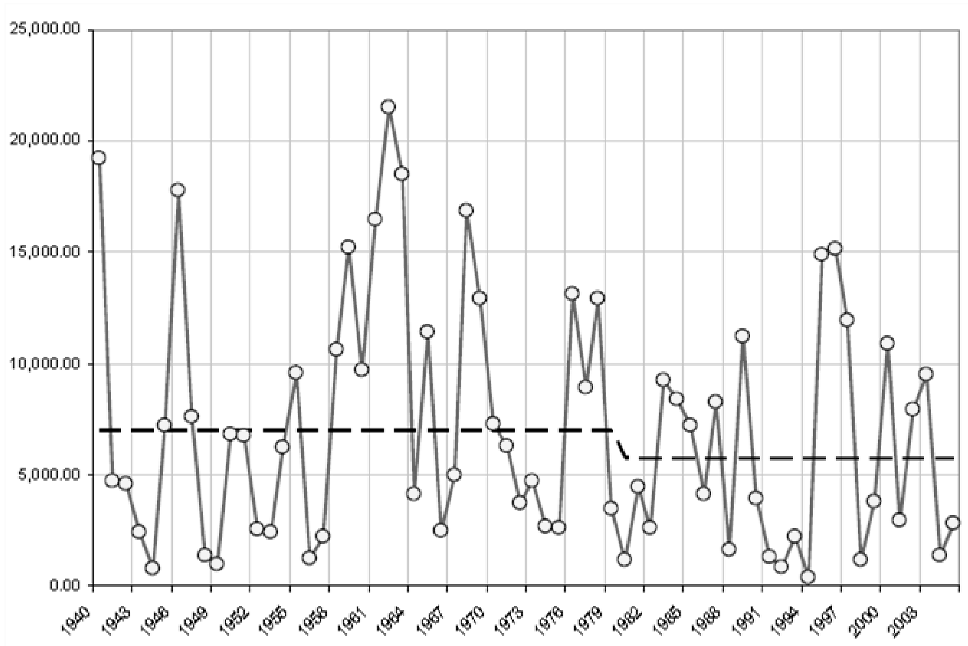

Figure 8.2. Water flows generation on the period 1940-2006. Source: CH Guadalquivir (2010a)
The dotted line shows the average flows for the period 1940-1980 and for the period 1981-2006.

In the Guadalquivir River basin, as an average, 77% of the agricultural demand is met (Junta de Andalucía, 2010b) and water delivered to farmers can be reduced during periods of drought. This restriction is integrated in the WF accounting (see Chapter 6 on methodology and data).

The assessment of the water bodies status in compliance with the EU Water Framework Directive (WFD) resulted in 164 out of 392 surface water bodies and 28 groundwater bodies out of 60 to be listed as in ‘poor status’ (CH Guadalquivir, 2010a) (Figure 8.3, a and b). This result is the combination of 19 groundwater bodies in poor quantitative status and 16 in poor qualitative status (Figure 8.3, c and d). In the definitive version of the Water Plan (CH Guadalquivir, 2013), there are only 27 groundwater bodies presented in ‘poor status’. In order to reflect more adequately the situation regarding groundwater use, however, it should be noticed that groundwater bodies in poor quantitative status (32% of total) correspond with 75% of groundwater used in the basin (620 hm^3) (Dumont et al., 2011a).

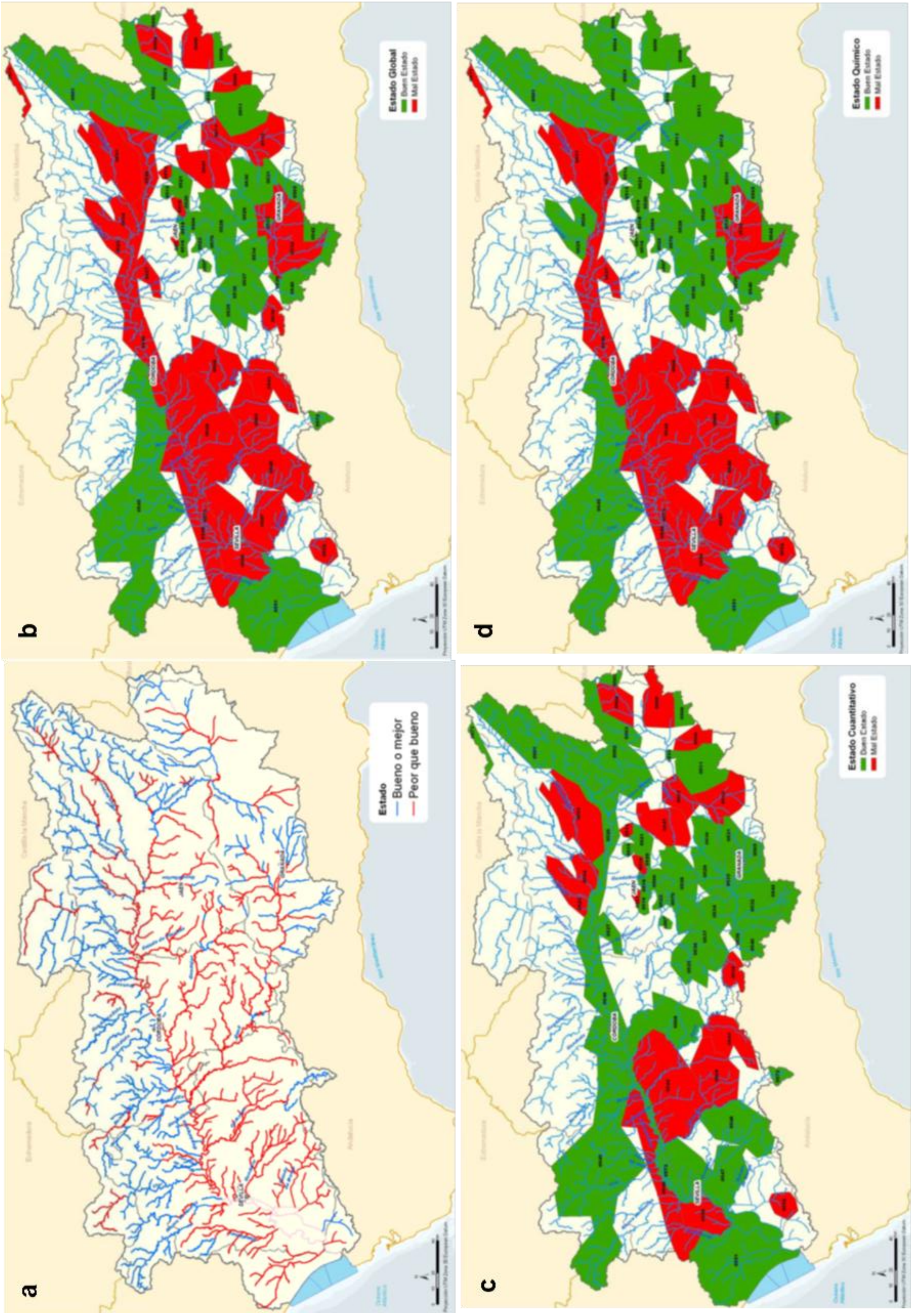

Figure 8.3. (a) Surface water bodies status; (b) Groundwater bodies general status; (c) Groundwater bodies quantitative status; (d) Groundwater bodies qualitative status. Source: CH Guadalquivir (2010a)

One of the main innovations in this chapter is to disaggregate the blue WF for surface water and groundwater for the majority of the economic sectors. The WF from groundwater distinguishes a component that results in the current reduction of surface water flows and a component implying a delayed impact on surface resources, through the aquifer stock depletion, i.e. current and future capture as described in Chapter 2 on groundwater flow dynamics. Another advance is a water balance of the WF for the different economic sectors and main land uses, including green water used by forests and pastures. The WF associated to dams is also quantified.

VIII.2 Specific methodological considerations

The general methodology and data sources for the WF accounting have been described in Chapter 6. However, it appears necessary to present some additional insights on specific methodological points for this case study regarding the calculation of the WF from groundwater, the distinction between current and future capture, and the integration of the different WFs in a water balance at basin scale.

VIII.2.1 Accounting of water footprint from groundwater

As the origin of water is not specified in the data set used for crop surfaces (MARM, 2011), detailed groundwater area by crop group is obtained only for the years 1997 and 2002, based on Junta de Andalucía (1999, 2003). For 2008, data on agricultural groundwater withdrawals is available for each groundwater body (CH Guadalquivir, 2010a), without specifying the irrigated crop category. The WF from groundwater for 2008 is obtained multiplying the withdrawals by an application efficiency of 0.80. To estimate the WF from groundwater for the whole period 1997–2008, a linear evolution between 1997 and 2002 and 2002 and 2008 is assumed.

The agricultural WF is also assessed separately for the upper (Ciudad Real, Albacete, Jaen and Granada Provinces), middle (Cordoba and Malaga) and lower (Cadiz, Huelva and Seville) sections of the Guadalquivir River basin (Figure 8.1). As the data set for the WF from groundwater in 2008 does not distinguish the irrigated crop category, it is assumed that the rise in value, as compared to the situation of 2002, corresponds only to olives for the upper and middle sections of the basin (based on Salmoral et al., 2011b)[73].

VIII.2.2 Distinction between current and future capture

The capture resulting from groundwater pumping (reduction in discharge and/or a rise in recharge) is mobilized through groundwater table drawdown that varies depending on the pumping intensity. Once pumping stops, the replenishment of the aquifer will continue to

[73] Between 2002 and 2008 the main crop change refers to olives, thus this is probably a valid assumption.

impact surface water bodies until the natural level of groundwater is attained again. In other words, there is a delayed impact of pumping on surface water resources ('future capture' as discussed in Chapter 2). However, the dynamic equilibrium is not attained in some situations and the groundwater table drops continuously. This is the case when:

- the pumping rate is too high to be compensated by the maximum capture;
- a new dynamic equilibrium takes a long time to be established.

Future capture should be introduced if surface water capture can be mobilized to replenish the aquifer, which is the case for the Guadalquivir river basin. Thus, in the consideration of the WF from groundwater, two fractions are distinguished:

- a fraction that currently consumes surface water resources;
- a fraction that will impact availability in the long run through future capture.

To obtain an estimation of withdrawals from the aquifer stock from the data presented for each of the sixty groundwater bodies in the draft Water Plan, two cases are considered:

- when withdrawals are reported to be higher than total inflows (recharge), the consumption of the stock is assumed to be the difference between the two values;
- when inflows are higher than withdrawals and a continuous decline of level is reported in CH Guadalquivir (2010a), which indicates a transient state (capture has not been fully mobilized and some of the withdrawals generates a depletion of the reserve), stock consumption is assumed to be half of the total withdrawals. The other half is linked to a current consumption of surface flows.

The WF from groundwater stock has been assumed to be associated entirely with agriculture.

VIII.2.3 Balance of green and blue water consumption at basin scale

An overview of the relative weight of the green and blue WFs and ecosystems water consumption at basin scale is obtained through the integration of these values within a water balance of the basin (Table 8.2). The average repartition of rainfall between total run-off generation and evapotranspiration (i.e. green water) is obtained from CH Guadalquivir (2010a) and the values of precipitation and WF for the reference year (2008) are used without water quota restrictions, on the following basis:

- Total run-off is the sum of the blue WF (without the WF from groundwater stock) and the remaining water flow running along the streams and recharging groundwater bodies.
- Evapotranspiration equals the sum of the green WF agriculture (cropping and non-cropping seasons), pastures and forest ecosystems. This allows estimating forest share.

Table 8.2. Repartition of rainfall between run-off and evapotranspiration and green and blue water consumption. Source: based on CH Guadalquivir (2010a).

Rainfall	Run-off	Evapotranspiration
507 mm $year^{-1}$ [a]	96 mm $year^{-1}$	411 mm $year^{-1}$
100%	19%[b]	81% [b]
28,850 Mm^3	5,480 Mm^3	23,370 Mm^3
	Blue WF (without the WF from aquifer storage) + Blue water flows	Green water (agriculture, pasture, forests)

[a] For the year 2000 – [b] According to CH Guadalquivir (2010a)

Based on the hydrologic model BalanceMED applied to an area of the "Sierra Norte de Sevilla" (Willaarts et al., 2012), it is assumed that green water consumption during the non-cropping season amounts to 40 % of green water consumed by agriculture for the whole year. Even if this model was applied to a small area of the Guadalquivir River basin, this value can be introduced as a first approximation. This factor is not included in the general approach of the green WF of agriculture, in line with standards on green WF calculation that consider only the crops growing period (Hoekstra et al., 2011). This point is discussed further in the Section 4.1 of this chapter.

VIII.3 Results

The evolution of the WF for the period 1997-2008 is only presented for agriculture. For the remaining results (other sectors and detailed results in agriculture), the reference year is 2008. This year presents an average rainfall, which allows presenting 'normal' conditions. However, because of the previous drier years, many management districts endured restrictions in water allowances (factor 'drought', see Sectio 1.2 of Chapter 6) in 2008. Thus, when the results for this year are presented to obtain a view of the situation independently from irrigation water restrictions, the factor 'drought' is not considered. Moreover, the majority of data from the draft Water Plan (CH Guadalquivir, 2010a) have been obtained around this year.

VIII.3.1 Synthesis of the green and blue water footprints

The WF of the different economic sectors within the Guadalquivir River basin for the year 2008 is summarized on Figure 8.4. The green WF amounts to 7,280 hm^3, while the blue WF value is 2,790 hm^3. Overall, agriculture represents the largest WF, with 95 % of the total WF or 75 % if considering only blue water. Evaporation from dams is also important, since it comprises 11% of the blue WF, which surpasses by more than 100 hm^3 the sum of the blue WF for all the sectors except agriculture. Sectors such as tourism and golf comprise a much lower share of the blue component (< 1%).

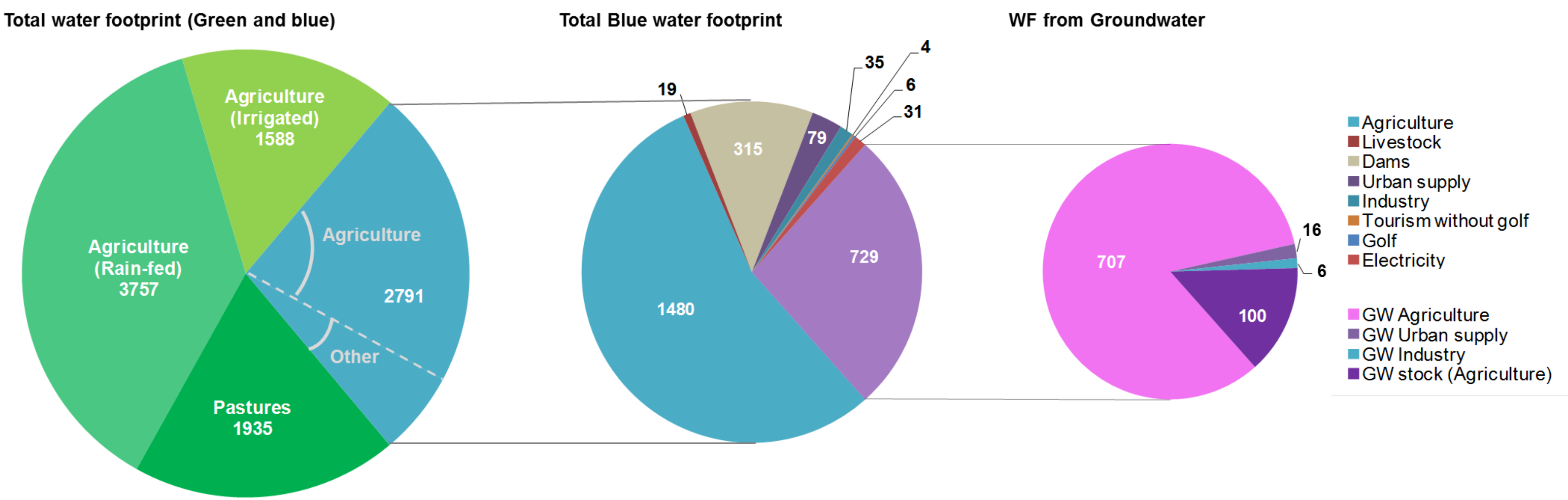

Figure 8.4. The total water footprint (hm^3) of the Guadalquivir River basin, distinguishing green and blue (surface and ground water) water footprints.
Source: Dumont et al. (2013a)

As regards the origin of blue water, around 25% (730 hm^3) of the blue WF is from groundwater, almost entirely associated to agriculture: 707 hm^3 as compared to 16 hm^3 for urban supply and 6 hm^3 for industry (based on CH Guadalquivir, 2010a) (Table 8.3). Groundwater constitutes around one third of the total blue WF of agriculture but represents only around 20% of the total WF of industry and urban sectors.

Withdrawals from the aquifer stocks, i.e. potential future capture of surface flows, amount to 100 hm^3 (14 % of WF from groundwater for agriculture), with 73 hm^3 referring to withdrawals in excess of recharge and 27 hm^3 pumped from aquifers presenting a lowering head.

Table 8.3. Groundwater withdrawals and water footprint from groundwater by sector. Source: CH Guadalquivir (2010a, 2013)

	Withdrawals				Water footprint			
	Agriculture	Urban	Industry	Total	Agriculture	Urban	Industry	Total
Inside GWB 2010	756.8	54.7	11.0	822.6	643.3	15.3	6.2	664.8
Outside GWB 2010[a]	74.5	1.3	0	75.9	63.3	0.4	0.0	63.7
Total 2010	831.3	56.0	11.0	898.5	706.6	15.7	6.2	728.5
Inside GWB 2013	774.3	54.7	11.0	840.0	658.1	15.3	6.2	679.6
Outside GWB 2013	79.4	1.3	0.7	81.5	67.5	0.4	0.4	68.3
Total 2013	853.6	56.0	11.8	921.5	725.6	15.7	6.6	747.9

GWB: groundwater body / [a] 'Outside GWB' refers to the groundwater withdrawals that take place in places where groundwater bodies have not been defined.

VIII.3.2 Water footprint by groundwater body

In this section, the WF is considered 'from the perspective of the groundwater bodies' (i.e. the WF is the same as the withdrawals). This study was undertaken when the provisional version of the Water Plan (CH Guadalquivir, 2010a) was available. Only when explicitly indicated, the final version (CH Guadalquivir, 2013) is used.

An overview of the total WF (agricultural, urban and industrial) for each groundwater body is presented on Figure 8.5 (based on CH Guadalquivir, 2010a). The sum of the five groundwater bodies with the highest WF ('Almonte-Marismas', 'Sevilla-Carmona', 'Úbeda', 'Altiplanos de Ecija' and 'Depresión de Granada') represents 42% of the total WF of 900 hm^3 (updated at 920 hm^3 in CH Guadalquivir, 2013).

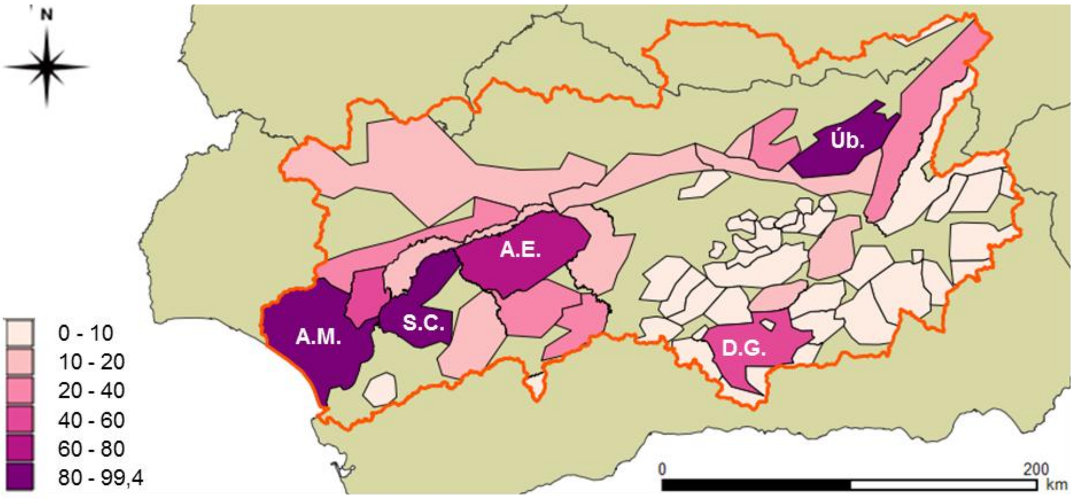

Figure 8.5. Groundwater withdrawals by groundwater body (hm^3). Source: own elaboration based on CH Guadalquivir (2010a).
A.M.: 'Almonte-Marismas' (99.4 hm^3); S.C.: 'Sevilla-Carmona' (83 hm^3); Úb.: 'Úbeda' (80.3 hm^3); A.E.: 'Altiplanos de Ecija' (63.9 hm^3); D.G.: 'Depresión de Granada' (51,3 hm^3).

Figure 8.6 presents the WF for urban supply. All the groundwater bodies that present a poor quantitative status have more than 80% of their withdrawals associated with agriculture. Exceptions to this rule are 'Jaén', 'Mancha Real-Pegalajar' and 'Bedmar-Jëdar', where respectively 99%, 96% and 64% are dedicated to urban supply (Figure 8.6). However, these are groundwater bodies of only local importance for urban supply, with less than 2 hm^3 of withdrawals each. The case of the 'Depresión de Granada' groundwater body should be noticed as well, since, even if agriculture presents 76% of withdrawals, it is the groundwater body with the highest withdrawals for urban supply (10.3 hm^3) and industry (2.4 hm^3) (Figure 8.6 and 8.7).

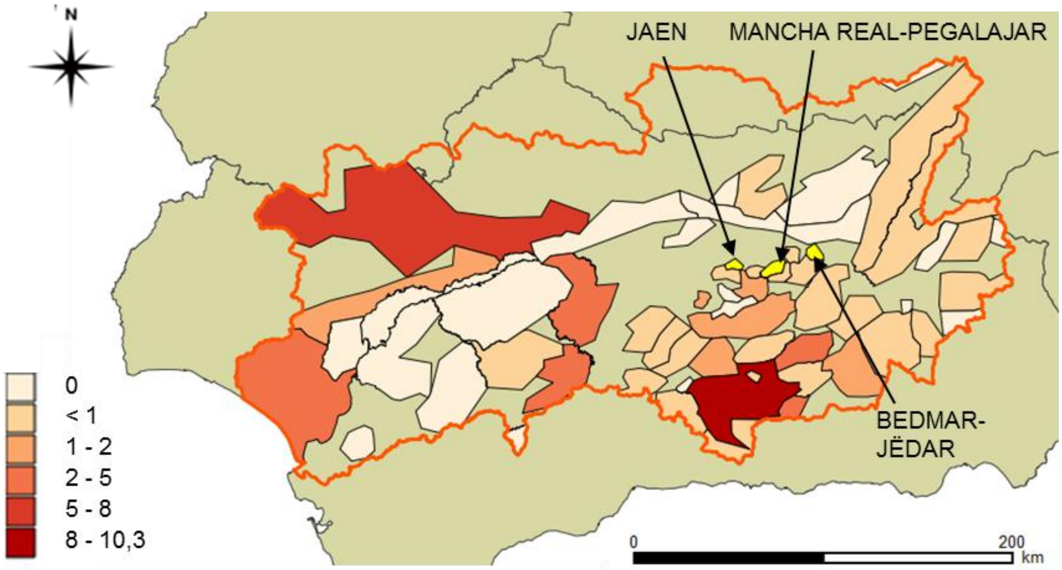

Figure 8.6. Groundwater withdrawals by groundwater body for urban use (hm^3). Source: own elaboration based on CH Guadalquivir (2010a).

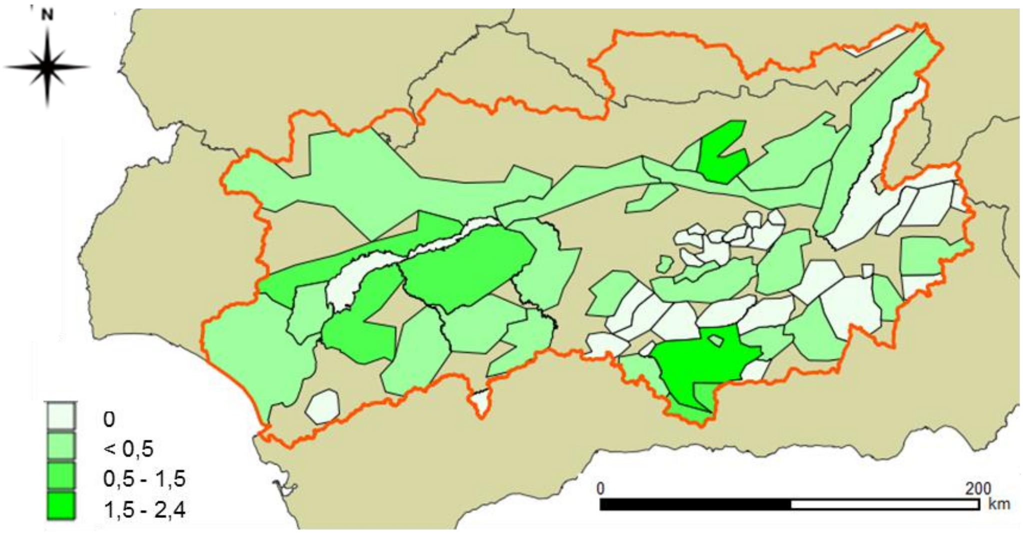

Figure 8.7. Groundwater withdrawals by Groundwater body for industrial use (hm^3). Source: own elaboration based on CH Guadalquivir (2010a).

Even if this overview is principally quantitative, since the focus is on the blue WF, it should be noticed that the groundwater bodies with the largest footprint have in general a poor qualitative status. On the 17 groundwater bodies with the highest WF, 13 are in poor qualitative status[74].

VIII.3.3 Water footprint of agriculture and its evolution over time

Between 1997 and 2008 the total WF (green and blue) of agriculture production ranged between 4,050 hm^3 (year 1999) and 7,230 hm^3 (year 2001). These variations are mainly due to the irregular pattern of rainfall within the basin, which has a high influence on the green WF and the need to use blue water (Figure 8.8). During the period, 69% of mean annual agricultural WF in the Guadalquivir River basin was green and the remaining 31% was blue, including both surface and groundwater. Overall, olive groves consumed the largest proportion of green and blue water with 74% and 31% of the total WF respectively.

Two scenarios are considered for the blue WF: if we include or not the restrictions imposed through the factor 'drought'. It allows distinguishing temporary situations (droughts) from the general evolution of the potential WF governed principally by the evolution of the irrigated area. In reality the WF is situated between these two values, since doubts can be raised on the actual implementation and compliance of normative restrictions.

[74] 16 groundwater bodies are in poor qualitative status (57% of the total WF). However, the groundwater body with the highest withdrawals ('Almonte-Marismas') has been classified as in good quantitative status, based on an extraction index slightly lower than the threshold of 0.8 (0.7969 in CH Guadalquivir, 2013) and is the only groundwater body classified in good qualitative status despite presenting some 'no compliances' in terms of nitrates, considered 'no significative' (CH Guadalquivir, 2013).

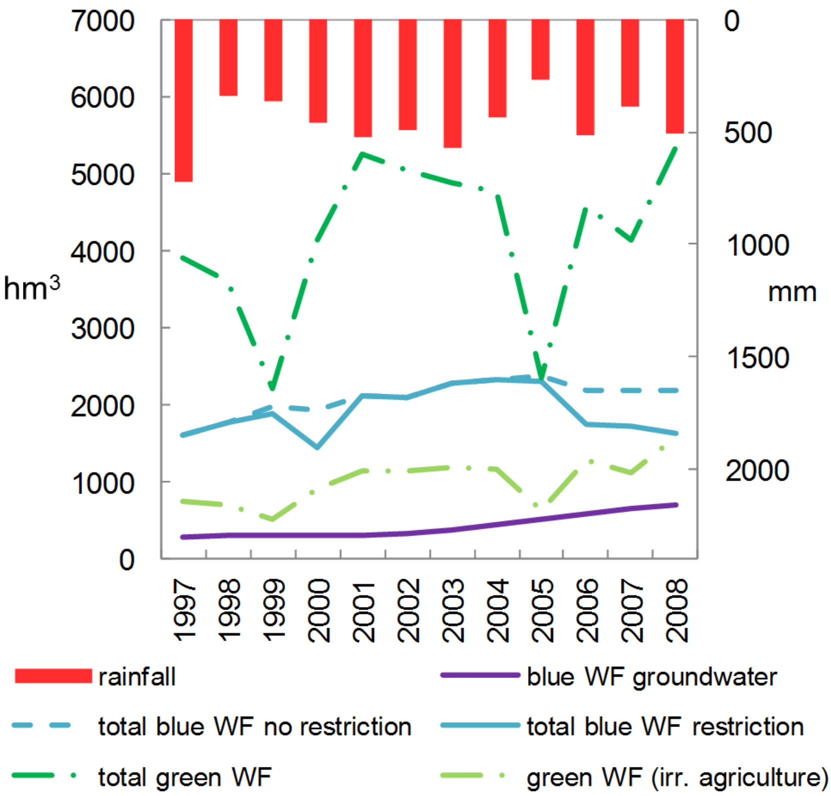

Figure 8.8. Rainfall and agricultural water footprint over the period 1997-2008. Source: Dumont et al. (2013a)

There has been a general rise in the WF until the year 2005. After, for the period 2006-2008, the WF decreases for both scenarios. Indeed, even without considering the normative irrigation restrictions, the low rainfall in 2005 meant a decline in the area irrigated from surface water, potentially because farmers preferred to secure the amount of water delivered to a smaller area. Meanwhile, the WF from groundwater rose from 290 hm^3 in the year 1997 to 700 hm^3 in the year 2008. This increase is mainly ascribed to the expansion of irrigated olive orchards, particularly over the last years (120 hm^3 in 2002 to 490 hm^3 in 2008).

A more detailed view by river basin section shows that in the upper part of the river basin the agriculture WF presents an average value of 2,430 hm^3, comprising 26% of blue water (Figure 8.9). This section is dominated by olive groves, both under rain-fed and irrigated conditions, the olives WF reaching about 1,570 hm^3 and 415 hm^3 of green and blue water, respectively. In the middle section, the average WF is 1,560 hm^3, 17% being blue water. There, the olives WF is around 1,010 hm^3. The lower part of the basin presents an average WF of approximately 1,960 hm^3, comprising green and blue water in similar magnitudes with 55% and 45%, respectively. The crops with a higher weight in the blue WF in this area are cotton (230 hm^3), rice (200 hm^3) and maize (100 hm^3).

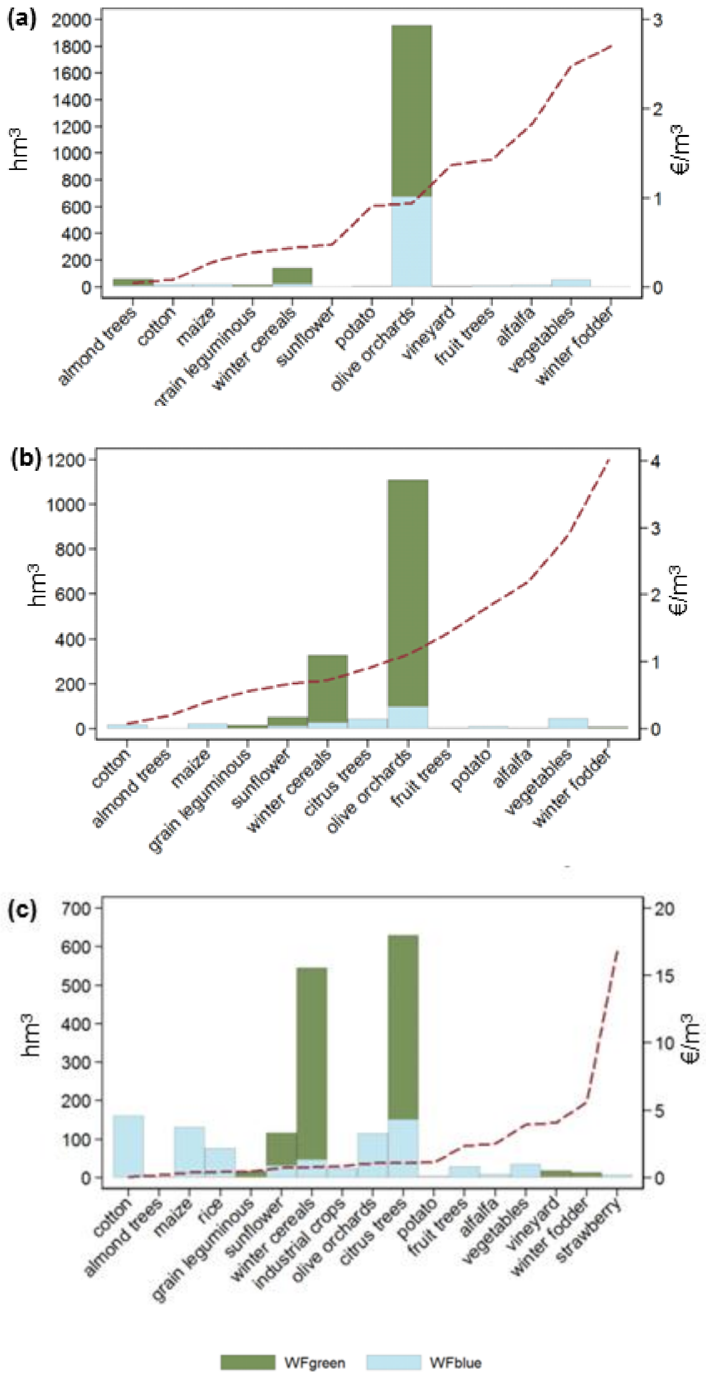

Figure 8.9. Water footprint and direct economic value by crop category and basin section for a normal climatic year (2008) and without water allowance restrictions. Source: Dumont et al. (2013a)

(a) upper section of the Guadalquivir River basin; (b) middle section; (c) lower section.

Regarding the origin of blue water (Figure 8.10), the upper section presents an increase in the WF from groundwater of more than 170% between 2002 and 2008, with 210 hm^3 linked to olives and 25 hm^3 for remaining crops in 2008. The consumption of the groundwater stock occurs mainly in this part of the basin (60%). In the middle section, the share of groundwater is more limited. The highest WF associated to groundwater is in the lower section, with a value of 410 hm^3 (60% of total WF from groundwater). The same as for surface water, there is more diversification of the crops irrigated from groundwater in the lower part, with only 57 % of the WF linked to olives. However, for all the sections of the basin, and both for surface and groundwater, the share of olives WF has increased over the study period of 1997 to 2008.

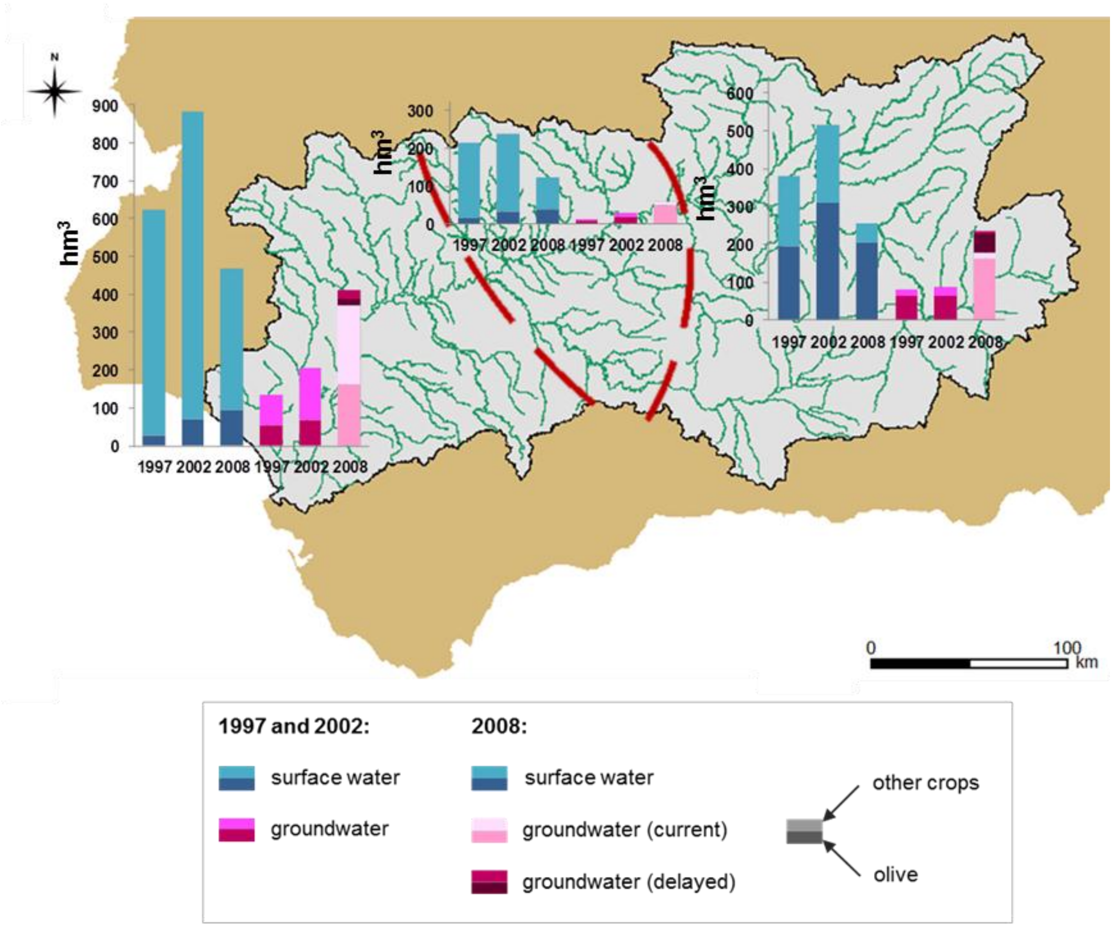

Figure 8.10. Blue water footprint by section of the basin and by origin of water distinguishing olives water footprint. Source: adapted from Dumont et al. (2013a).

For the years 1997 and 2002, it was not possible to estimate the share of groundwater from stock consumption (future capture).

Even if part of the groundwater withdrawals comes from aquifers stock, mainly in the upper stretch of the basin, the major part of the increase in the WF from groundwater implies a higher pressure on surface water and associated ecosystems through capture. While the increase in the upper section is clearly linked to the extension of irrigated olive groves, the more intensive use of groundwater in the lower section can be partly explained by a reduced availability of surface water flows in the last years of the study. Since the conjunctive use of surface and ground water is common in this area (Junta de Andalucía, 2003), these figures might be linked to a greater use of groundwater to compensate for surface flow reductions. However, the drop in WF from surface water as presented on Figure 8.10 may be exaggerated since the restrictions for the year 2008 might not have been fully effective.

VIII.3.4 Economic value of blue water consumed by agriculture

Between 1997 and 2008, 40% of the blue WF is for crops with a Direct Economic Value (DEV) of less than 0.40 €/m^3, mainly cotton, rice and maize. Crops generating more than 1.50 €/m^3 only account for 10% of total blue WF. These are principally open air vegetables, vineyards, winter fodder and strawberry. For the reference year (2008), in the upper part of the basin, olives present a DEV of 0.9 €/m^3, while for vegetables and winter fodder, it reaches a value of 2.5 and 2.7 €/m^3 respectively, although their blue WF is minimal (Figure 8.9). In the middle part, DEV of olives is 1.1 €/m^3 and vegetables (3.5 €/m^3) and winter fodder (4 €/m^3) again present the highest values. Cotton has a DEV of only 0.1 €/m^3. In the lower section, although cotton, maize and rice are the largest blue water consumers, their DEV is less than 0.4 €/m^3. Strawberries reach the highest direct value of blue water (16.9 €/m^3), followed by winter fodder (5.6 €/m^3) and vineyards (4.1 €/m^3). This confirms that the largest proportion of blue water resources is allocated to produce low value crops.

VII.3.5 Integration of water footprint within the water cycle

Throughout the water cycle, more than 80% of the rainfall turns into green water and only 20% is available for rivers and aquifers as blue water (Figure 8.11).

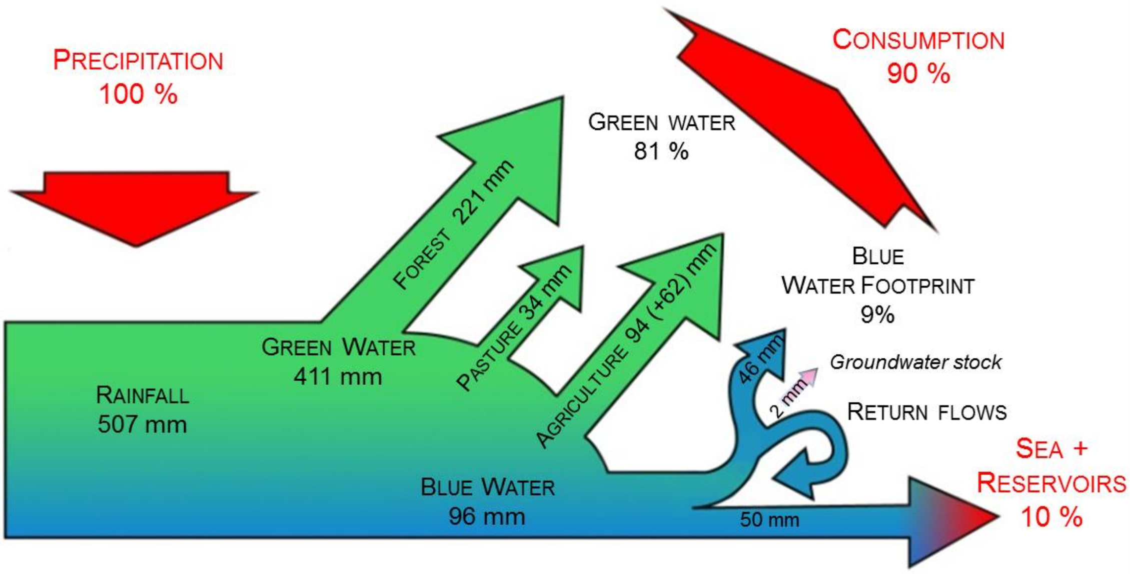

Figure 8.11. The water footprint within the water cycle, for a normal climatic year (2008) without irrigation allowances restriction. Source: Dumont et al. (2013a).
The first value of the water footprint of agriculture corresponds to the consumption during the growing period and the second value to the rest of the year.

The majority of green water is consumed by forests (54%), while the direct human appropriation of green water (WF of agriculture and pastures) represents 46%. Regarding blue water, as an average, 50% of the total run-off is consumed annually (blue WF) and the other half partly discharges into the ocean, after contributing to sustain the ecological functions of aquatic ecosystems on its way to the river mouth. A fraction is also kept in reservoirs to meet future demand. This general representation corresponds to a year with average climatic conditions. Depending on annual conditions, both the total amount of rainfall and the repartition between evapotranspiration and flow generation can vary significantly.

VIII.4 Discussion

VIII.4.1 The water footprint as human appropriation of water resources: distinguishing environmental and human water consumption

Off-stream blue water uses can generally be directly ascribed to the related human use, since water is withdrawn from streams and aquifers to a specific use. However, the assignation of green water is more complex since, in parallel to human activities, land uses associated to green water consumption also sustain ecosystems. The distinction between (human) WF and ecosystems consumption we proposed relies on the fact that, for instance, agricultural systems diverge from the initial ‘natural’ land use and the evapotranspiration can be associated entirely to the WF of crops. Nevertheless agricultural ecosystems are also valuable beyond the perspective of crop production as they contribute to the landscape and biodiversity, for example.

Although green water consumption during non-growth period could also be related to the agricultural land use, our main approach only considers the green WF for the growing period (in line with the traditional methodology of the WF; Hoekstra et al., 2011),. On the contrary, forests are considered as ‘natural’ land use, even if direct economic activities are associated, like lumber or papermaking industries (Van Oel & Hoekstra, 2011). Pastures were also considered to be a land use associated to human activity (i.e. a ‘pasture WF’). The apparently simplistic distinction between ‘natural’ land use and human activities proposed for green water is thus in line with the current developments of the WF and fits with our objective to present a general overview of the integration of the WF within the water cycle.

VIII.4.2 Forests water footprint

The value of forest evapotranspiration was indirectly obtained as the difference between total evapotranspiration and green water consumption for all other land uses, which means errors in these values are aggregated. The resulting consumption of 5,300 m^3/ha is slightly higher than other estimates. i.e. 5,100 m^3/ha in average for Spain (Willaarts, 2012). Since groundwater tables are generally deep in this region and direct groundwater pumping by tree roots is negligible, forests are not consuming blue water. An exception would be the Doñana region (Guadalquivir river estuary), where eucalyptus plantations have been reported as a factor of groundwater depletion (Aldaya et al., 2010c).

VIII.4.3 Water productivity: towards the reallocation of water resources?

A common argument to promote olive grove irrigation is that it presents a higher productivity than many other irrigated crops located downstream, which is confirmed by our results (Figure 8.9). Other crops, such as strawberry or vegetables, are also commonly presented as opportunities because of their high DEV. It should be reminded that DEV constitutes an estimation of the average value generated by the totality of current water use. As discussed earlier, what should be considered for the reallocation of water based on economic productivity considerations is the marginal value (Hanemann, 2006). For instance, in repeated situations of overproduction of olive oil, it could be considered that the increase in production allowed by irrigation makes the prices drop, thus reducing marginal water productivity (see Section 5.3 of this chapter on the case of La Loma de Úbeda for more details).

VIII.4.5 Groundwater availability

The data on 'withdrawals index' (the ratio between withdrawals and 'available resources') is introduced by the Water Plan (CH Guadalquivir, 2010a) (Figure 8.12).

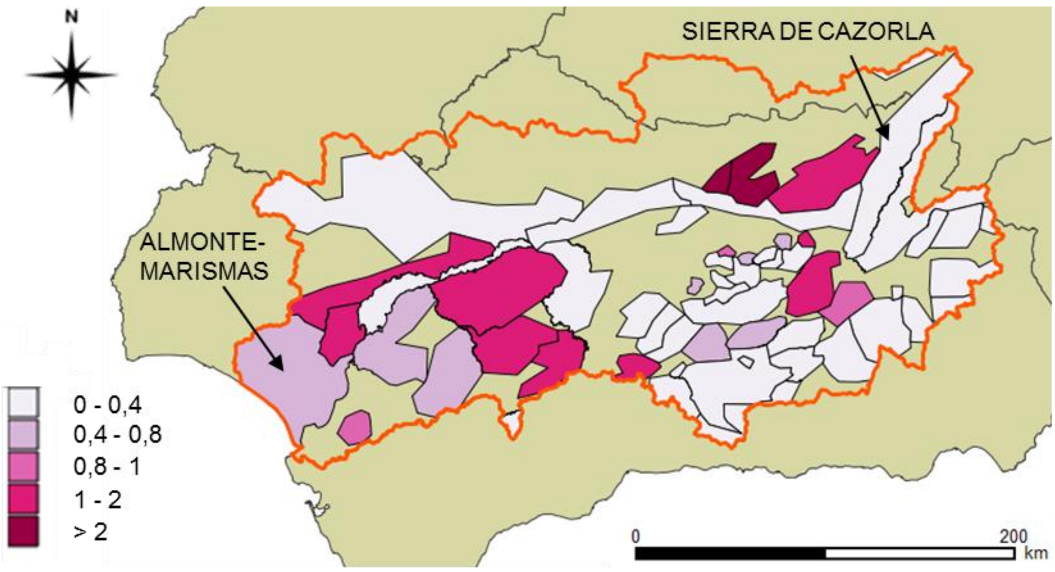

Figure 8.12. Withdrawals index of the groundwater bodies. Source: own elaboration based on CH Guadalquivir (2010a).

Available resources correspond to 80% of the total inflows, except for 'Almonte-Marismas' and 'Sierra de Cazorla' groundwater bodies, where dependant ecosystems are identified as the reason for establishing a higher ‘ecological outflow’ (50% of inflows). When the withdrawals index is equal or higher than 0.8, the groundwater body is classified as in poor quantitative status. This is the reason for the poor quantitative status of 14 groundwater bodies. The decrease in the outflows, lowering of the groundwater table and direct environmental impacts are the other criteria for the classification in poor quantitative status. Five groundwater bodies are in poor quantitative status because of a decreasing groundwater level.

Nevertheless, as presented in Chapter 4 on the assessment of the state of groundwater resources in Spain, the definition of ‘available resources’ does not take into account the need to maintain outflows for downstream use. More precisely, in the case of the Guadalquivir River basin, only 20% of the outflows should be maintained (apart from two groundwater bodies, where the ‘ecological flow’ is considered to be 50% of the outflows). It means that 80% of the water that enters into a groundwater body is identified as a resource for its own users. However, the major part of groundwater withdrawals has a direct impact on surface water flows, through the ‘current capture’ of surface water flows, which has been estimated to amount to 630 hm^3. This water resource consumption should be integrated within the management of the whole river basin. Only the depletion of the aquifer stock (´future capture’) constitutes an additional contribution to the basin water resources for now. The implication of this groundwater stock consumption should be addressed relatively to the long-term impacts.

These remarks are particularly relevant for groundwater-based irrigation of olive groves within the upper stretch of the river basin that have been increasing over the last few years. It constitutes an additional pressure on water resources for the whole basin, affecting potentially all pre-existing users and the environment, now and in the future. Indeed this additional WF makes it harder to implement ecological flows downstream, which is all the more problematic as olive groves irrigation is mainly located in the upper part of the river basin, consuming resources that would have contributed to flows on the major part of the river course.

VIII.4.6 Evolution in the next years and concluding remarks

Agriculture, without the dams WF, represents 80 % of the blue WF. The attractiveness of irrigation comes from the substantial increase in profitability allowed by blue water. However, many users benefit from this resource and environmental and social conflicts are recurrent in a context of high climatic variability. The state of water resources presented in relation to the EU WFD clearly reveals the need for a reduction in the current WF to relieve the pressure on the environment, as well as to improve water quality.

The current and future evolution of water management in the Guadalquivir River basin is based on the development of some common solutions to address the ‘water crisis’: rising efficiency through irrigation modernization and wastewater reuse (as presented in CH Guadalquivir, 2010a). However, these potential solutions should be introduced with caution. In many cases, these actions result in an increase of the WF, for instance through the shift to more water-intensive crops or an increase in the irrigated area thanks to the ‘water savings’. This exacerbates the pressure on other users downstream, including the environment. Indeed, a significant share of the water that is seen as ‘saved’ (thanks to irrigation efficiency or other policies) or a potential source for ‘reuse’ is actually already reused downstream after discharging into the river basin. In this situation, increasing efficiency can hardly be considered as demand management, since the WF increases. On the contrary, it allows retaining water upstream, making it available there, just like a dam (Dumont et al., 2013a, 2013b; see Box 8.1).

In the current situation of quantitative and qualitative pressure on water resources in the Guadalquivir River basin, new uses should not result in a rise of the overall basin WF. Reallocation of water rights should be promoted and be accompanied by an assessment of potential social and environmental impacts, with adequate compensation. This should be taken into account for the development of thermo-solar plants, which could imply an additional WF of 8 hm^3 in the next few years (CH Guadalquivir, 2010a). Effective demand management would improve the possibilities for new water uses that potentially generate more value than agriculture, such as thermo-solar plants or tourism, to obtain water rights from current users. This is also valid within the agricultural sector for the more productive crops.

Groundwater does not constitute an additional resource to compensate for surface water failures and should be fully integrated in this process. Its use, amounting to 720 hm^3 in 2008, reduces surface water availability now and/or in the future, as we have estimated that 86% corresponds to current capture.

Box 8.1. Why a higher efficiency can contribute to increase the water footprint?

1) First reason: local savings are not river basin savings

The usual objective of increasing efficiency in the application of irrigation water (e.g. switching from surface irrigation to sprinkler or drip irrigation) is to reduce the water withdrawals from rivers or aquifers, as less water is applied to meet the crop water requirements. Thus, more water is left at the source. Traditionally, all the reductions in withdrawals are considered as ‘water savings’ and all return flows are viewed as losses (‘Traditional approach’ on Figure 8.13). However, this vision overlooks that part of the water that is recovered downstream for further use in the river basin. Only the reduction in the non-recoverable return flows and non-beneficial ET (see Chapter 6 for more details on the reusability of return flows), would correspond to real savings (‘Detailed accounting’ on Figure 8.13). This issue has been also introduced under the difference between ‘dry’ and ‘wet’ water savings (Seckler, 1996).

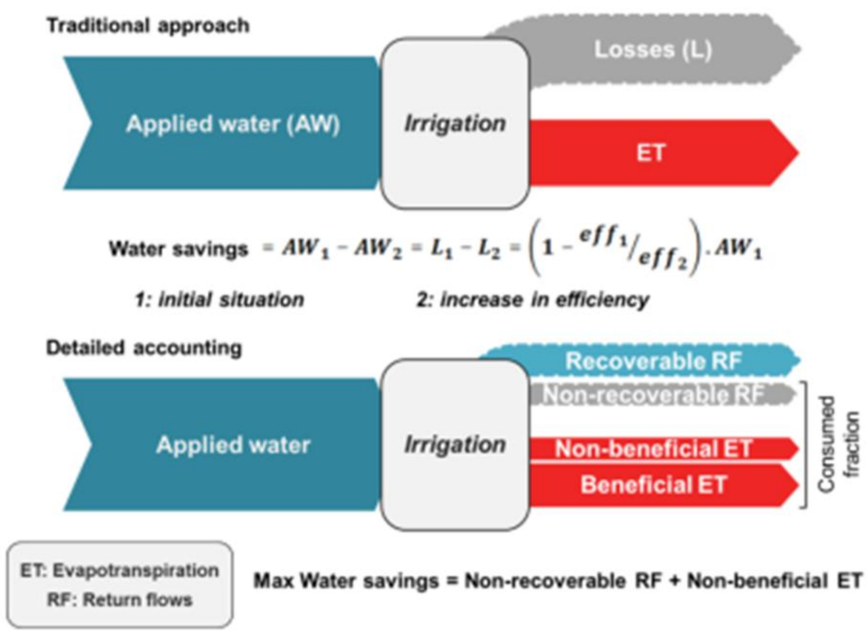

Figure 8.13. Traditional and detailed accounting frameworks to identify the potential savings from an increase in efficiency. Source: Dumont et al. (2013b)

The wrong identification of savings is particularly problematic when these supposed savings are the justification for new uses (e.g. irrigation area extension), since it has the opposite effect of the one sought and results in an additional pressure on water resources (Dumont et al., 2013b; Van der Kooij et al., 2013).

Beyond the agricultural sector, this warning is also valid for any other ‘water savings’ that would be calculated from a reduction in withdrawals, without undertaking a detailed accounting, for example a programme that would encourage urban dwellers to reduce their use of water. Based on the same premise, the reuse of wastewater, e.g. for irrigation, cannot be considered as new resources unless it is previously assessed that it was effectively lost or unavailable for further use, in the initial conditions.

This warning aims at pointing out that there are hardly additional resources available for use in the river basin as a result of ‘water savings’ and efficiency measures. However, these measures have other benefits such has the potential to reduce agricultural contamination (fertilizers are applied more precisely with drip irrigation), to reduce energy consumption in the case of urban water, or to maintain more water in some stretches of river streams and reservoirs, which can be important at the local scale (Gleick et al., 2011), or for resilience to droughts (Dumont et al., 2014).

2) More efficient is more effective and reliable

When a system aimed at applying irrigation water is more efficient (less water is applied to meet the crop water requirement), it is also more effective (it is able to meet better the crop water requirements, i.e. it consumes more water) (Dinar and Zilberman, 1991; Dumont et al., 2013b). This is mainly because more efficient systems such as sprinkler or drip irrigation can be run on demand, which allows irrigating any time the crop needs it. On the contrary, surface irrigation systems are usually irrigated by turns, which implies the crops suffer some stress. Since the crop water requirement is better satisfied, this is accompanied by higher yields. The higher reliability of the system also allows farmers to grow crops that are more sensitive to water stress or that consume more water since it is theoretically possible to have access to more water along the irrigation campaign, thanks to lower withdrawals. The result is an increase in the WF.

Additionally, a spatial effect can be noticed: as a result of a reduced demand, more water is kept in the reservoirs located upstream, which allows satisfying better the demand of the users located in the upper part of the basin. This impacts potentially the users and environment downstream (Dumont et al., 2013b). However, this also allows facing better droughts periods, as the water conserved in reservoirs can be delivered on a larger period, which may imply a more resilient system in some situations (Dumont et al., 2014).

VIII.5 La Loma de Úbeda Aquifer

VIII.5.1 Case study introduction

- <u>Localization and description of the groundwater body</u>

La Loma de Úbeda Aquifer is located at the headwaters of the Guadalquivir River basin, in the Jaén Province, right East of the confluence of the Guadalquivir River and the Guadalimar River (Figures 8.14 and 8.15). The area is characterized by a continental Mediterranean climate, with an average annual rainfall of 611 mm.

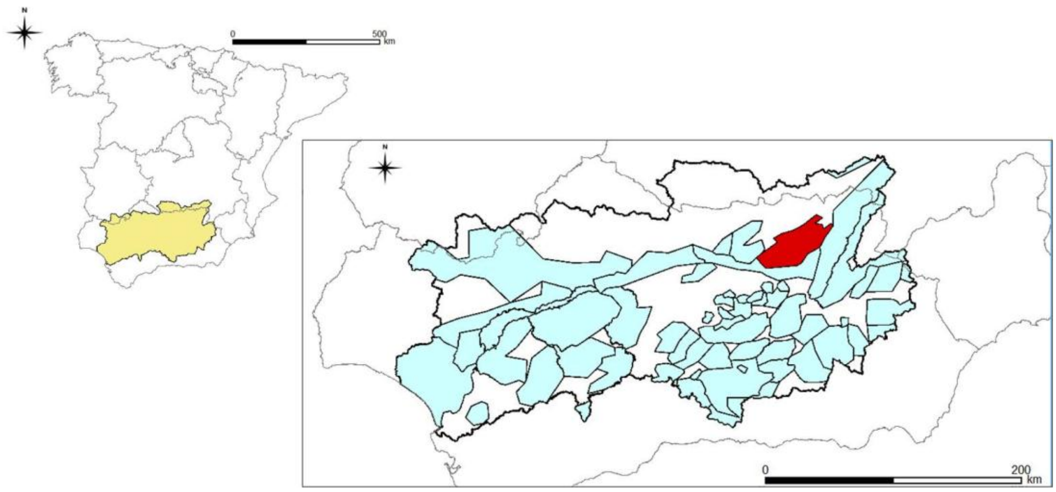

Figure 8.14. Localization of the Úbeda Groundwater Body.

The groundwater body 'Úbeda' covers an area of 1,173 km^2 and has two main aquifers, separated by impervious Miocene marls, meaning that their hydrogeological behavior is independent. The upper Miocene aquifer is totally unconfined. The main aquifer, which is the main object of this section, and referred to as the La Loma de Úbeda Aquifer, is a carbonated aquifer that is confined on the major part of its area, the unconfined area amounting to 232 km^2, located in the Northern part of the aquifer (Figure 8.15). The course of the Guadalimar River on the groundwater body is principally located in this unconfined area.

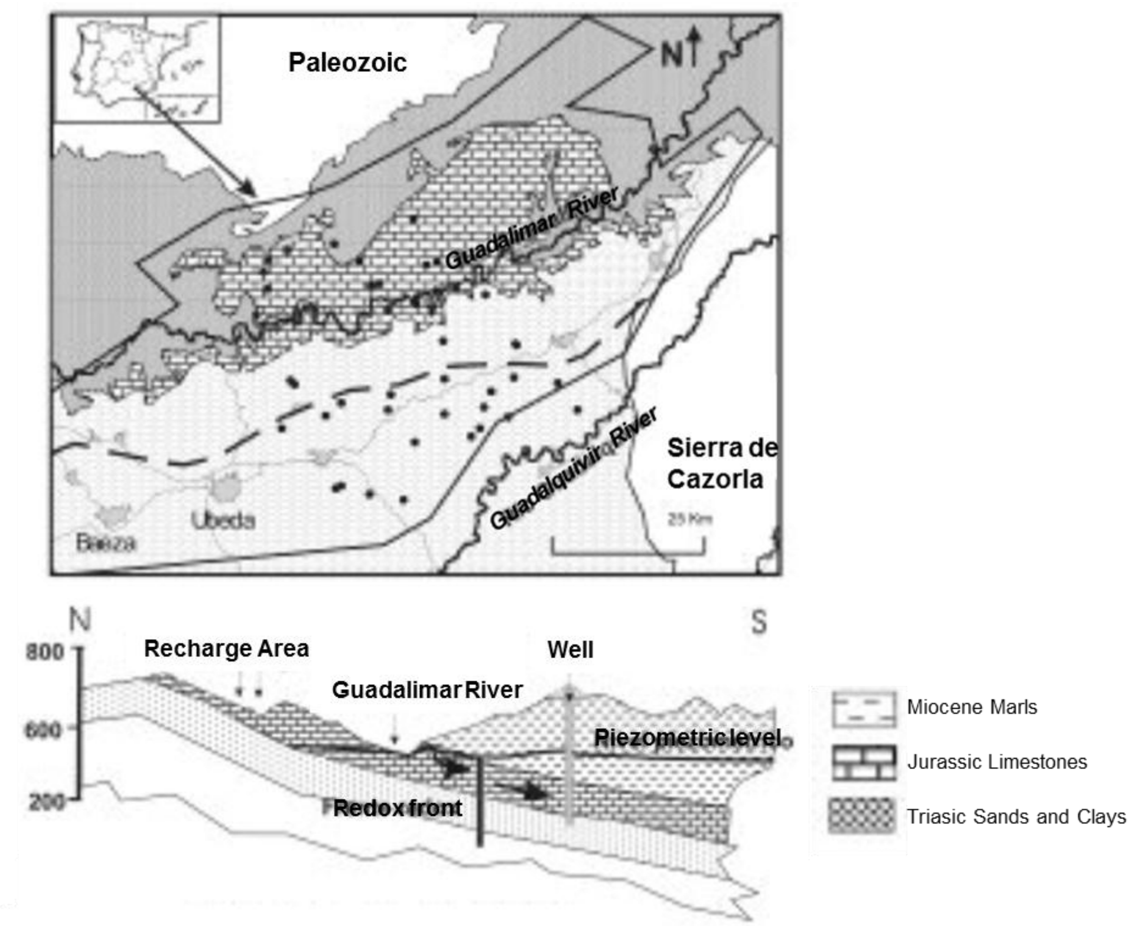

Figure 8.15. Geological and hydrogeological settings of the La Loma de Úbeda Aquifer. Source: adapted from Nuñez et al. (2006)

- Use of groundwater and the issue of illegal use

The water withdrawals in La Loma de Úbeda aquifer are relatively recent, as they are related to the boom of the olive groves irrigation that took place since the mid-1990's and which responded mainly to increased subsidies due to the EU common agricultural policy. The area is now totally dedicated to the monoculture of olives (Plate 8.1).

Some cereals were cultivated until the 1990's but subsidies to support olives represented an end to cereals. An area of irrigated olive groves of more than 25,000 ha, with approximately 46 hm^3 of groundwater withdrawals, i.e. 1840 m^3/ha, was identified for the year 2002 (Gollonet et al., 2002). By the year 2008, the Water Plan estimated the total amount of withdrawals at 80 hm^3, 63 hm^3 related to the La Loma de Úbeda Aquifer, the rest being associated with the unconfined upper Miocene aquifer. Considering a water application of 1,600 m^3/ha[75], it would correspond to an irrigated olive groves area of around 50,000 ha for the whole groundwater body.

[75] Irrigation water application efficiency has improved between 2002 and 2008, mainly thanks to the higher efficiency of areas recently converted to irrigation.

Plate 8.1. Olive monoculture in the Jaén Province. Source: The author

Groundwater use development for olive groves irrigation has undertaken without formal authorization from the water authority of the Guadalquivir River basin. The perforation of the boreholes has been the result of the own initiative from farmers. However, the high investment costs due to the depth of the aquifer on the confined part, with boreholes up to 700m, were unaffordable in most cases for a sole farmer. Thus, neighbouring farmers got together to build boreholes and devised rules to share the pumped groundwater, usually according to their participation in the initial investment (Rica et al., 2014). This mode of access to groundwater is different from the situation of, for instance, the Western Mancha Aquifer, where wells are individually owned by the farmers.

Based on the fact that the Guadalquivir River basin was closed for new water concessions or the perception of an overuse of the aquifer[76], the Water Authorities were initially reluctant to allocate withdrawals rights. Nevertheless, a series of reasons, such as the importance of olive farming for the local economy and political pressures, implied it was hard for the authorities to act against non-authorized use. There has even been a progressive and partial regularization of informal use in the last years.

[76] See Section 5.3 of this chapter for a discussion of the definition of the available resources in the area.

- <u>Aquifer dynamics under natural and current situations</u>

The main origin of recharge is the percolation from rainfall on the unconfined area. Losses from the Guadalimar River and potential underground transfers from nearby groundwater bodies have also been identified. According to CH Guadalquivir (2010a), the whole amount of inflows to the main aquifer is 50 hm^3 (57 hm^3 for the whole groundwater body), however it is not specified if this corresponds to the natural or to the current situation. The major part of the natural discharge area goes to the Guadalimar River. Thus, naturally the Guadalimar River receives 50 hm^3 as an average from the aquifer.

Under natural conditions, the confined part of the aquifer is full and the movement of groundwater is very slow in this part (Figure 8.15). Nuñez et al. (2006) have identified groundwater more than 25,000 years old. Only the unconfined part in relation with the Guadalimar river is active. Rainfall percolates into the aquifer and discharge to the river, which is a gaining stream as a whole, even if some sections are losing streams. Nevertheless, groundwater pumping in the confined part has altered this functioning. The Guadalimar River is a source of capture and groundwater now flows more rapidly from the unconfined section to the confined section, which used to be the place of a very slow groundwater movement. This compensates for pumping (see Chapter 2 on groundwater dynamics). In confined aquifers the perturbation due to pumping is especially quickly propagated to the source of capture in the aquifer.

VIII.5.2 Specific method for water footprint accounting

The carbonated aquifer is a confined aquifer for its major part. Consequently, it will be assumed that the WF is constituted by the whole groundwater withdrawals as presented by CH Guadalquivir (2010a). An irrigation water application of 1,600 m^3/ha is considered.

The economic water productivity is presented for the period 1999-2011, in order to show the variations depending on olives market price. The price of olives is hardly presented in the official statistics. The value included usually refers to olive oil prices. Notwithstanding, the price of olives paid to farmers is introduced by the Observatorio Andaluz del Empleo Agrario (2013) for the period 2005-2007. The price in the Jaén Province is then around five times lower than the price of olive oil of intermediary quality (*aceite fino*). This ratio is contemplated for the whole period of study and the price of the intermediary quality oil, which is used as a reference for the period 1999-2011, is obtained from MAGRAMA (2013). The number of working days has been obtained from Junta de Andalucía (2002).

VIII.5.3 Results and discussion

- <u>The water footprint of olives and olive oil in the La Loma de Úbeda Aquifer</u>

Based on CH Guadalquivir (2010a), the WF of the main aquifer reaches 63 hm^3, the withdrawals reaching 80 hm^3 for the whole groundwater body. This should be compared to the WF of 23 hm^3 reported for 1999 and 46 hm^3 for 2002 by Gollonet et al. (2002). However, these values may underestimate the real WF, since informal withdrawals may have not been properly reported. The market value of irrigated olive production in the area has reached around 180 million euros per year for the period 2008-2011 (Figure 8.16). However, the additional value compared to rain-fed production is only 50 million euros per year. Higher values have been obtained in the years 2006 and 2007, with 224 million euros and 214 million euros respectively, in spite of a lower irrigated area. This is due to a lower olives market price for the period 2008-2011 (Figure 8.17).

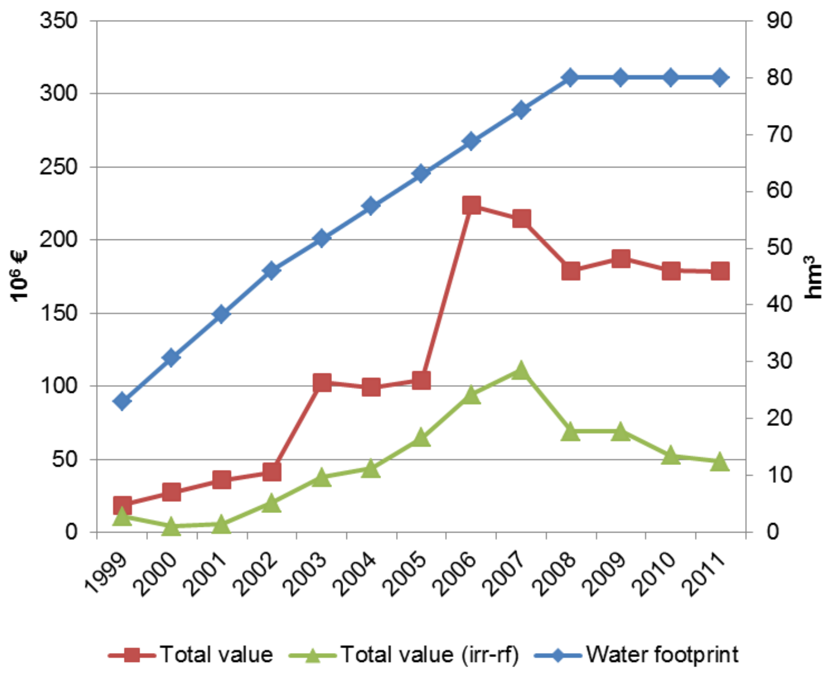

Figure 8.16. The water footprint of the La Loma de Úbeda Aquifer and market value of the irrigated production with and without integrating rain-fed production for the period 1999-2011.

The Direct economic value (DEV) of water has stabilized around 1.4 €/m^3, or 0.5 €/m^3 taking into account the value of rain-fed production. The effect of market prices is also one of the main reasons for the evolution of the DEV (Figure 8.17).

With regards to the labour generated by olive growing, which is an essential activity in many rural areas, it is not meaningful to associate employment with the use of water. Irrigation generates more jobs, as compared to rain-fed olive growing, based on one hectare of olive groves with the assumption of manual harvest (around 28 working days/ha as compared to 23 working days/ha, Figure 8.18).

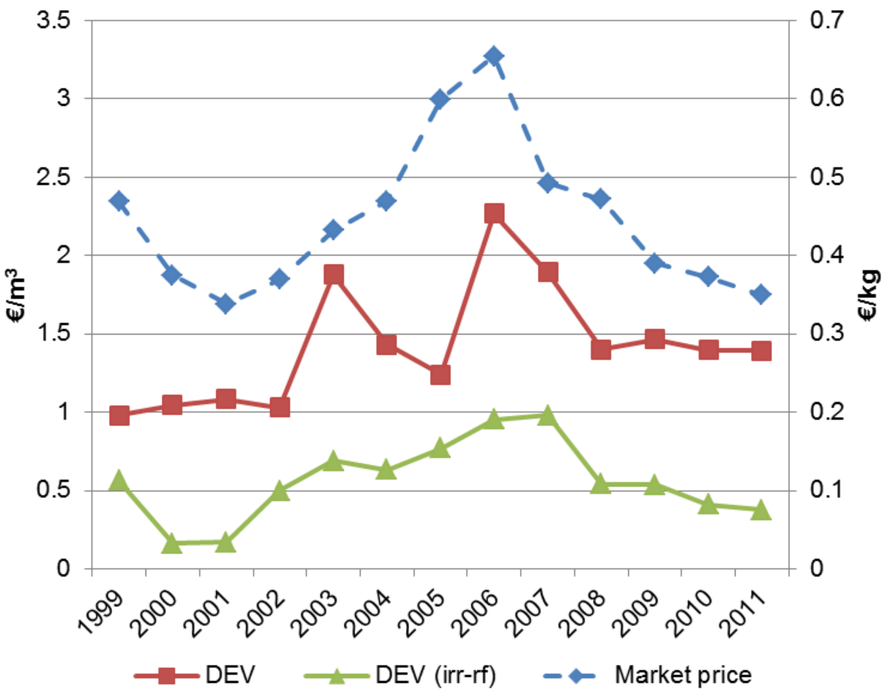

Figure 8.17. Market price of olive and direct economic value of water based only on irrigated production and integrating the value of rain-fed production for the period 1999-2011.

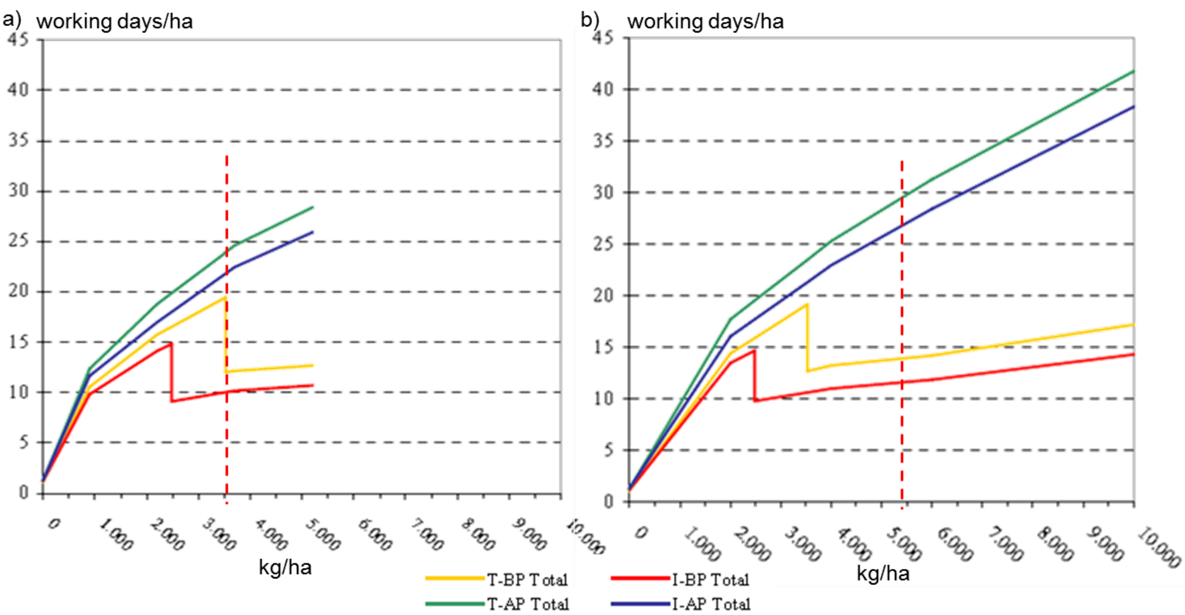

Figure 8.18. Working days for one hectare of olive grove for rain-fed (a) and irrigated (b) productions. Source: adapted from Junta de Andalucía (2002)

T-BP Total: traditional, low slope; T-AP Total: traditional, high slope; I-BP Total: intensive, low slope; I-AP Total: intensive, high slope. Red dot lines show the working days corresponding to the Jaén Province (3,200 kg/ha (rain-fed) and 5,200 kg/ha (irrigated) as an average on the period 1999-2011).

However, as irrigated olive growing often takes part in a more intensive mode of production, with higher yields, the number of jobs per kilogram of olive produced is lower in irrigated system (5 working days by tonne of olive as compared to 7 days). Also, the mechanization of olives harvest is more common in irrigated growing than in rain-fed production, which implies a loss of jobs. This is visible on the Figure 8.18 by the drop in working days for the production on low sloping land when the yield rises.

The irrigation of olive groves is often presented as a way to generate economic activity and jobs in depressed areas. This is linked to the fact that irrigating olive groves makes this crop profitable in the current context of low prices. Traditional rain-fed olive growing is not profitable, which jeopardizes the associated jobs. However, this is questionable since irrigation is one of the factors contributing to the fall of olive oil prices, thus triggering a vicious circle.

- Implication of the development of irrigated olive groves on production and prices

Two major drivers for the adoption of irrigation for olive groves in the area have been the drought of the mid-1990s and the EU CAP subsidies. These subsidies were directly tied to production. Other problematic issues driven by subsidies concern the use of fertilizers and the erosion resulting from new agricultural practices (Scheidel & Krausmann, 2011).

The subsidies were initially justified by the specific nature of olives, since it is often viewed as a traditional Mediterranean crop, adapted to the climatic conditions, which has a low demand for inputs and constitutes a characteristic feature of the landscape. Additionally, it generates employment since its labour requirements are high. Rainfed olives growing is hardly profitable without subsidies. However, this vision is applicable to only part of a production system that is very diverse. Three main types of olive production can be distinguished (Table 8.4).

Table 8.4. Main types of olive production.Source: Beaufoy & Pienkowski (2000)

Type	Description by Beaufoy & Pienkowski (2000)
Low-input traditional plantations and scattered trees	Often with ancient trees and typically planted on terraces, which are managed with few or no chemical inputs, but with a high labour input.
Intensified traditional plantations	Follow traditional patterns but are under more intensive management making systematic use of artificial fertilisers and pesticides and with more intensive weed control and soil management. There is a tendency to intensify further by means of irrigation, increased tree density and mechanical harvesting.
Intensive modern plantations	Smaller tree varieties, planted at high densities and managed under an intensive and highly mechanised system, usually with irrigation.

The subsidy structure has changed from the year 2005-2006 with the main part linked to past production and with a share still associated to production but with conditions regarding the environment and agricultural practices (European Commission, 2010). Nevertheless, irrigation implies a substantial advantage compared to rainfed production, since higher yields mean higher profits and a kind of assurance for the driest years. As the development of irrigation implied an increase in the production of olive oil, principally through less variability in production, stocks have accumulated (Figure 8.19). This has directly contributed to lower the market price of olives. It is the rainfed traditional production that has been paradoxically more affected by the drop in prices. According to the Junta de Andalucía (2013), traditional production farmers have lost between 0.2 €/kg (moderated slope) and 0.37 €/kg of olive oil (high slope), while irrigated production is profitable with 0.12 €/kg of olive oil margin for the campaign 2009/2010[77]. This drives the development of irrigation, which is the only way to remain profitable (a 'snow-ball' effect or vicious circle similar to the situation of irrigated vines in the Western Mancha Aquifer).

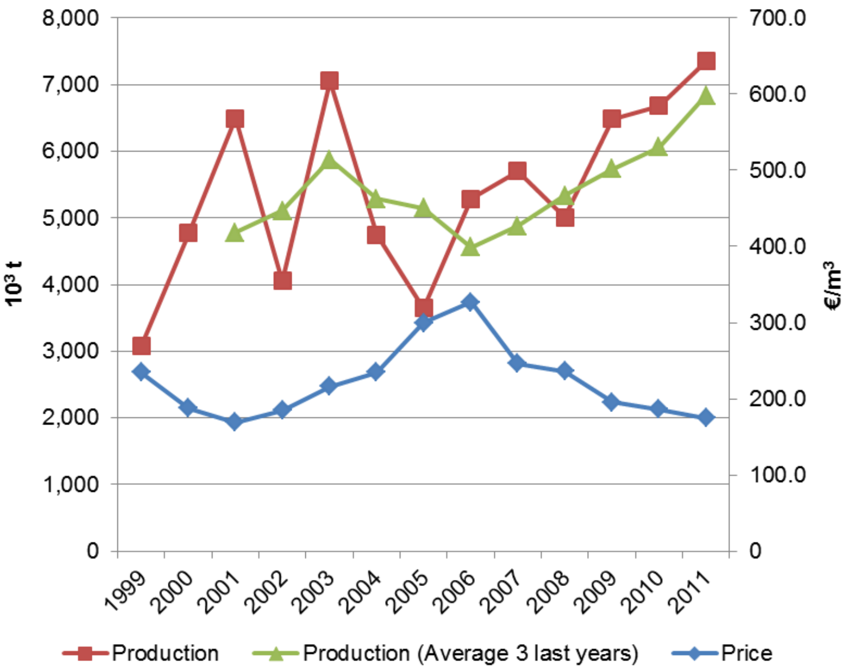

Figure 8.19. Production (annual and averaged on three years) and price of olive oil in Spain. Production averaged on three years is presented as a proxy for the evolution of the olive oil stocks, which have an influence on prices.

Nevertheless, the overproduction triggered by irrigation is often not appreciated. Instead the view that exports constitute a solution is very common. Yet, even though the external market is growing, there are uncertainties on the evolution of this trend. Furthermore, in case the

[77] These results do not take into account subsidies. Losses do not mean that the activity will cease. It implies that some costs are not fully paid; typically the working hours of the farmers or the employees.

development of international sales would imply higher prices, the irrigation of olive groves would be acceptable only if the water rights were granted, giving security of tenure.

Irrigation could be justified on the basis of profitability and the fact that the higher yields obtained allow for a profitable activity that does not need subsidies. Water productivity can also be higher than other crops. Conversely, traditional olive growing is unprofitable without subsidies. However, this vision does not integrate positive externalities, such as the value of traditional olive groves maintenance for land management and landscape.

- Groundwater availability and the regularization of informal use

According to the criteria established by the Guadalquivir River basin authority, eighty per cent of the estimated recharge to the main aquifer, i.e. 40 hm^3, is considered as available resource (CH Guadalquivir, 2010a). Yet the assessment of the groundwater body in the Water Plan has clearly established that the natural outflows go to the Guadalimar River. Thus, the predominance of the traditional view means that the capture of these flows is not taken into account. This is particularly problematic in the context of a confined carbonated aquifer, since the perturbation generated by pumping rapidly propagates to capture areas.

The groundwater pumped has some chemical characteristics and temperature that can make think that we are in a situation of ‘groundwater mining’. Although it could be referred to as fossil groundwater, since it is sometimes more than 25,000 years old (Nuñez et al., 2006), the term ‘mining’ is not accurate here since impacts from pumping affect current generations through river flows consumption (see Section 4.7 of Chapter 3).

The ‘groundwater availability’ of 40 hm^3 defined by hydrogeologists has been used as the basis for the claim from aquifer users to get rights. This shows the importance of hydrogeologists in the process of decision-making, but also questions the role of technical expertise and the robustness of scientific knowledge to take decisions. After a few years, with the authorities quite reluctant to allocate new rights, the process of regularisation of informal users on the basis of available resources has started in La Loma de Úbeda Aquifer. It is however unclear if this will imply action against informal users who will not have rights granted (potentially half of the current withdrawals of 80 hm^3). The process of regularization of informal groundwater users in La Loma de Úbeda Aquifer is a complex issue that is driven by political interests and cannot be detailed here (for more details, see Rica et al., 2014). The aim of this section was to show the implication of the irrigation of olive groves for the local economy and how the definition of available resources is key, particularly when the dynamic of groundwater use mean that a river is directly affected.

VIII.6 Synthesis

- In the WF accounting at the scale of the Guadalquivir River basin, in addition to general accounting of the blue and green WF (7,280 hm^3 and 2,790 hm^3, respectively), this chapter has innovated by specifically considering the WF from dams (315 hm^3), current and future capture (730 hm^3 and 100 hm^3, respectively), and the integration of the green and blue WFs in the whole water cycle.
- This chapter illustrates that the innovative approach of the WF in terms of 'water resources appropriation' is also subject to debates for the attribution of water consumption between human and 'natural' use, particularly for green water.
- Differences in Direct Economic Value for crops shows that more value could be potentially obtained using less water. Nevertheless, water rights reallocation is more complex than usually considered as it implies a change in the conditions of use and a loss of revenue for some users, which they may not accept.
- An increase in the use of groundwater in the last years, particularly for olive groves irrigation, has implied a rising pressure on surface water in the whole river basin, which jeopardizes environmental flows and water quality. As groundwater withdrawals result in current and future capture, they should be integrated within water resource management at the scale of the whole river basin, which is poorly recognized so far.
- The Guadalquivir River basin is illustrative of the problem of considering groundwater availability at the scale of the groundwater body. For instance, the Water Plan (CH Guadalquivir, 2010a) has established an available resource of 40 hm^3 for the La Loma de Úbeda Aquifer. This value does not integrate the fact this water was naturally flowing to the Gualdalimar and Guadalquivir Rivers. Yet is has been the basis for the regularisation demands of users and rights attribution.
- Olive grove irrigation has increased production, which has reduced prices and jeopardized the less intensive traditional olives growing, thus triggering a 'vicious circle' towards the intensification of production and irrigation.

Chapter IX

CAMPO DE DALÍAS AQUIFER

IX. CAMPO DE DALÍAS AQUIFER

Objective: The main objective of this chapter is to assess the water footprint of an aquifer located on a coastal area, which implies a series of particularities regarding the impacts of groundwater pumping, such as a potential seawater intrusion. While the current water origin is mainly groundwater, future water supply options, which include sea water desalination, are contemplated, in a context where groundwater is used by multiple sectors that have to secure their water supply. A more specific aim is to assess the agricultural water footprint of an intensive agricultural system: the case of vegetables grown in greenhouses.

IX.1 Introduction

IX.1.1 Location and definition of the case study boundaries

The study area is located in the south of the Province of Almería, which is the easternmost province of Andalusia (Figure 9.1). It consists in southern part of the 'Campo de Dalías-Sierra de Gádor' groundwater body as defined by the Water Plan of the Mediterranean River Basins (Agencia Andaluza del Agua, 2010)[78]. The 'Campo de Dalías' area corresponds to a coastal platform, surrounded by the Mediterranean sea, except to the North, where it is bordered by the Sierra de Gádor Mountains. In the absence of rivers in this area, the only natural source of water is groundwater. This chapter focuses on this coastal plain, as one of the main objectives is to characterize the WF from the intensive production in greenhouses, which are entirely located there. Based on Junta de Andalucía (2010a), it can be estimated that intensive agriculture in the coastal platform, located in the municipalities of El Ejido, Roquetas de Mar, Vicar, La Mojonera and part of Berja, corresponds to 92% of the groundwater withdrawals for agriculture from the whole groundwater body (Figure 9.1). Urban withdrawals are also almost entirely located there. Thus, the study focuses on these municipalities.

More exactly, the case study contemplates all groundwater users, even if they are located outside the boundaries of the aquifer. This refers particularly to the urban supply of the city of Almería from the 'Rambla Bernal' wells (located in El Ejido).

Regarding climatic conditions, the average annual rainfall is 196 mm (1971-2000 in Almería), with an annual mean temperature of 18,7 °C (minimum 12,5 °C and maximum 30,7 °C) (Frot et

[78] In this chapter, the references to the 'Water Plan' refers to the 'Water Plan of the Mediterranean River Basins of Andalusia ' (*Plan hidrológico de las Cuencas mediterráneas*) that has been elaborated by the Water Agency of Andalusia (Agencia Andaluza del Agua, 2010), which is the organism in charge of water planning and management in the case study area.

al., 2007). These climatic factors are one of the main reasons for the development of intensive agriculture in the area, where the smooth winter temperatures make possible the cultivation of horticultural products in greenhouses during winter[79]. Meanwhile water availability is assured by the aquifer recharge in the more rainy Sierra de Gádor Mountains.

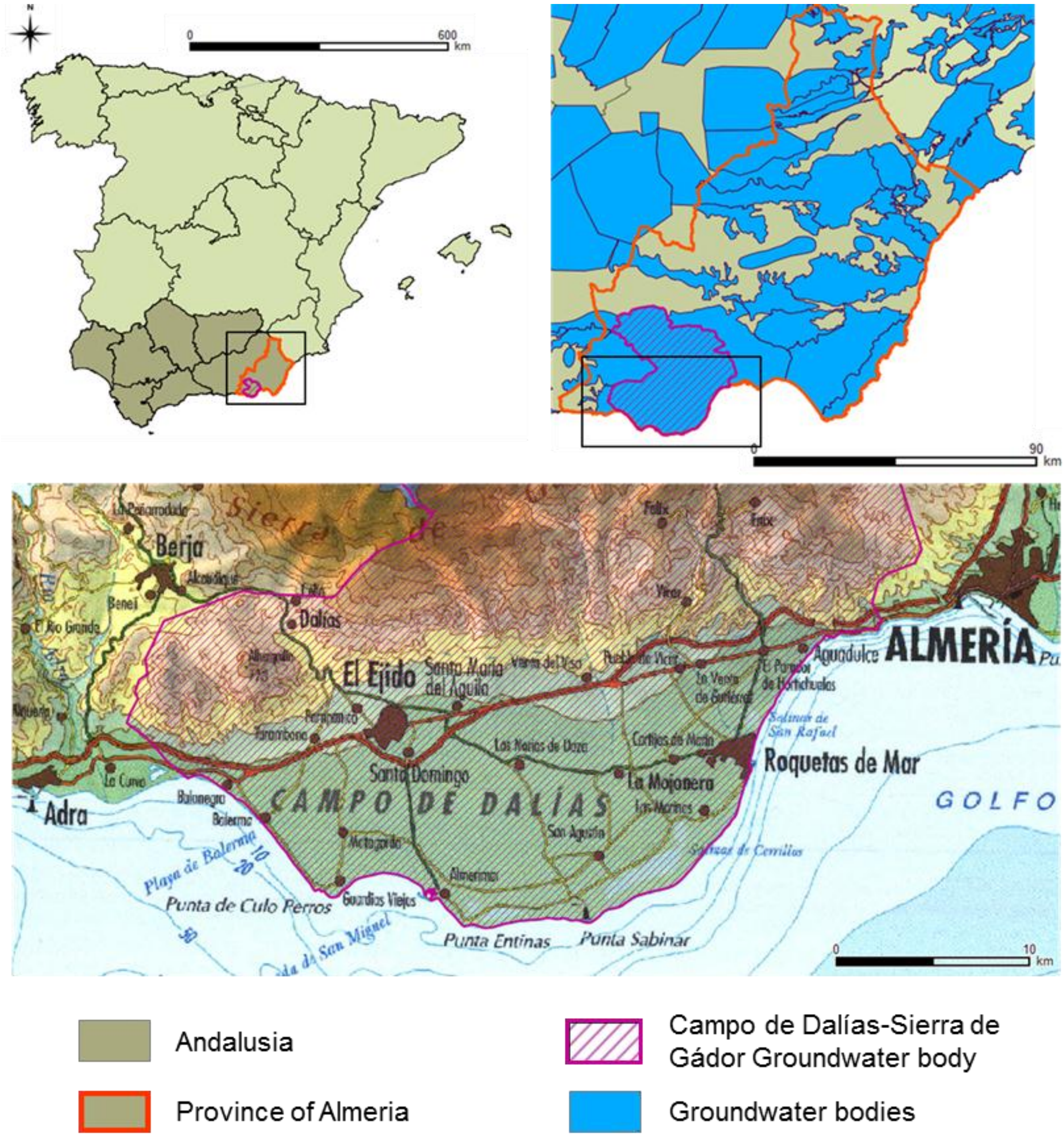

Figure 9.1. Location of the Campo de Dalías and the associated groundwater body.

[79] The specific shape of the coastal platform implies that the area is also prone to be swept by the winds, which also plays a role in making the area favorable for greenhouse agriculture, particularly during the hotter periods of the agricultural campaign. These conditions are virtually not found anywhere else on the Spanish Mediterranean coast.

IX.1.2 Geological and hydrogeological settings

In the Campo de Dalías coastal platform, three main units can be distinguished (Figure 9.2):

- The main Aquifer System is constituted by two Triassic units of carbonates resting on metapelite (Frot et al., 2007; Pulido-Bosch et al., 2003). These units appear on the surface of the whole northern part of the groundwater body and particularly on the Sierra de Gádor Mountains, which is an E-W oriented anticline. This thick carbonate layer (up to 600m) is highly permeable thanks to its fractured and karstic nature. It is also very compartmented.
- Upon the Campo de Dalías coastal platform, this layer is covered by a thick aquitard made of Miocene and Pliocene materials, with a basal conglomerate and an impermeable marl formation.
- Pliocene calcarenites, gravels, sands and conglomerates (100-150m) and the permeable quaternary alluvial deposits constitute a superficial aquifer unit.

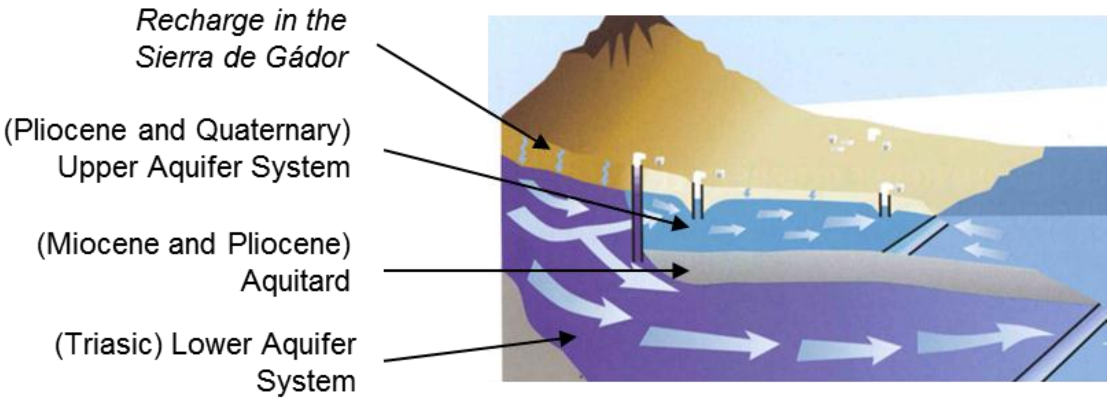

Figure 9.2. General model of the Campo de Dalías Aquifer with limited withdrawals. Source: adapted from Junta de Andalucía (2010b)

To sum-up, the main aquifer units are constituted by the thick fractured carbonate Lower Aquifer System, which currently is the main source of water, and the thin Upper Aquifer System, which used to be the main source of water at the start of groundwater use in the area, but it is currently hardly used because of the lower water quality. The Lower Aquifer System is recharged from the Sierra de Gádor Mountains, thanks to the infiltration occurring mainly during intensive rainfall events, typical in the Mediterranean climate (Pulido-Bosch et al., 2012). The value of the recharge is uncertain, with a value of 50 hm^3 according to Pulido et al. (1993) and 79 hm^3 according to ACUAMED (2006). In natural conditions, part of this recharge flows through the Upper Aquifer System, which also receive water from rain water infiltration. Even if the general distinction between the Lower and Upper Aquifer Systems represents a first picture of the area, it is necessary to introduce the different sub-aquifers (Figure 9.3).

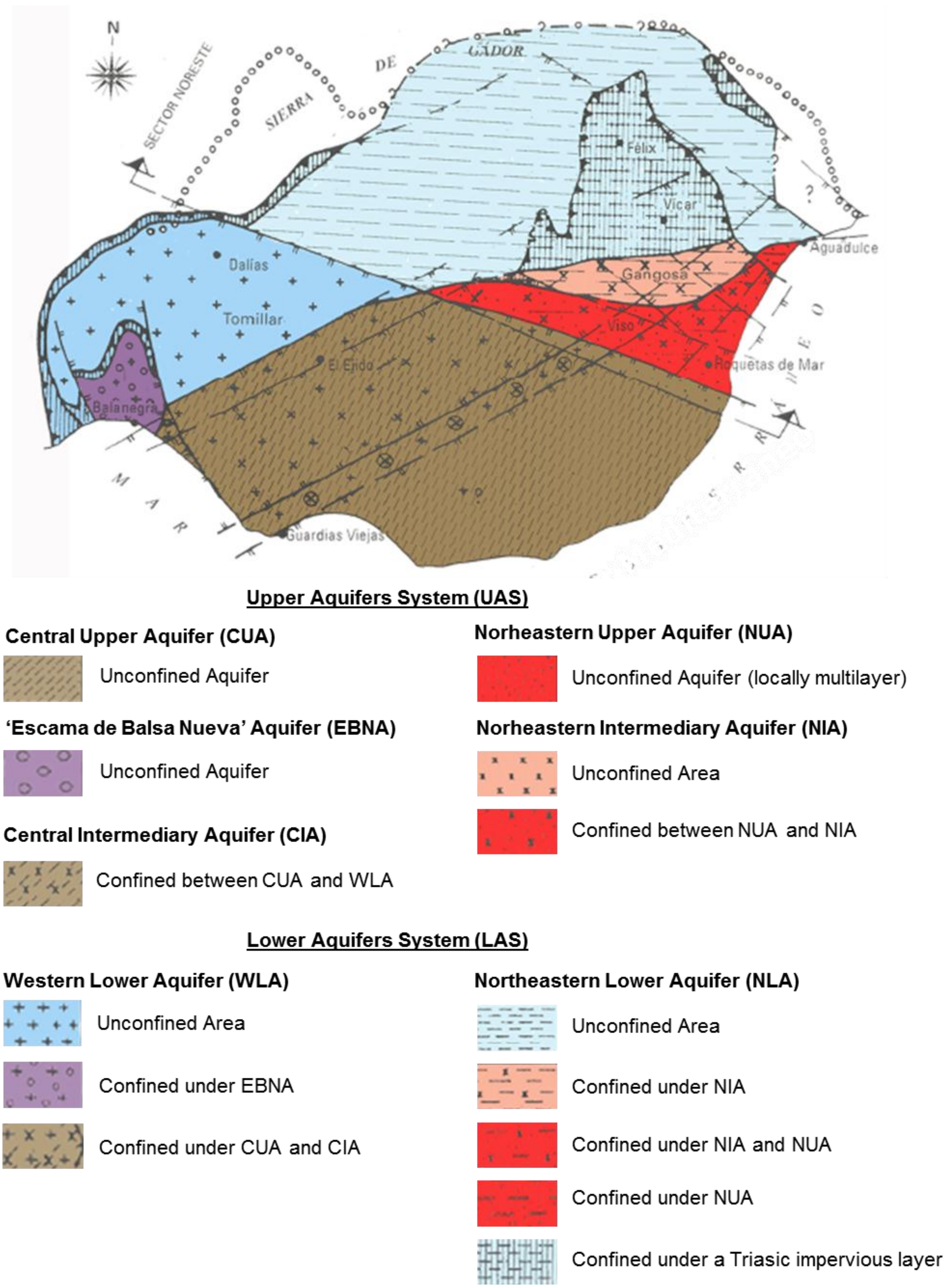

Figure 9.3. Definition of the main aquifers of the Campo de Dalías. Source: adapted from González-Asension et al. (2003)

The Lower Aquifer System is composed of the North-eastern Lower Aquifer and the Western Lower Aquifer, which are separated by a major fault. The Upper Aquifer System is composed of the North-eastern Upper Aquifer, the North-eastern Intermediary Aquifer, the Western Upper Aquifer, and the Escama de Balsa Nueva Aquifer. Many of these aquifers are connected to one another and to the sea. This implies a complex groundwater flow system. Moreover, some local aquifers are also present.

Another classification of the different aquifer units is presented by Molina et al. (1998) (Figure 9.4), with the Aguadulce Aquifer corresponding basically to the North-eastern Aquifers, the Balanegra Aquifer to the Western Lower Aquifer and Escama de Balsa Nueva and Balerma-Las-Marismas Aquifers to the Western Upper Aquifer.

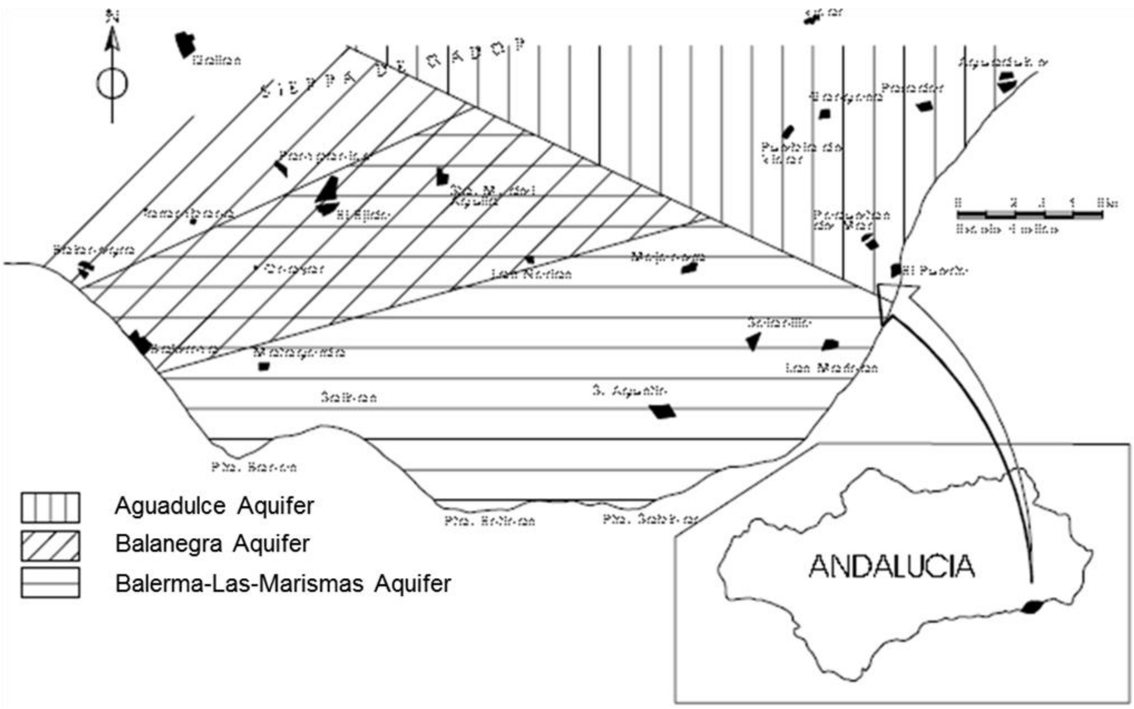

Figure 9.4. An alternative description of the aquifers of the Campo de Dalías. Source: adapted from Molina et al. (1998)

IX.1.3 History of the agricultural development

The development of the area started with the first ‘Project of Colonisation’ in 1953, after the area had been classified of interest in 1941 by the National Institute of Colonisation, in spite of the adverse climatic conditions for agricultural activity (Molina et al., 1998). The objective of this project was to develop the area and to settle small farmers, through irrigated agriculture from groundwater use. The role of this public initiative has been essential, as a catalyst for groundwater use. However, much of the territory was also developed under the initiative of private individuals, who were initially big landowners who equipped their land with irrigation facilities before selling it on (sometimes in order to prevent a possible expropriation) (Rivera-Menéndez, 2000). Private initiative also took the form of small landowner groups, who invested together in a well in order to have common access to water[80]. These two processes – public enterprise with the subsequent ceding of the wells to farmers, and private initiative – generated the current landscape of Irrigation Communities.

The development of a highly productive irrigated agriculture, and a boom in groundwater use (Figure 9.5), led to the economic development of the area, which has been one of the poorest regions in Spain. The GDP per capita in the province of Almeria rose from less than 60 % of the GDP per capita of Spain in 1955 to almost more than 95 % in 2000 (Ferraro & Aznar, 2008).

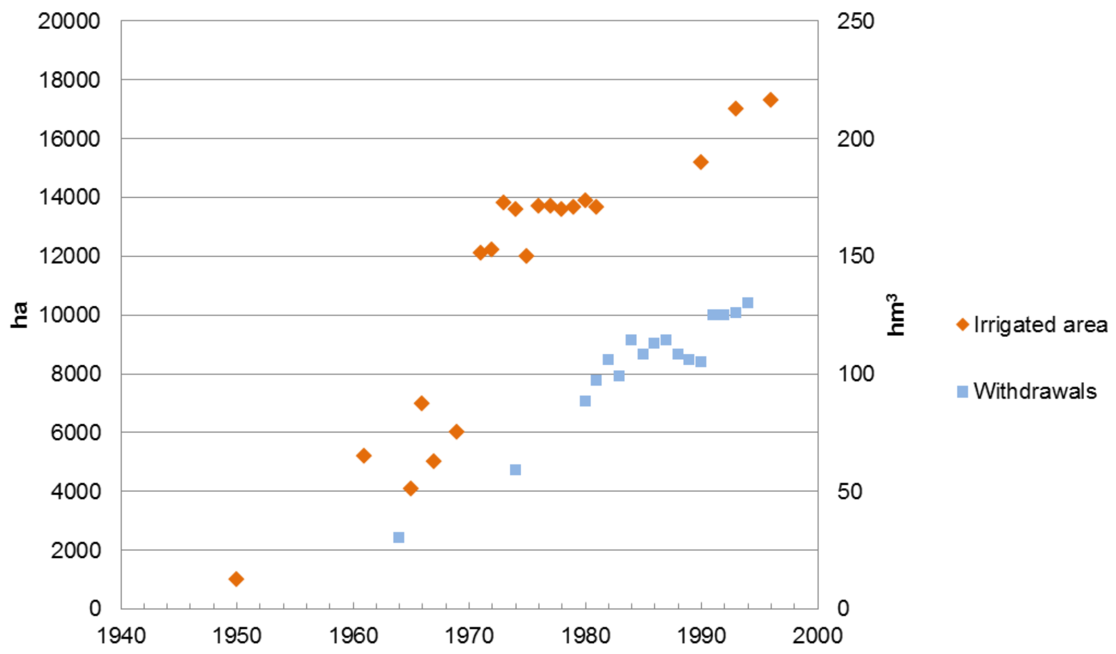

Figure 9.5. Irrigated area and groundwater withdrawals on the period 1950-2000. Source: adapted from Cuitó-Sabaté et al. (2006).

Climatic conditions and access to groundwater partly explain the high profitability of the agriculture in the area. However, technological innovations have also played a major role, e.g. the use of the *enarenado* technique, which consists in adding a layer of sand (*arena* in Spanish) to conserve water in the soil, and the introduction of the greenhouses. These innovations limited the adverse effects of natural conditions and enhanced the natural competitive advantages, e.g. the smooth winter temperatures kept stable thanks to greenhouses[81].

[80] The return on investment was assured thanks to the high profits allowed by irrigation. The Banks, which loaned the money, have also played a major role in this process.

[81] Another major innovation that took place in the last year is the biological control of plagues through the use of predatory insects. It allowed reducing the use of pesticides significantly.

Another reason for the success of the model has been its capacity to adapt to very profitable crops, originally with the production of a specific type of grape that endured boat transport, and allowed its export, and, subsequently, with the production of horticultural crops during winter. Access to the European market, due to the entrance of Spain in the European Economic Community in 1985, has also been essential.

All these factors contributed to the dynamism of the agricultural sector, which was also tied with a constant increase in groundwater use. This rise has been limited thanks to the introduction of another technical innovation: drip irrigation technology. Indeed, efficiency in groundwater use is the reason that explains why groundwater withdrawals have not increased as much as the irrigated area (Figure 9.5).

IX.1.4 Origin of water

In addition to groundwater, water resources from other origins are used in the case study area:

- *Surface water from the Beninar reservoir*. This reservoir has been built in order to constitute an additional resource for the city of Almeria. However, an agreement was made between the city and an Irrigation Community of Campo de Dalías in order to let farmers use this source of water, while Almeria would use the equivalent amount of groundwater from the Campo de Dalías Aquifer.
- *Desalinated water*. Currently, there is no use of desalinated water in the Campo de Dalías as such; however, this source of water is already used by Almeria and should therefore be considered in the overall pool of water resources, as it constitutes a substitute resource of groundwater. Furthermore, a desalination plant located in the Campo de Dalías is due to be put into service in 2015.
- *Rainwater*. The radical shift in land use due to greenhouses on much of the territory implies that rainwater no longer infiltrates or is stored in the soil. In order to prevent the flooding of roads, farmers retain rain water falling on their greenhouses: some farmers use it to irrigate, while some other inject it directly in the upper aquifers through wells.

IX.1.5 Impacts of groundwater pumping and land use changes on water resources

As described in Section 1.2 of this chapter, the main origin of water is recharge from the Sierra de Gádor Mountains to the Lower Aquifer System. Only the eastern side of the Lower Aquifer System is in direct contact with the sea (through the North-eastern Lower Aquifer, on the surroundings of the municipality of Aguadulce[82]). It is a first natural area of discharge for the

[82] *Agua dulce* means fresh water, which is probably related to the fact that fresh water outflows from the aquifer to the sea take place (or used to take place before intensive groundwater use).

aquifer. Another main natural discharge area from the Lower Aquifer System is through the transmission of flows to the upper aquifers that subsequently discharge to the sea[83]. This process also takes place naturally on the western part of Campo de Dalías, in the area surrounding Balanegra (see Figure 9.1), where the Western Lower Aquifer is connected with the superficial Escama de Balsa Nueva Aquifer, which is itself connected to the sea.

The main consequence of groundwater intensive use from the 1970s has been a constant decline in the groundwater level in the Lower Aquifer System, with groundwater tables clearly below the sea level from the beginning of the 1980s (Figure 9.6). Consequently, marine intrusion started to occur in the two areas of direct (North-eastern Lower Aquifer) and indirect (through the Escama de Balsa Nueva Aquifer) contact between the Lower Aquifer System and the sea:

- In the North-eastern Lower Aquifer, the phenomenon of marine intrusion was noticed rapidly, at least in the areas closer to the sea, where some wells had to be abandoned, but pumping continued in the innermost areas (Domínguez-Prats et al., 2012; Molina et al., 1998; González-Asension et al., 2003). However, the intensification of marine intrusion has been confirmed recently (Figure 9.7).
- In the Western Lower Aquifer, the probable marine intrusion through the Escama de Balsa Nueva Aquifer has been identified since the beginning of the 1980s (Domínguez-Prats et al., 2012; Molina et al., 1998). The level reached -48 m in the Western Lower Aquifer in 2008-2009 (Figure 9.6). In the context of a coastal aquifer, the actual possibility to reach this value confirms that the aquifer was isolated from the sea, at least partially. It may be linked to the fact that saline flows have to go first through the Escama de Balsa Nueva Aquifer. It resulted in salt intrusion detected for the first time in 2010 in the bottom of some wells in the Western Lower Aquifer (Figure 9.8).

[83] Some springs were located also at the foot of the Sierra de Gádor Mountains, constituting a source for local public water consumption.

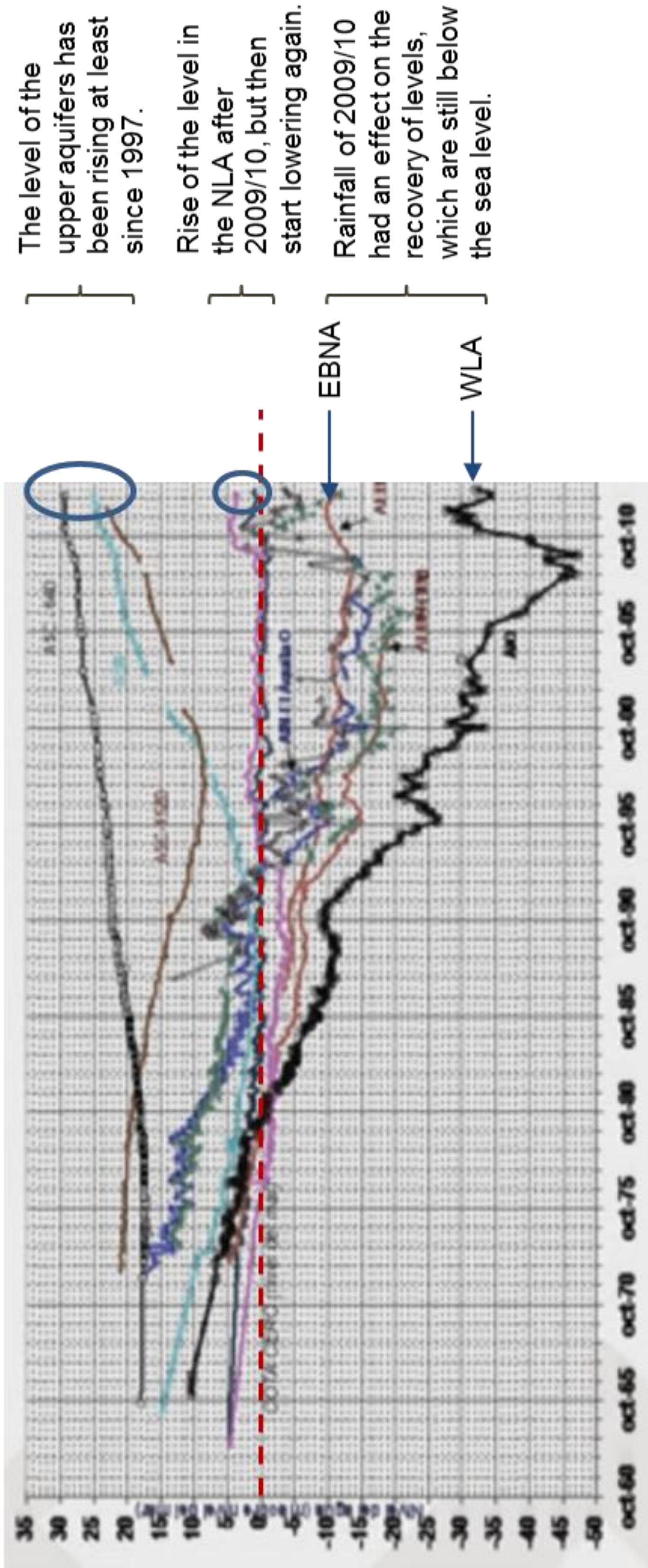

Figure 9.6. Evolution of groundwater levels in the different aquifers of Campo de Dalías. Source: adapted from Domínguez-Prats (2013)
EBNA: Escama de Balsa Nueva Aquifer / WLA: Western Lower Aquifer / NLA: North-eastern Lower Aquifer

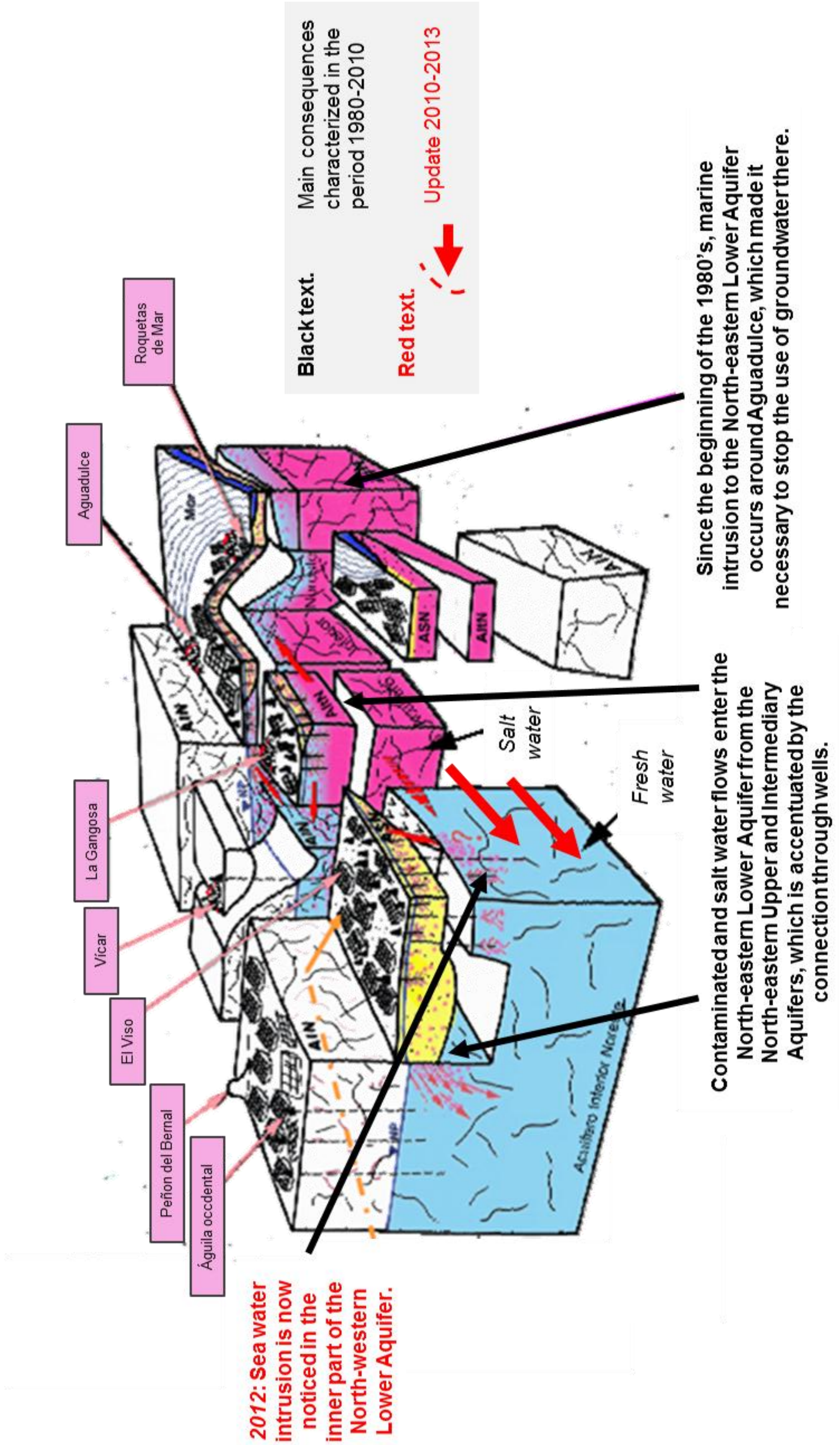

Figure 9.7. Saltwater intrusion on the North-eastern part of the Campo de Dalías Aquifer. Source: adapted from Domínguez-Prats et al. (2012) and González-Asensio et al. (2003)

In addition to marine intrusion, another source of contamination of the Lower Aquifer System is the percolation of flows from the Upper Aquifer System, which are contaminated by nitrates or suffer from salt intrusion. For instance, at the foot of the Sierra de Gádor Mountains, where outflows from the Lower Aquifer System used to take place, it is now water from the upper aquifers that is entering the lower layers, and contaminating them. Another connection between the upper and lower aquifers happens through wells that are not properly isolated (Figure 9.8 and 9.9).

Regarding the upper aquifers, which have been initially the main origin of water, levels have been falling until the end-1980s or mid-1990s (Figure 9.6). However, they have been progressively abandoned, which made the water level rise in the central part of Campo de Dalías to a point that flooding from groundwater discharge has been one of the major concerns over the last few years. This phenomenon is reflected in the 'La Balsa del Sapo', an artificial wetland generated by excavations to obtain material for the *enarenado* technique, that has filled up with water, when the level in the Central Upper Aquifer started rising (Figure 9.8 and 9.9) (Pulido-Bosch et al., 2000; La Calle & Martínez-Rodríguez, 2013). Adding to the end of pumping in the superficial aquifer, a possible cause of this phenomenon is the infiltration of irrigation return flows and rain water through injection wells, which may explain why the level is currently higher than under natural conditions. The wetland has been recognized as having a high ecological value and water management in the area now has to integrate this new situation. Even if the option of using the water from the Central Upper Aquifer (after treatment) has been proposed, there are contradicting views on the viability of the project. Faced with the emergency generated by the flooding of greenhouses and residential buildings, the decision has been made to evacuate water directly to the sea through a pipe.

Furthermore, quality of the upper aquifers has been affected by contamination from agricultural activities, mainly nitrates from fertilizers and pest control products, as well as wastewater from urban and industrial use (Agencia Andaluza del Agua, 2010). All these factors mean that the 'Campo de Dalías-Sierra de Gádor' groundwater body was designated as in 'poor status', for both quantitative and qualitative reasons, under the WFD (Agencia Andaluza del Agua, 2010). Measures have been proposed to remediate this situation (see Section 4.3 of this chapter) and the time horizon that has been set to reach 'good status' for the groundwater body is 2027.

The gap between withdrawals and available resources and the evidence of falling groundwater levels led to the official 'Declaration of overexploitation' of the aquifer in 1995 (after a provisional declaration in 1986), with the objective of reducing the groundwater withdrawals to 50 hm^3. It implies the establishment of a withdrawals regulation plan that has so far not been set up.

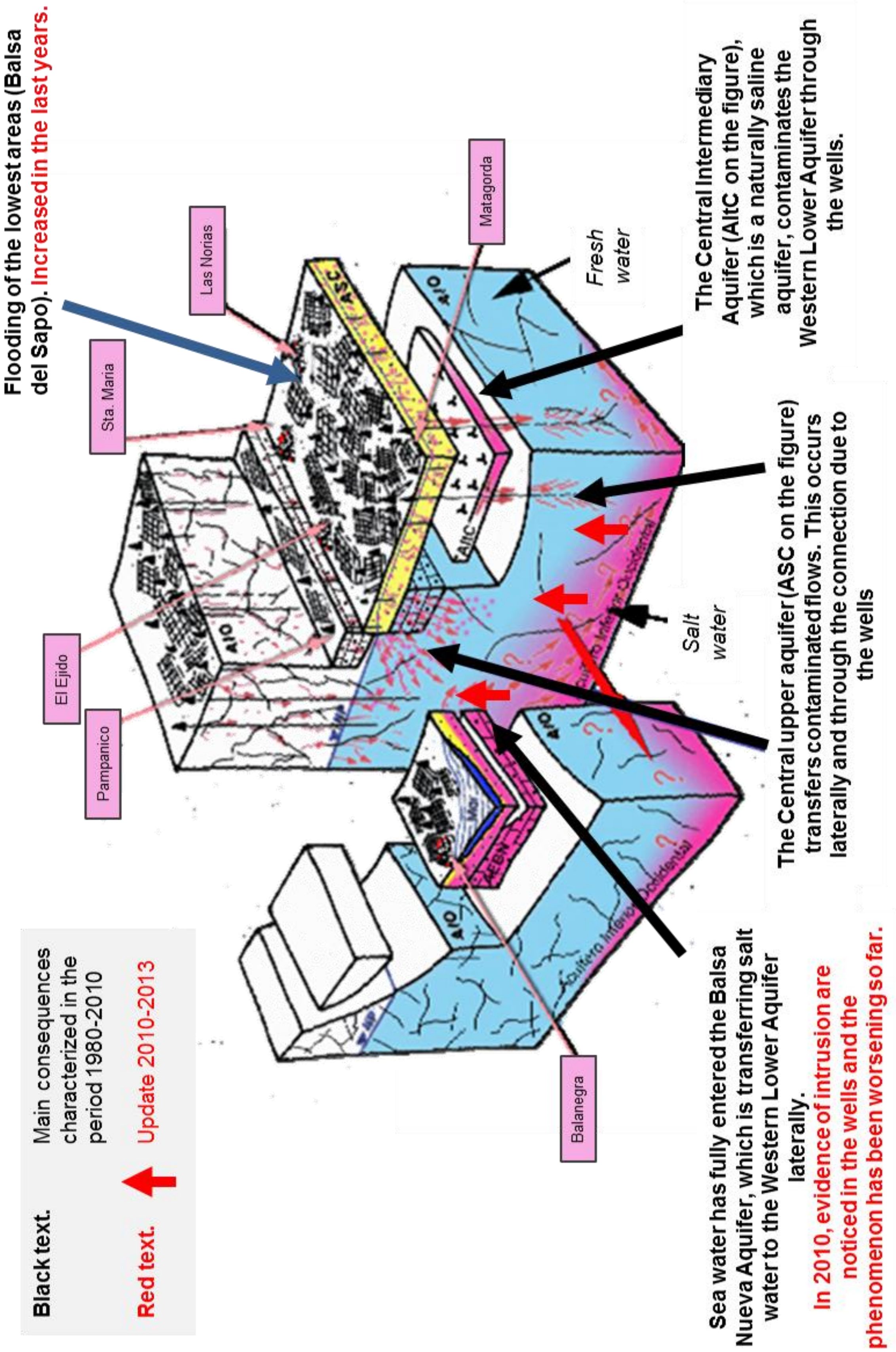

Figure 9.8. Saltwater intrusion on the Western part of the Campo de Dalías Aquifer. Source: adapted from Domínguez-Prats et al. (2012) and González-Asensio et al., (2003)

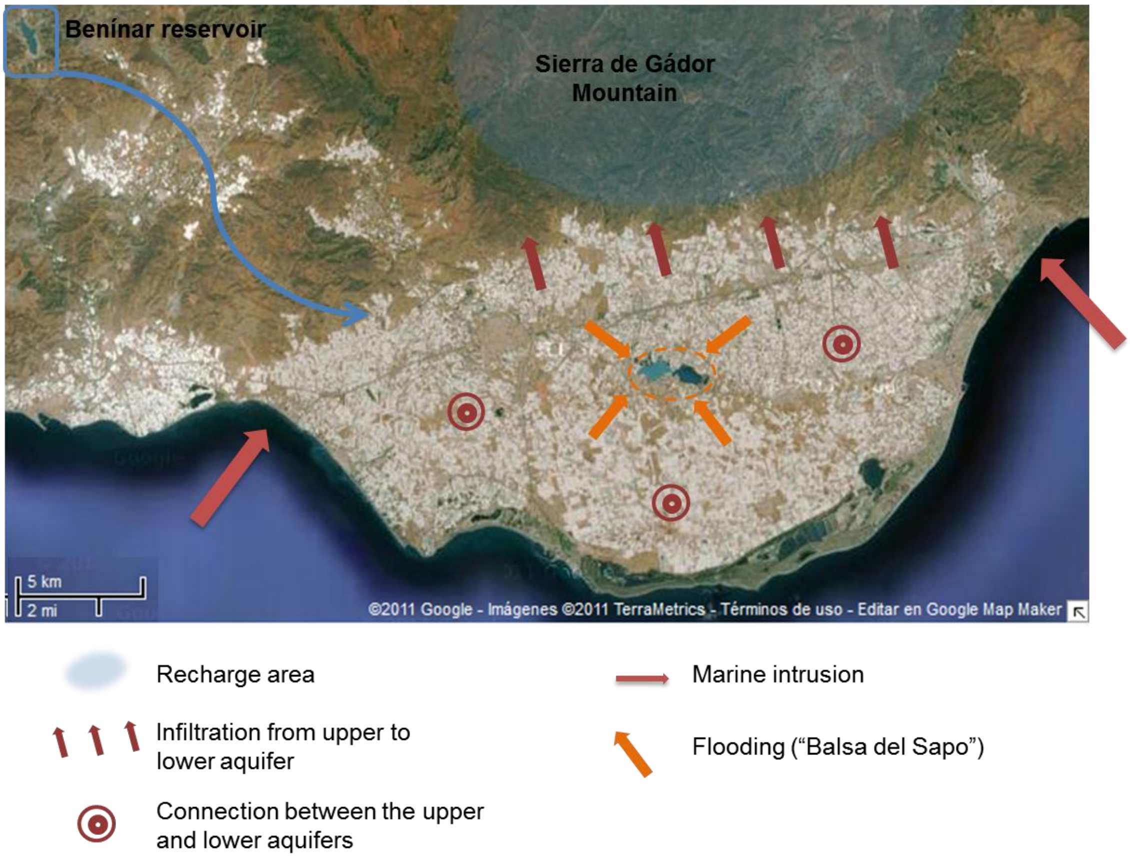

Figure 9.9. Illustration of the issues linked to water management in the Campo de Dalías. Source: own elaboration based on Dumont et al. (2011b)

IX.2 Specific methodological considerations

IX.2.1 Irrigated area by crop and the problem of cropping cycles

As presented in Chapter 6 relative to the methodology of the case studies and data, the irrigated area by crop is obtained at the municipal scale from the *Oficina Comarcal Agraria* of La Mojonera (OCA La Mojonera, 2012). The considered crops are cucumber, melon, watermelon, eggplant, zucchini, pepper, tomatoes and green beans (Plate 9.1). Altogether these crops represent 99% of the greenhouse area in Campo de Dalías (OCA La Mojonera, 2012).

A spatial analysis based on the Irrigation Areas Inventory for the year 2008 (Junta de Andalucía, 2010a) shows that the Campo de Dalías groundwater body comprises the totality of the intensive production in the municipalities of El Ejido, Berja, La Mojonera, Roquetas de Mar, Vicar and Dalías. Since 72% of the greenhouses in the municipality of Berja are within the groundwater body, this percentage is assumed for each crop cultivated in this municipality.

Plate 9.4. Cucumber cultivation in a greenhouse (Las Palmerillas Station). Source: The author

Table 9.1 presents the evolution of the areas for each of the crops for three agricultural campaigns (2009-2010, 2010-2011 and 2011-2012) and the detail by municipality for the 2011-2012 campaign, with the distinction between ‘first or main occupation’ and ‘later occupation’. This refers to the fact that intensive greenhouse production allows to grow two different crops in the same year, an autumn-winter cycle and a spring cycle.

For the reference campaign (2011-2012), the main occupation area (i.e. the total greenhouse area) reaches 16,900 ha, while the later occupation only reaches 7,800 ha. It does not mean that less than half of the farmers produce a second crop in the campaign; while this might be true for some farmers, this is more linked to the fact that data on later occupation does not integrate the area that is cultivated with the same crop for the two cycles or crops that are cultivated on a long cycle[84]. While some crops are necessarily grown in the later occupation (specifically melon and watermelon), there are a variety of options relative to other crops. Short or long cycles can be modulated, e.g. according to the planting date, which means there is a wide diversity of water consumption and yield for the same crop (for instance, a longer cycle allows producing more, which implies more water consumption and prevents planting a second cycle). Furthermore, some crops, such as tomato and pepper, present a range of species that are cultivated, with their own specificities. There are also differences according to the cropping technique (natural or artificial soil).

[84] Some crops, e.g. tomato or pepper, can be grown both on a short cycle to be combined with a second cycle and on a large cycle.

Table 9.1. Irrigated area by crop for the irrigation campaign 2010-2011 and 2011-2012. Source: adapted from OCA La Mojonera (2012)

	2010-2011 (in ha) [a]			2011-2012 (in ha)																
				Berja 'All'		Berja 'In' [b]		Dalías		El Ejido		La Mojonera		Roquetas de Mar		Vicar		Total		
	Prin.	Post.	**Total** [c]	Prin.	Post.	Prin.	Post.	Prin.	Post.	Prin.	Post.	Prin.	Post.	Prin.	Post.	Prin.	Post.	Prin.	Post.	**Total** [c]
Watermelon	0	2,347	**2,347**		389		280		89		1,536		343		189		126		2,563	**2,563**
Melon	0	2,700	**2,700**		145		104		16		1,668		257		214		107		2,366	**2,366**
Zucchini	3,267	773	**4,040**	138	48	99	35	46	16	2,312	396	335	225	242	92	277	139	3,311	903	**4,214**
Cucumber	3,316	530	**3,846**	131	30	94	22	19	11	2,187	370	256	64	279	73	314	107	3,149	647	**3,796**
Pepper	5,841	476	**6,317**	513	64	369	46	305	60	3,758	334	310	67	574	77	354	74	5,670	658	**6,328**
Eggplant	1,372	200	**1,572**	67		48	0	15		543	110	150	53	156	21	497	125	1,409	309	**1,718**
Tomato	2,636	166	**2,801**	187	25	135	18	18	5	1,595	119	143	41	423	50	297	42	2,611	275	**2,886**
Green bean	450	14	**464**	39	9	28	6	9	2	609	72	12	3	29	5	35	8	722	96	**819**
Total	16,882	7,205	**24,087**	1,075	710	774	511	412	199	11,004	4,605	1,206	1,053	1,703	721	1,774	728	16,873	7,817	**24,690**

Prin.: Main or first occupation / Post.: Later occupation.

[a] The figures are the same for the previous campaign (2009-2010).

[b] Crop areas of the municipality of Berja that are assumed for the study area (72% of the figures reported in Berja 'All').

[c] Includes the main or first occupation and later occupation.

In this context, some simplifications have to be made, in order to assess the WF and value of the production. Only the most common combinations of cropping cycles are included. An additional difficulty is that it is not always clear if the figures presented in the different data sets (e.g. yields or production costs) correspond to a long cycle, a short cycle, or to the whole campaign.

On the basis of crops production periods and main crop rotations (Junta de Andalucía, 2013; Fernández et al., 2007) (Figure 9.10), it is possible to make an assumption on the assignation of the greenhouse areas that have not been associated with a later occupation in the data set. That is the gap of 9,000 ha between total greenhouse area and the area reported in later occupation (see Table 9.1).

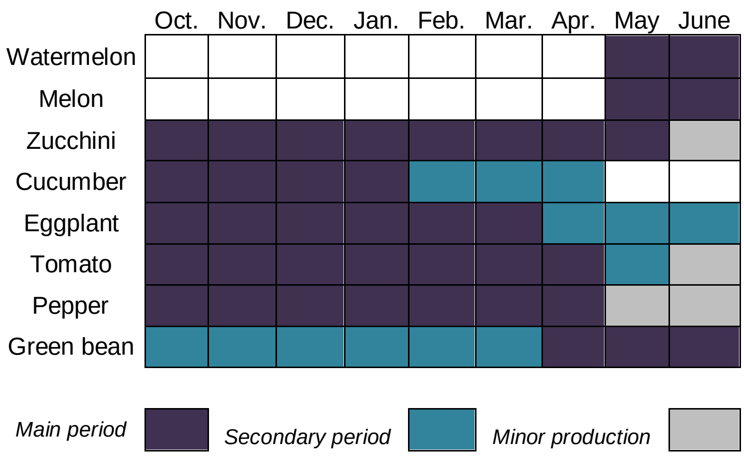

Figure 9.10. Crop production periods. Source: based on Junta de Andalucía (2013)

First of all, it is assumed that eggplants and tomatoes are cultivated on a long cycle, i.e. the same area (amounting to 4,750 ha) can be reported in later occupation. Part of the pepper production is also grown on a unique ‘long cycle’, assumed to be 2,800 ha, based on the data on monthly production presented by Junta de Andalucía (2013). In addition, green beans are usually cultivated on two cycles, which means that the area reported for the first cultivation is assumed to be grown again in a second cycle (720 ha). It is also partly the case for zucchinis; however, there is no data to inform the share of zucchini area that is grown on two cycles, and we will assume that this area corresponds to half of the zucchini first occupation area. The corrected crop areas, based on the previous assumptions, are presented in Table 9.2.

It can be observed that the sum of the areas that are now considered for the second period of the campaign equals the total greenhouse area based on the data for the first occupation (16,872 ha).

Table 9.2. Corrected crop areas (ha) to integrate the full occupation during the second cycle. Source: adapted from OCA La Mojonera (2012).

Crop	Principal[a] A	Posterior[a] B	Total[a] A+B	Additional 'Long cycle'[b] C	Additional '2nd cycle'[b] D	Total '2nd cycle' B+C+D	Total corrected A+B+C+D
Cucumber	3,149	647	3,796			647	3,796
Pepper	5,670	658	6,328	2,800		3,458	9,128
Watermelon		2,563	2,563			2,563	2,563
Melon		2,366	2,366			2,366	2,366
Eggplant	1,409	309	1,718	1,409		1,718	3,127
Tomato	2,611	275	2,886	2,611		2,886	5,497
Green bean	722	96	819		722	818	1,540
Zucchini	3,311	903	4,214		756	1,659	4,970
No crop						756	756
Total	16,872	7,817	24,690	6,820	1,478	16,872	33,743

[a] Source: OCA La Mojonera (2012) / [b] own assumptions (details in text)

IX.2.2 Methodological issues regarding the water footprint quantification

- Implications from the case study general setting on water accounting

Due to the specific geographical and hydrological setting of the case study area, which is located on the coast, with no surface water bodies able to receive return flows, all the water that is withdrawn can be assumed to be entirely consumed. Indeed, the shallow water aquifer system is no longer used; this means that water infiltrating in the ground is lost and the whole amount of water resource withdrawals can therefore be accounted as a WF[85]. Moreover, return flows contribute to the flooding of various areas, as described earlier, which means that in addition to the loss for the originating body, return flows have other detrimental impacts, which cannot be integrated directly within the WF indicator.

- Additional water footprint out of the growing period

Water consumption does not only take place during crop growth; additional consumption associated to other activities amount to 12,2 hm^3 (Table 9.3; Thompson et al., 2009). The main component of this additional water footprint (WF_{add}) is the disinfection of the soil undertaken each year, usually during the summer. Thompson et al. (2009) introduce also a factor to take into account the water that is applied before transplanting the crop (4,1 hm^3). We assume that this additional WF is shared between the different crops, according to their relative importance in the total WF calculated on the basis of water application only during the cropping period.

[85] Since, the area is almost entirely composed of greenhouses, crops do not receive water directly from rainfall and there is no green WF. Nevertheless rainwater is collected and sometimes used to irrigate.

Table 9.3. The water footprint out of the cropping period. Source: Thompson et al. (2009).

Operation	WF (hm^3)
Disinfection	5.6
Pre-transplants	4.1
Other[a]	2.5
Total	12.2

[a] For instance: development of the imported soil technique (*enarenado*) or greenhouse washing.

IX.2.3 Exports of virtual water and their value

The production of the Campo de Dalías area is mainly exported to EU countries. Taking into account this aspect allows presenting a more general picture of the main driver for agricultural productivity and, groundwater use. In an approach of the WF under the perspective of consumers, this allows also relating the impacts on water resources to the final consumer.

Data on the production exported by country and by crop relative to the whole Almeria province is obtained from Datacomex (2013). It has been readjusted taking into account the percentage of production coming from the Campo de Dalías for each crop. Export in virtual water is then obtained by multiplying the exported quantity of each crop by its virtual water content. The value of exports is calculated multiplying the exported quantity by the crop market value.

IX.2.4 The Integral Water Footprint: computing food losses in the water footprint

As proposed in Chapter 5 on the discussion of the WF approach, the actual impacts on water resources of the consumption of a product should take into account losses in the supply chain. Many reasons can explain that products are not consumed, with losses mainly during transport, storage, retailing, and preparation, and due to expiry of products. A product that is finally not consumed has the same environmental impact than the one reaching its final use, which adds to the overall environmental impact of the supply chain.

The case of Campo de Dalías Aquifer is convenient to show these aspects since the products (vegetables) are mostly not processed. It implies that no other step in the supply chain adds a significant WF and that no further calculation is necessary to attribute the WF to processed products (e.g. the transfer from olives to olive oil, see Salmoral et al., 2011a)[86].

[86] In fact, many stages of the supply chain imply the direct or indirect use of water, such as cooking or the indirect WF of transport and storage. However, these are irrelevant compared to the WF in the field and can be excluded from this study. Even when more intense processing is required, the WF added is small as compared to the WF of production in the field.

The percentage of losses for vegetables in Europe has been obtained from FAO (2011). It amounts to 42 %[87].

IX.3 Results

IX.3.1 The water footprint of agriculture

The total WF of the different crops in the study area is obtained by multiplying the water application values by the irrigated areas (Table 9.4). Including the WF_{add}, the total WF associated to the agricultural sector amounts to 103 hm^3. The distribution by crop (Table 9.5 and Figure 9.11) shows that pepper production has the largest WF (27 hm^3), followed by tomato (24 hm^3) and zucchini (18 hm^3). The other five crops represent together less than one third of the WF. The 'additional WF' constitutes 11.6% of the total WF.

Table 9.4. The water footprint of the Campo de Dalías case study by crop.

	Crop	Area main cycle (ha)	Area 2^{nd} cycle (ha)	Applied water (m^3/ha)	WF (hm^3)	WF Corrected[a] (hm^3)
2 cycles	Cucumber	3,149	647	2,700	10.2	11.6
long cycle	Pepper	5,670	3,458	3,110	23.0	26.1
2^{nd} cycle	Watermelon		2,563	1,890	4.8	5.5
2^{nd} cycle	Melon		2,366	1,770	4.2	4.8
long cycle	Eggplant	1,409	1,718	4,300	9.8	11.1
long cycle	Tomato	2,611	2,886	5,000	20.3	23.0
1^{st} cycle	Green bean 1	722		1,580	1.1	1.3
2^{nd} cycle	Green bean 2		818	1,970	1.6	1.8
1^{st} cycle	Zucchini 1	3,311		2,500	8.3	9.4
2^{nd} cycle	Zucchini 2		1,659	4,400	7.3	8.3
	Total	16,872	16,115		90.7	102.9

[a] Including the WF out of the cropping period (see Table 9.4)

The value of the total WF is higher than the figure presented by Thompson et al. (2009) (88.3 hm^3) and lower than the value of water withdrawals presented in the Irrigation Areas Inventory of Andalusia (Junta de Andalucía, 2010a) (120 hm^3). It is also the case for the greenhouse area,

87 Waste percentages of the different steps are: 20 % for agricultural production, 5 % for postharvest handling and storage, 2 % for processing and packaging, 10 % for supermarket and retail and 19 % for consumption. This is a first estimate, relative to Europe including Russia. It serves mainly to illustrate the concept of an 'Integral WF' developed in this thesis and should be refined. On the one hand, the supply chain in the Campo de Dalías may be more efficient as it is well organized, for instance with a cluster organization and transport infrastructures. On the other hand, fresh vegetables are more fragile and expire more rapidly, compared to processed vegetables, which are also included in these estimates.

since the total irrigated area presented in this inventory reaches 18,480 ha. Finally, the WF per ha is 5.5% higher, 6,590 m^3/ha compared to 6,245 m^3/ha.

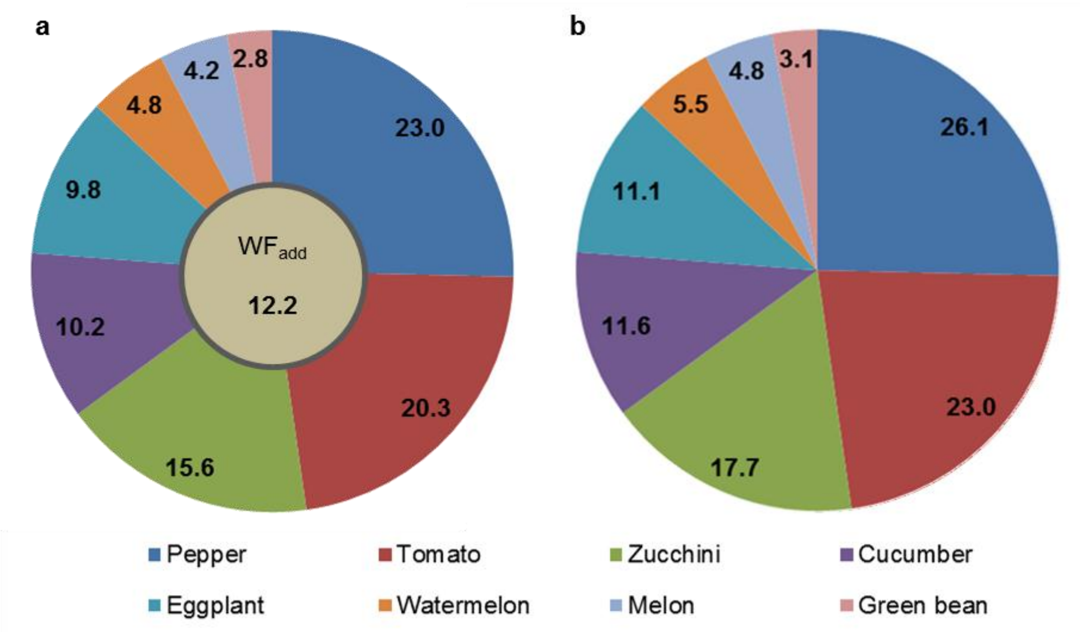

Figure 9.11. The agricultural water footprint of Campo de Dalías by crop (hm^3), with (a) and without (b) distinguishing the water footprint out of the growing period.

The WF of the different crops in L/kg ranges from 29.8 L/kg for cucumbers to 91.7 L/kg for green beans (first cycle), which present both the lowest applied water and the lowest yield (Table 9.5).

Table 9.5. Irrigation water application and water footprint by crop.

	Applied water (m^3/ha)	Yield (t/ha)	WF (L/kg)	WF_{corr} [c] (L/kg)	Integral WF (L/kg)	Weight (kg)	WF (L/item)	WF_{corr} [c] (L/item)	Integral WF[d] (L/item)
Cucumber	2,700	90.5	29.8	34.6	59.7	0.2	5.9	6.9	11.9
Pepper	3,110	69.4	44.8	52.0	89.7	0.22	9.9	11.5	19.8
Watermelon	1,890	61.8	30.6	35.5	61.2	3.5	107.0	124.3	214.3
Melon	1,770	36.1	49.0	56.9	98.1	1.5	73.5	85.4	147.2
Eggplant	4,300	99.2	43.4	50.4	86.9	0.25	10.8	12.6	21.7
Tomato	5,000	95.6	52.2	60.6	104.5	0.13	6.8	7.8	13.4
Green bean 1[a]	1,580	20.0	79.0	91.7	158.1	-	-	-	-
Green bean 2[b]	1,970	25.0	78.8	91.5	157.8	-	-	-	-
Zucchini 1[a]	2,500	50.0	50.0	58.1	100.2	0.25	12.5	14.5	25.0
Zucchini 2[b]	4,400	95.0	46.3	53.8	92.8	0.25	11.6	13.5	23.3

[a] First cycle / [b] Second cycle / [c] Corrected WF including WF_{add} / [d] Based on WF_{corr}

The figures presented by item range from 6.9 L to 14.5 L for vegetables and amount to 85 L and 124 L for melon and watermelon, respectively. The 'Integral Water Footprint' of each crop is 72 % higher than these figures (based on a loss ratio of 42 %, see Section 2.4 of this chapter). It ranges from 12 L for a cucumber to 214 L for a watermelon. As noticed in the methodology section of this chapter, this is a first estimate based on a general value of losses in the supply chain presented for fruits and vegetables for the whole of Europe.

The figures of the virtual water content are mainly presented here for illustration, to raise awareness on the quantity of water that is necessary to obtain a vegetable or a fruit. However, the 'Integral WF' allows showing that a significant reduction in terms of groundwater consumption could be achieved with a decrease in losses and waste in the supply chain.

IX.3.2 Economic value and employment linked to the agriculture water footprint

- Total economic value and production costs

The total economic value, based on the market price of production, reaches 1081 million euros; however, it is reduced to 333 million euros, when production costs are included. The production costs considered include direct costs (e.g. seeds, water, energy, fertilizers, labour), indirect costs (e.g. greenhouses and irrigation installation) and general and financial costs (Table 9.6). A view per crop reveals that pepper is the crop that generates more value in the area, followed by tomato, zucchini and cucumber (Table 9.7).

However, it should be remembered that these values constitute a first overview, due to the uncertainty inherent in the methods that have been used and the diversity of the production systems. This is especially true for labour costs and, more particularly, for family labour costs, which are based on the declaration of farmers, e.g. for labour taxes (Junta de Andalucía, 2013).

The discrepency in the WF value, compared to the results of the Irrigation Areas Inventory (Junta de Andalucía, 2010a), is not mirrored for the economic value of production. In terms of both market value of production and farmers' profits, the values are similar, around 900-1000 million euros for the former and 350 million euros for the latter (Table 9.7).

Table 9.6. Market value, costs, profits and water and land productivities for the different crop cycles and crop rotations. Source: various and own elaboration

	WF[a] (m^3/ha)	Market prices[b] (€/kg)		LP (€/ha)	WP (€/m^3)	Costs[d] (€/ha)				LP (€/ha)	WP (€/m^3)	Family labour[d]		LP (€/ha)	WP (€/m^3)
		2003/12[c]	2011/12	*Based on market price*		Direct	Total	Water[e]	Labour[e]	*Based on profits*[f]		Days	Cost (€/ha)	*Based on profits*[g]	
Crop cycle															
Cucumber	2,700	0.44	0.41	39,820	12.7	23,000	27,900	800	9,300	11,920	3.8	274	4,300	7,620	2.4
Pepper	3,110	0.69	0.63	47,886	13.3	23,900	30,200	700	9,200	17,686	4.9	254	3,400	14,286	4.0
Watermelon	1,890	0.29	0.26	17,922	8.2	12,000	15,300	700	1,200	2,622	1.2	53	1,400	1,222	0.6
Melon	1,770	0.39	0.38	14,079	6.8	10,200	13,000	300	2,900	1,079	0.5	75	900	179	0.1
Eggplant	4,300	0.46	0.39	45,632	9.1	29,150	36,650	1400	14,900	8,982	1.8	364	3,300	5,682	1.1
Tomato	5,000	0.49	0.51	48,756	8.4	32,200	40,100	1900	12,500	8,656	1.5	425	8,600	56	0.01
Green bean 1	1,580	1.24	1.33	24,800	13.5	11,300	15,100	400	6,300	9,700	5.3	189	3,100	6,600	3.6
Green bean 2	1,970	1.24	1.33	31,000	13.5	12,500	18,300	600	6,700	12,700	5.6	246	5,400	7,300	3.2
Zucchini 1	2,500	0.47	0.44	23,500	8.1	9,900	13,400	500	4,700	10,100	3.5	198	5,100	5,000	1.7
Zucchini 2	4,400	0.47	0.44	44,650	8.7	14,900	19,600	900	6,500	25,050	4.9	252	6,100	18,950	3.7
Crop Rotations															
Cucumber/(Water)Melon	4,530			55,821	10.6	34,100	42,050	1,300	11,350	13,771	2.6	338	5,450	8,321	1.6
Pepper/(Water)Melon	4,940			63,887	11.1	35,000	44,350	1,200	11,250	19,537	3.4	318	4,550	14,987	2.6
Pepper	3,110			47,886	13.3	23,900	30,200	700	9,200	17,686	4.9	254	3,400	14,286	4.0
Eggplant	4,300			45,632	18.3	29,150	36,650	1,400	14,900	8,982	3.6	364	3,300	5,682	2.3
Tomato	5,000			48,756	16.8	32,200	40,100	1,900	12,500	8,656	3.0	425	8,600	56	0.02
Green bean	3,550			55,800	13.5	23,800	33,400	1,000	13,000	22,400	5.4	435	8,500	13,900	3.4
Zucchini	6,900			68,150	8.5	24,800	33,000	1,400	11,200	35,150	4.4	450	11,200	23,950	3.0
Average	**5,377[a] / 6,245[h]**			**64,118**	**10.3**	**35,093**	**44,359**	**1,515**	**13,892**	**19,760**	**3.2**	**427**	**7,308**	**12,452**	**2.0**

[a] Without WF_{add} – [b] Source: Cabrera and Uclés (2012) – [c] Average market price 2003-2012 – [d] Source: Junta de Andalucía (2013) – [e] Included in direct costs – [f] Without family labour – [g] Including family labour – [h] Including WF_{add}

Table 9.7. The agricultural water footprint of Campo de Dalías and its value per crop.

	$WF_{irr.}$ (hm^3)	$WF_{corr.}$ (hm^3)	Value[a] (10^6 €)	Value[b] (10^6 €)		Value[c] (10^6 €)	
Cucumber	10.2	11.6	151.2	45.2	*29.9 %*	2.8	*1.9 %*
Pepper	23.0	26.1	354.3	130.9	*36.9 %*	4.6	*1.3 %*
Watermelon	4.8	5.5	45.9	6.7	*14.6 %*	0.6	*1.4 %*
Melon	4.2	4.8	33.3	2.6	*7.7 %*	0.1	*0.3 %*
Eggplant	9.8	11.1	103.5	20.4	*19.7 %*	2.6	*2.6 %*
Tomato	20.3	23.0	197.7	35.1	*17.8 %*	0.02	*0.0 %*
Green bean	2.8	3.1	43.3	17.4	*40.2 %*	7.9	*18.2 %*
Zucchini	15.6	17.7	151.9	75.0	*49.4 %*	6.3	*4.2 %*
Total	90.7	102.9	1.081.0	333.2	*30.8 %*	25.0	*2.3 %*
Irrigation areas inventory[d]		120.9	905.4	354.3			

[a] based on market price / [b] profits without family labour / [c] profits including family labour / [d] Junta de Andalucía, (2010a) / Percentages are relative to the value based on market prices.

- <u>Direct economic value by crop combination</u>

In Figure 9.12, the Direct Economic Value is shown for the main crop combinations.

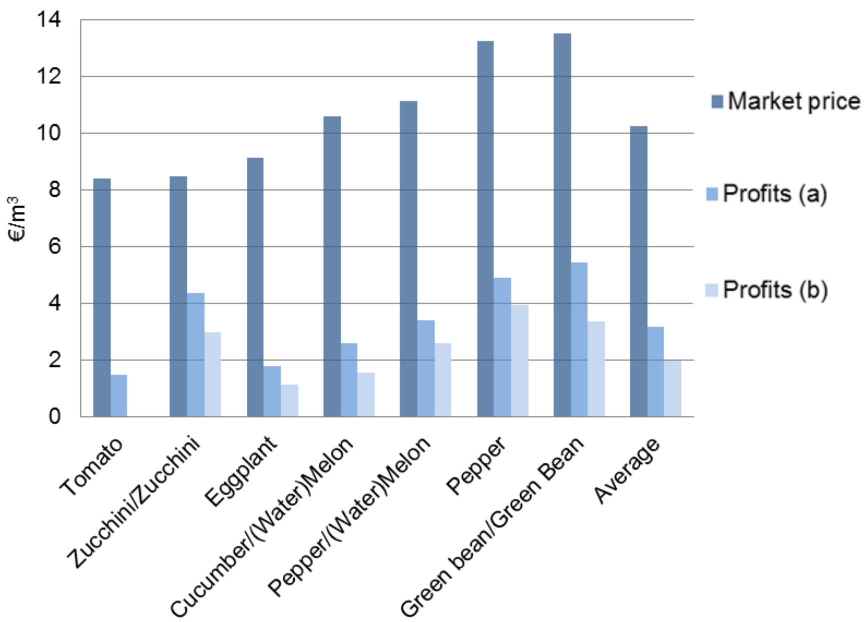

Figure 9.12. Direct Economic Value for the main crop combinations (whole campaign). The Direct Economic Value is based on production market prices ('Market price') and farmer profits with (b) and without (a) family labour. The water factor includes the WF out of the growing period (WF_{add}). Only the main long cycles and combinations of short cycles are presented, as explained in Section 2.

Green beans and peppers are the options that present the highest values, which range from 8.4 to 13.5 €/m^3, the average being 10.3 €/m^3 (Figure 9.12). When production costs are included, the Direct Economic Value ranges from 2.6 to 5.4 €/m^3, with an average value of 3.7 €/m^3.

- Land productivity

The highest land productivity (based on market prices) is presented for the most intensive crop combinations, i.e. crops that are grown on two cycles: zucchini, green beans, and cucumber or pepper, both followed by melon or water melon (Figure 9.13). More or less the same pattern is reproduced based on farmers profit, with tomato and eggplant most affected by higher production costs relative to their market value.

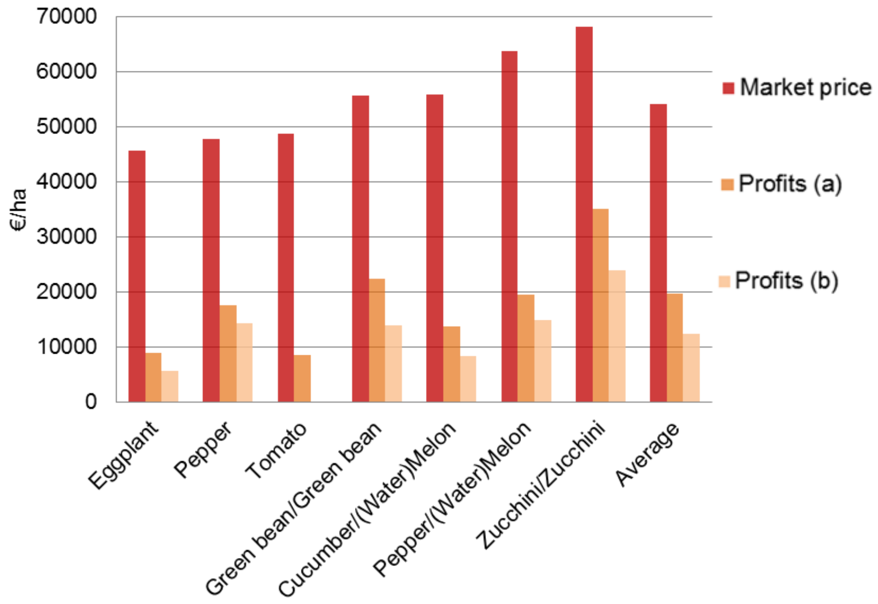

Figure 9.13. Land productivity (€/ha) for the main crop combinations (whole campaign).
Land productivity is based on market prices for production ('Market price') and the farmer profits without ('Profits a') and including ('Profits b') family labour.

- Direct Employment

An estimation of the direct employment generated in the area can be obtained on the basis of the labour cost and family working hours. As an average, it has been estimated that around 430 family working days per ha are necessary for crop production (Table 9.6). It corresponds to a cost of 7,300 €/ha. Assuming the same ratio between working days and labour cost for hired labour, this would imply that hired labour represents 820 working days per ha. It means that in total 3.4 workers per ha are employed full time, i.e. around 57,900 persons working directly in the greenhouses. However, this is a first estimation, since, for instance, many family working hours might not be declared. Moreover, indirect jobs should also be taken into account, to have a full picture on the impacts of agricultural activity on employment in the area.

In addition to the direct generation of jobs, a characteristic of Campo de Dalías agriculture is the high number of farmers. According to Junta de Andalucía (2010a), there are 11,200 farmers in the area, which implies an average farm size of 1.5 ha (considering a total greenhouse area of 16,900 ha). This is a small area compared to other agricultural systems and this means that the benefits of the agricultural sector are shared among many people and do not only accrue to some big companies or farmers.

IX.3.3 The water footprint of urban and tourism uses

All the municipalities in Campo de Dalías are supplied by groundwater (Table 9.8), whether through their own wells or through participation in irrigation communities, from which they usually receive water in the summer, when demand peaks because of tourism[88]. Furthermore, the city of Almeria also uses water from the Campo de Dalías aquifer, even when the construction of the desalination plants allowed reducing the supply from the aquifer. In 2010, the desalination plant supplied only 5 hm^3 from a total annual capacity of 18 hm^3 (Martínez-Rodríguez, 2011).

Table 9.8. The water footprint of urban water supply in Campo de Dalías. Source: adapted from Martínez-Rodríguez (2011).

Municipality	Water footprint (hm^3)
Berja	1.8
El Ejido	8.7
La Mojonera	1.7
Roquetas de Mar	12.4
Vicar	1.8
Almeria	10.8 (groundwater)
Almeria	5.2 (desalination)
Total (without desalination)	37.2

The total WF of urban water supply is 37 hm^3. It corresponds to the consumption of the year-round residents plus tourism. In addition to domestic water use, urban networks supply other activities, such as services, tourism or recreational use. An additional WF of 1.5 hm^3 is associated more specifically to the golf courses of the area (Cuitó-Sabaté et al., 2006).

IX.3.4 Synthesis: origin of water and water footprint

The total WF of the case study domain, including the whole WF for urban water supply of Almeria, reaches 147 hm^3, with agriculture having the largest share (70%). Including the surface water supply from the Beninar reservoir (10 hm^3 as an average) and the desalinated water supply for Almeria (5.2 hm^3), it results in a WF for groundwater of 132 hm^3 (Figure 9.14). This value is

[88] This is a period during which greenhouses are not cultivated. However, there is a demand from the agricultural sector for the disinfection of greenhouses that usually takes place in this period.

similar to the one from the Water Plan, where two values, 130 hm^3 and 150 hm^3, can be found in different sections of the document (Agencia Andaluza del Agua, 2010).

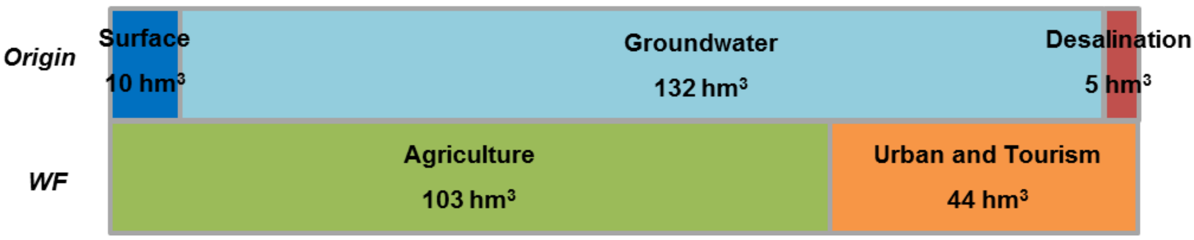

Figure 9.14. Origin of water use and water footprint of the different sectors.
Desalination for the city of Almeria (5 hm^3) has been included because it constitutes a substitutable resource to groundwater for the supply of the city.

IX.3.5 Exports of virtual water and their value

The total virtual water exports from the Campo de Dalías Aqufier amounts to 66 hm^3 for the campaign 2011/2012, i.e. 64 % of the WF (Figure 9.15). This corresponds to a value for the exported production of 700 million euros. The main virtual water flows destination is Germany. Five countries (Germany, France, Netherlands, United Kingdom and Italy) receive more than three quarters of the virtual water flows exports.

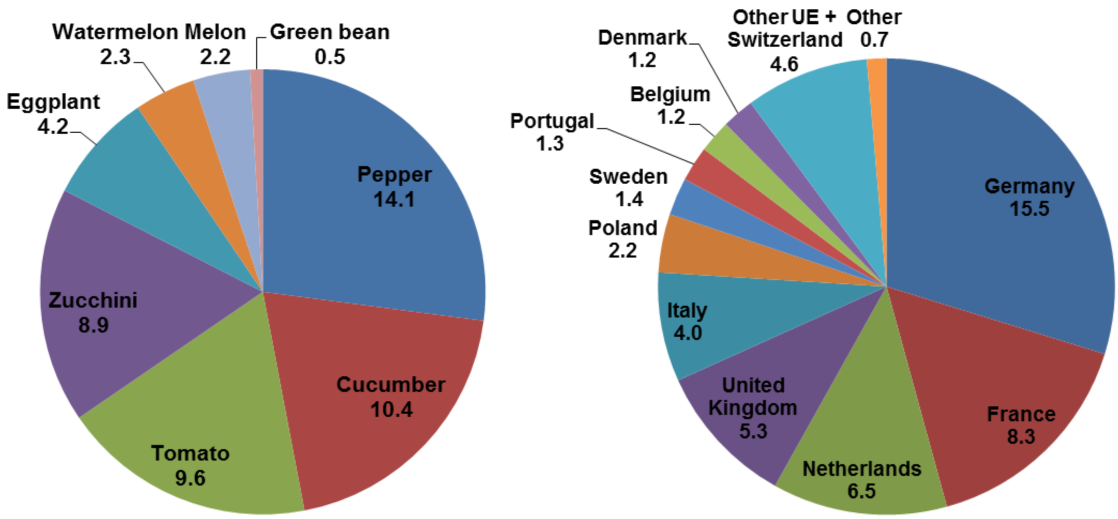

Figure 9.15. Exports of virtual water by crop and by country of destination (hm^3).

IX.4 Discussion

IX.4.1 Impacts of groundwater use: evidence and uncertainties on seawater intrusion

As the value of recharge is uncertain, it may be problematic to use it as a reference to assess the consequences of groundwater use, even though the value of withdrawals is clearly above recharge estimates: 50 hm^3 according to Pulido-Bosch et al. (1993) or 79 hm^3 according to

ACUAMED (2006). A more reliable indicator is the evolution of groundwater levels, which are below sea level in the two lower aquifers from the year 1980 at least (Figure 9.6). The presence of points of such low piezometric level close to the sea – up to -48 m in the Western Lower Aquifer, connected to the sea through the Escama de Balsa Nueva Aquifer, and water tables below the sea level, in aquifers in direct contact with the sea in the North-eastern area – is evidence for the potential threat for seawater intrusion.

Warnings on the unsustainability of withdrawals, with the risk that intense seawater intrusion could mean an end to groundwater use, and the associated economic benefits, have been heralded since the end of the 1970s. However, groundwater withdrawals have been increasing since then and only some wells near the coast line in the North-eastern Lower Aquifer, where seawater intrusion is most direct, had to stop pumping groundwater. As the promised ‘perfect storm’ did not happen, it led to some discredit of scientific technical expertise. This adds to the inherent scientific uncertainty of hydrogeology as a science, often based on models and the interpretation of often incomplete data.

While it is true that some experts may have ‘cried wolf’ and that reducing withdrawals since the 1980s below the level of inflows could have resulted in important economic losses for the area during the last 30 years, groundwater tables below the sea level are an evidence that cannot be disregarded. The latest trends confirm the diagnosis established then. Figure 9.16A illustrates schematically the evolution of seawater intrusion based on the description made in Section 1.5 of this chapter. It shows that saltwater may have accumulated in the bottom of the aquifer since the beginning of the 1980s.

This illustration could explain why saltwater has only been detected recently in the Western Lower Aquifer. It only constitutes a possible interpretation of the situation. For instance, it has been assumed that the majority of natural outflows happens through the Escama de Balsa Nueva Aquifer and that only a reduced fraction occurs more deeply, which explains why saltwater can be ‘trapped’ for a long time (Figure 9.16B).

The relative isolation of the aquifer from the sea allowed using the fresh water stock in addition to the recharge over the last thirty years. However, the other side of the coin is that the depletion of the stock has induced saltwater intrusion not only now, as described previously, but also potentially into the future, until the water table attains again the sea level. This dynamic is similar to the future capture described in Chapter 2 on groundwater dynamics.

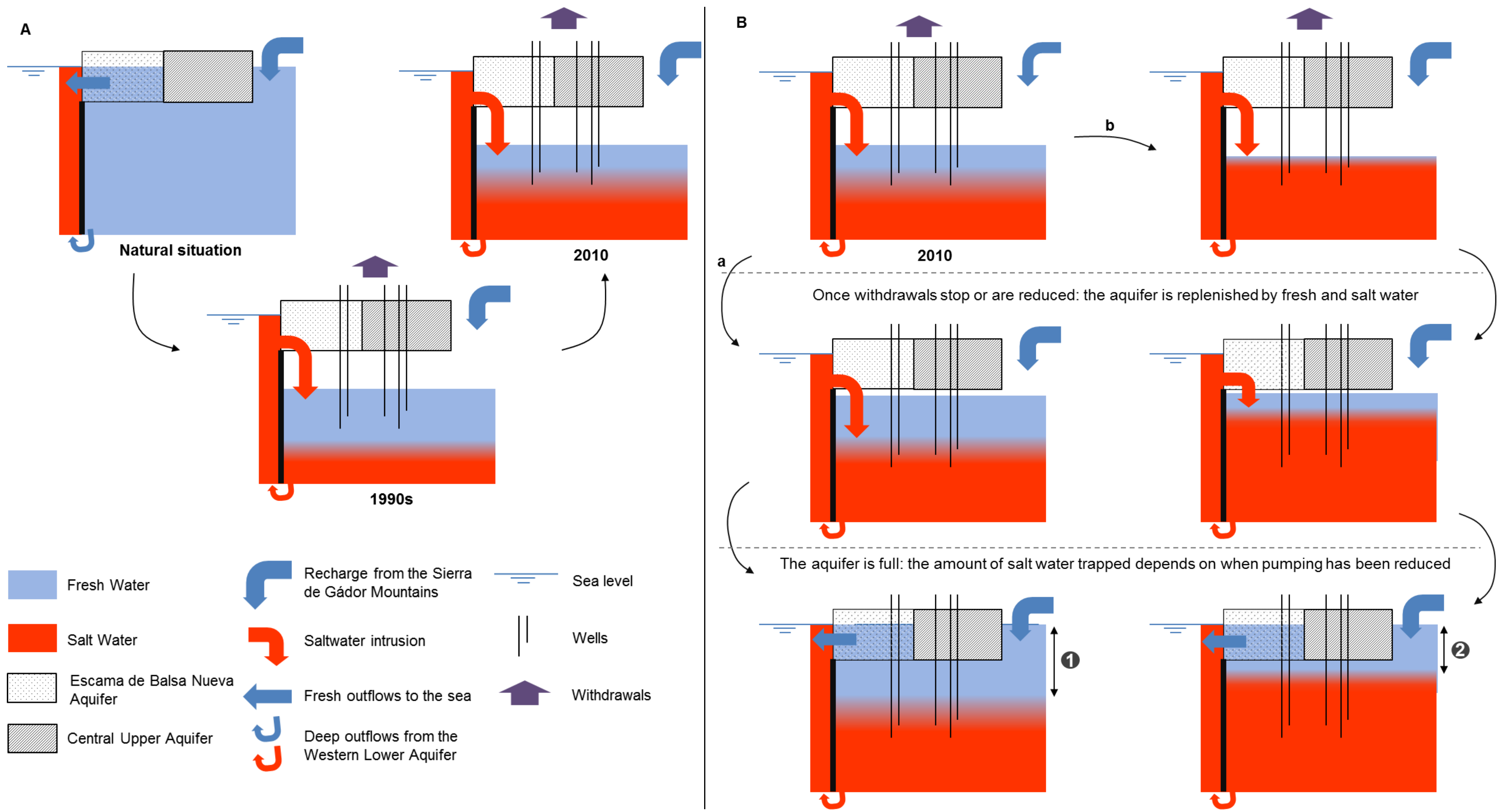

Figure 9.16. A: Illustration of the sea water intrusion to the Western Lower Aquifer from the Escama de Balsa Nueva Aquifer; B: Final saltwater intrusion if withdrawals are reduced in the short or long term.
(a) Pumping is reduced in the short term / (b) Pumping is reduced in the long term / The 'freshwater lens' is thicker in 1 as compared to 2, which increases the probability to use of the aquifer.

Thus, even if withdrawals were currently reduced to match in average the recharge, the stabilization of the level would not be sufficient to stop seawater intrusion and saltwater would continue to accumulate in the aquifer. This might jeopardize the use of the aquifer in the long run. Thus, 'there is no free lunch': past withdrawals above recharge mean withdrawals should be reduced well below recharge now, as the stabilization of levels is not enough.

The final situation potentially changes depending on when withdrawals are reduced and to what extent (Figure 9.16B). This has direct consequences on the possibility to use groundwater resources in the future, since a thin 'freshwater lens' above the trapped saltwater would complicate pumping compared to the situation where it would be thicker. The impacts of pumping are therefore not proportional to their amount. There is potentially a threshold beyond which the possibility of maintaining benefits from fresh water would be limited. The detection of salt water in some wells may indicate that this threshold may be reached in the short term.

IX.4.2 Integrating seawater intrusion in the water footprint of coastal aquifers?

- First option: a grey water footprint for marine intrusion

At first sight, marine intrusion could be considered as a contamination of the aquifer by salts, and the grey WF could be the appropriate indicator to report it. However, it may be meaningless to determine how much water would be necessary to dilute the salinity of the aquifer, since the proper cause of marine intrusion is groundwater pumping (a quantitative cause)[89]. What is generating the problem is not the emission of contaminants but rather groundwater withdrawals.

- Second option: future appropriation of water resources

A direct consequence of salt water intrusion is the impossibility to keep using groundwater because of its salinity. It means that it is no longer possible to use a fresh water flow (the recharge) that was previously available, because of past withdrawals. It is as if past withdrawals would have appropriated future groundwater use. Thus, the WF from groundwater in an aquifer that experiences marine intrusion could be considered as the sum of actual current withdrawals plus the future availability of groundwater that it has contributed to cancel out.

The quantification of this future appropriation is quite challenging as it should contemplate, among other factors, the share of recharge that would effectively be no longer available and the period of this restriction[90]. For a typical coastal aquifer, a strong reduction in withdrawals could

[89] The grey WF quantifies the water that would be necessary to dilute a pollutant to its acceptable concentration (Hoekstra et al., 2011).

[90] Other indicators than a WF may reflect better the future impacts of excessive current withdrawals, such as the cost to provide an alternative source of water once the aquifer is salinized. This replacement cost should ideally be considered when evaluating the profitability of the current situation.

allow re-establishing natural conditions in a few years, as freshwater outflows 'flush' salt water. However, in the case of Campo de Dalías aquifer, saltwater may accumulate in the bottom of the aquifer, where it could be trapped (Figure 9.16), affecting the quality of withdrawals for a long period. For instance, assuming 80% of the recharge is unavailable for a period of 40 years from 2020 (i.e. after 40 years of having surpassed the groundwater availability), the future WF of the current pumping would be 80% of the recharge, that should be added to the current WF.

IX.4.3 Future supply options and associated costs and energy consumption

Faced with the problem of marine intrusion and the issue of securing water supply for the city of Almeria, a first decision was to build a desalination plant for the city in 2003, with an annual capacity of 18 hm^3. In addition, within the framework of the program 'AGUA'[91], which have replaced the National Water Plan (Downward & Taylor, 2007), a desalination plant has been planned in the area of Campo de Dalías, with a capacity of 30 hm^3, potentially extensible to 40 hm^3 (ACUAMED, 2006). The plant should have been put into service in 2010. However, a series of technical upgradings were decided during the construction (García-Arancón et al., 2013), which is one possible explanation for the delay, as delivery is now due in 2015. For both plants, part of the financing was obtained from the EU, with the condition that withdrawals from groundwater should be reduced in proportion to the desalinated water produced.

Regarding the desalinated water allocation, 22.5 hm^3 are destined for urban use and 7.5 hm^3 for agriculture (8,000 ha) (ACUAMED, 2006; ACUAMED, 2013). However, part of this water (around 10%) is intended to be delivered to Adra city and the Vega de Adra irrigation district, which are located outside the case study area. Wastewater reuse in the cities of El Ejido and Roquetas de Mar is also available, which could add an additional 10 hm^3 to the overall water resources. This could be used by the agricultural sector. However, there is an issue of acceptability of this source of water by farmers, who are used to accessing good quality groundwater. This water source is therefore currently not used (Martínez-Rodríguez, 2012). Improvements in the efficiency of urban networks could also allow reducing groundwater withdrawals for urban use. The different options to increase supply are shown in Table 9.9.

In addition to desalination and wastewater reuse, the main measures proposed by the Water Plan to reach the 'good status' of the groundwater body are: elaboration of the pending plan of regulation of uses associated to the declaration of overexploitation, constitution of a unique users community, irrigation modernization[92], revision of the infrastructures to improve

[91] AGUA is the acronym that stands for '*Actuaciones para la Gestión y la Utilización del Agua*'.

[92] The cost of groundwater pumping has been a strong incentive for farmers to switch to drip irrigation from the 1980`s. Few additional savings would be obtained from the technical point of view (infrastructures), even if the management of water by farmers can still be improved occasionally.

regulation, continuing with the program of measures to deal with nitrate contamination in vulnerable areas, improving the wastewater networks and wastewater treatment systems in line with the EU WFD (Agencia Andaluza del Agua, 2010).

Table 9.9. New balance of water resources in Campo de Dalías according to different options of use of the different sources of water (hm^3).

	Current	With full use of the existing capacity	With full use of the existing capacity + New desalination plant
Groundwater	132	109	82
Surface	10	10	10
Desalination	5	18	45 (27 +18)
Reuse	-	10	10

The potential evolution of the WF in the future (urban demand) has not been included.

The energy requirements of the different supply options and an estimation of their cost are presented in Table 9.10.

Table 9.10. Energy requirements and estimation of the costs of the sources of water. Source: various and own elaboration.

	Groundwater	Desalinated water	Wastewater reuse	Total
Energy requirement (kWh/m^3)	0.7 – 1.4[a]	5.2[b]	0.2[c]	
Cost estimation[d] (Energy) (10^{-2} €/m^3)	3.9 – 7.8	29.0	1.1	
Total cost (10^{-2} €/m^3)	15[e] – 28[f]	62 / 124[g]	-	
Energy requirement of the current supply[h]	144	26	-	170
Energy requirement of the future supply[h]	94	234	20	348

[a] Martínez-Rodríguez (2011) / [b] ACUAMED (2013) / [c] Martínez-Rodríguez (2012) and only tertiary treatment (no delivery) / [d] With 0.0056 €/kWh (Lapuente, 2012) / [e] Junta de Andalucía (2010a) / [f] obtained from Junta de Andalucía (2013) / [g] Running at 40% of the capacity (based on Cooley & Ajami, 2012) / [h] Based on the scenarios presented in Table 9.9, without surface water

The values should be taken as a first approximation, as there are potentially great variations according to the actual use of the infrastructure, particularly for desalination. The final value can also vary greatly depending on components that are introduced, e.g. direct energy consumption, pay back, financial cost or distribution network. This is very pertinent when comparing different estimates. A pivotal point is also the lifespan considered for the plant.

According to Lapuente (2012), who undertook a survey of seven desalination plants in the Segura river basin (Spain), the total energy consumption, including delivery, ranges between

3.85 and 5.02 kWh/m^3, with an average value of 4.45 kWh/m^3 and with distribution costs amounting to up to 20%. It resulted in an average energy cost of 0.25 €/ m^3, on the basis of an energy price of 0.0056 €/kWh (Table 9.11). This is in line with the estimates for the Campo de Dalías plant (5.17 kWh/m^3) (ACUAMED, 2013). Since energy represents only 38% of the full cost for desalinated water, a final estimate is 0.66 €/m^3. Again this value is close to the figure presented for the Campo de Dalías plant, 0.62 €/m^3 (ACUAMED, 2006)[93]. Nevertheless, it should be kept in mind that this value is obtained assuming the plant is running full time at full capacity. According to Cooley and Ajami (2012), functioning at 40 % capacity implies doubling the cost of desalinated water. Based on the experience of desalination plants already built in the Almeria Province (e.g. Almeria or Carboneras), it is very probable that the new plant will not work at full capacity, a point that will be discussed later.

With respect to wastewater reuse, energy requirements are limited to tertiary treatment, since secondary treatment is mandatory according to EU legislation. Thus, there are limited costs for the user of this source of water (0.01 €/m^3) (Table 9.10).

As regards groundwater, there is a difference between costs calculated based on the energy consumption and the value reported by other sources. This gap can be explained in part by the cost of the construction of the wells and irrigation installation that may have been included in the data set from Junta de Andalucía (2013). Another reason is the value selected for the cost of energy, which is quite low since it corresponds to the price negotiated by a desalination plant, running for the whole year, according to Lapuente (2012).

It is also noteworthy that the energy requirements of the future supply scheme would double (Table 9.9), which can question its viability. Furthermore, an overview on the implications from the introduction of desalination on the carbon footprint of production is presented in Box 9.1.

IX.4.4 Scenarios for the repartition of water costs between agriculture and cities

For the Campo de Dalías desalination plant, the price that farmers will pay for water is planned to be 0.30 €/m^3 (ACUAMED, 2006), which means that the remaining costs will be supported by public subsidies[94] or through a higher price paid by urban dwellers (i.e. a potential cross subsidy from urban use to agricultural use).

[93] This value corresponds to the first project of the plant, which have been modified later (see García-Arancón et al., 2013). Moreover it corresponds to an ‘engineering estimate’ that tends to consider the ideal conditions (Cooley and Ajami, 2012), which are virtually never found in reality, since many factors such as delay in the construction or partial use of the production capacity can increase the cost.

[94] Subsidies from the EU amount to 20% of the construction cost, i.e. 17.2 million euros. The majority of other public funds are to be paid off by the users progressively during the lifespan of the plant (ACUAMED, 2006). However, it is doubtful if this payment will effectively happen.

Box 9.1: Increase in the carbon footprint of 1 kg of tomatoes linked to the choice of desalination

One of the main concerns for the sustainability of the food supply chains is the carbon footprint linked to energy consumption, among other factors, such as the use of fertilizers. A particular issue is to assess the impacts from different modes of production and to guide the choice of consumers. For instance, Hospido et al. (2012) deals with the issue of seasonality of lettuce production under a Life Cycle Analysis (LCA) approach, comparing the production in the United Kingdom during summer (open-field) and the United Kingdom and Spain in winter (greenhouses). Other issues include comparing greenhouse production in Spain with heated greenhouses in Northern European countries as suppliers of fresh vegetables during winter for the North European market, or the comparison of open-field production with intensive greenhouse production.

Based on the results of the additional energy consumption linked to the introduction of desalination, the additional CO_2 emissions for the production of 1kg of tomato is 0.066 kg CO_2 eq. (Table 9.11). Compared to the results from Torrellas et al. (2012) (0.25 kg CO_2 eq./kg) and Page et al. (2012) (0.39-1.97 kg CO_2 eq./kg), who present the impact of tomato production within an LCA approach in Campo de Dalías and supply to Sydney respectively, the additional carbon footprint is relevant, even more since the carbon footprint of the desalination plant has not been estimated here. However, other factors, like fertilizers or transport, contribute more to the carbon footprint.

Table 9.11. Additional energy and carbon footprints linked to the use of desalinated water, compared to groundwater.

	Water footprint	Additional Energy footprint[a]	Additional Carbon footprint[b]
1 m^3	1 m^3	4,2 kWh	1,260 kg CO_2 eq.[a]
Tomato (1 kg)	52,2 L	0,220 kWh	0,066 kg CO_2 eq.

[a] with 1 kWh = 0,300 kg CO_2 eq. (Gencat, 2013)
This is a first estimation, to obtain the order of magnitude of the consequences of integrating the carbon footprint of desalinated water, which should be refined.

The aim of this section is to have a broader view of the issues at stake when dealing with the repartition of desalination costs, introducing questions such as fairness and responsibility of the different users relative to the necessity to resort to this source of water. Here, desalination and groundwater are considered as the same pool from which urban and agricultural users obtain their resources[95]. This means that, beyond the fact that some users effectively receive groundwater or desalinated water, all users have to contribute to the costs of the water supply system. Three main questions arise: first, who should bear the costs, second, who benefits from the use of water and, third, who contributes to groundwater overuse, making resorting to desalination necessary. Thus, we explore the inter-sectoral repartition of the costs between cities and agriculture, which is hardly undertaken, as the issue of costs recovery is usually debated by sector (Barraqué, 2001).

[95] Surface water is not considered in this section to simplify the problem.

Different options or scenarios can be contemplated:

- *Scenario 1: 'Pay what you receive v.1'*: Users have to pay the full cost of the water they receive. In this first version, only farmers who receive desalinated water (8000 ha) have to pay for the desalinated water they receive.
- *Scenario 2: 'Pay what you receive v.1 – 0.30 €'*: Same as 'Pay what you receive v.1'; however, farmers will pay only 0.30€ for desalinated water, not the full cost. This situation will be effectively applied in Campo de Dalías (ACUAMED, 2006).
- *Scenario 3: 'Pay what you receive v.2'*: Same as 'Pay what you receive v.1'; however, all farmers of Campo de Dalías share the costs of desalinated water received by agriculture.
- *Scenario 4: 'Agriculture's fault'*: This scenario considers that the overuse of groundwater in the agricultural sector has generated the current situation. It means that the full cost of desalination should be borne by agriculture[96].
- *Scenario 5: 'Everyone's fault'*: Agriculture and cities participate in the cost of desalinated water proportionally to their respective WF. The current WF is considered here, but a variant of this scenario would include the accumulated WF from the time the total WF has been higher than the inflows to the aquifer, to take into account the responsibility in saltwater intrusion.
- *Scenario 6: 'Priority to basic needs'*: A minimum urban use, supplied from groundwater, aims at satisfying the basic needs of the population is contemplated (200 L/hab./day, i.e. 23 hm^3). The additional urban WF is considered as linked to economic activities and should participate fully in the desalinated water costs.
- *Scenario 7: 'Same cost'*: Farmers and city dwellers pay the same cost for water.

The results of the different scenarios are presented in Table 9.12. These scenarios distinguish the monetary cost and implications in terms of energy use for the agricultural sector and cities. The distribution of energy use is a proxy for the impacts in terms of CO_2 emissions.

Only under the scenario 'Pay what you receive', do city dwellers bear most of the cost of desalination. However, this option would only be acceptable if it recognized that agriculture has a higher legitimacy to use groundwater from the Campo de Dalías Aquifer. If one of the other scenarios is in fact fairer 'for society', this means that there is a cross-subsidy from cities to agriculture. This is also important for the application of the WFD, since it requires that final users pay the full cost of their water supply.

[96] An equivalent scenario for cities, i.e. where all the costs of desalination should be supported by urban use, is not considered. However, the 'Pay what you receive v.1' scenario is very similar, with 86.5% of desalinated water destined to cities.

Table 9.12. Attribution of water sources (groundwater and desalinated water) for different scenarios and related water cost and energy consumption for agriculture and urban supply.

	Agriculture (hm^3)		City (hm^3)		Cost ($10^{-2}€/m^3$) [a]		Energy (kWh/m^3) [a]	
	GW	Des.	GW	Des.	Agri.	Urb.	Agri.	Urb.
'Pay what you receive v.1"	43[b]	7[b]	6	38	22 [b]	56	1.59 [b]	4.63
'Pay what you receive v.1- 0.30 €'	43[b]	7[b]	6	38	17 [b]	61	1.59 [b]	4.63
'Pay what you receive v.2'	86	7	6	38	19	56	1.32	4.63
'Agriculture's fault'	48	45	44	0	38	15	3.03	1.00
'Everyone's fault'	64.5	31.5	27.5	13.5	30	30	2.38	2.38
'Priority to basic needs'	58	38	34	7	34	23	2.66	1.72
'Same cost'	64.5	31.5	27.5	13.5	30	30	2.38	2.38

[a] 0.15 €/m^3 and 1kWh/m^3 for groundwater – 0.62 €/m^3 and 5,2kWh/m^3 for desalinated water (excepting scenario 'Pay what you receive v.1- 0.30 €') / [b] Only farmers who receive desalinated water are considered in this scenario (they still use 43 hm^3 of groundwater in addition to desalinated water) / GW: groundwater; Des.: desalinated water; Agri.: griculture.; Urb.: urban supply

Other scenarios would be possible, for instance 'First users have legitimacy on groundwater', which would be similar to the situation of senior water rights in the United States (Shupe et al., 1989). In the scenario 'Priority to basic needs', the amount of water reserved for human consumption could also be changed. Another plausible scenario would be to consider the water supply to Almeria differently to the municipalities in Campo de Dalías. Furthermore, from a methodological point of view, this example illustrates how the process of determining potential cross-subsidies is based on the socio-political context and perceptions of what is fair for society.

Another implication from the use of desalinated water by the agricultural sector in terms of higher water costs can be obtained based on Table 9.12. It is often claimed that urban users are able to pay more for water and can afford desalination, contrary to farmers, who have a lower capacity to pay, and for whom it is not affordable to use desalinated water.

Indeed, doubling the cost of water, as a minimum, based on figures from Table 9.11, would impact significantly on the profits of many of the cropping rotations (Table 9.6). It is estimated farmers would bear an additional cost of 1,500 €/ha as an average. Thus, the profitability of many crops may not be assured if they were irrigated only with desalinated water. This situation serves as an argument to request subsidies for the agricultural sector (e.g. ACUAMED, 2006). However, in the baseline scenario for Campo de Dalías ('Pay what you receive v.1', Table 9.13), the final cost of water is much lower than in the option of using only desalinated water.

This is because the final water cost is the average between desalinated water and groundwater costs, as groundwater is still used as the first source of water. In the case of sharing the cost among all agricultural users (‘Pay what you receive v.2’), it is even lower.

In addition, when looking at the composition of the production costs (Table 9.6), it appears that water represents as an average 3.4% of the production costs. Other components such as labour or fertilizers have a much higher burden on the profitability of farms. However, among all these factors, water may be the only option where public subsidies can still be obtained[97]. On the side of the profit, crop market prices are governed by the market, at least in the case of vegetables, and direct subsidies are not possible.

IX.4.5 Desalination for future supply: a simple solution to a complex problem

The need to preserve agricultural use and its benefits for the local economy is not the only reason to justify subsidies for desalination plants. Environmental benefits – like e.g. stopping seawater intrusion, or arguments linked to the requirement of the WFD to reach the ‘good status’ of the groundwater body – are also usually presented as benefits from desalination. This is the case for instance in the *Viability assessment of the Campo de Dalías desalination plant* (ACUAMED, 2006). These benefits are viewed as public benefits for the whole of society, which justifies also subsidies for the desalination plant, particularly from the EU[98].

However, it might be questionable to support desalination based on the recovery of the groundwater body ‘good status’. It would be like assuming the cost of cleaning a river should be borne by the whole community on the basis that the activity of the polluters generates profits for the local economy, and that having a clean river is a benefit for everyone. The ‘poor status’ of the groundwater body is also the result of a private enterprise, and benefits for farmers, and the question of the acceptability of externalities and who should bear the cost is essential. Finally, the application of the EU WFD is a justification to increase water supply through desalination, with its associated environmental impacts.

The potential of future supply options to remediate to current situation also appears as optimistic. For instance, farmers are reluctant to use treated wastewater. Equally the example of the desalination plants in Almeria and Carboneras, which are far to be used to their full capacity, raises doubts on the actual future use of this resource. Indeed, the fact that energy requirements associated to the supply of water would more than double illustrates the potential

[97] This situation might change in line with the application of the EU WFD, where the full costs of water should be supported by the final users.

[98] There is a paradox of the EU potentially subsidizing infrastructures against its own principles, e.g. in terms of cost recovery. This used to take place also for water transfers (Barraqué, 2001).

difficulties. Users may be reluctant to pay more for water and they may keep using groundwater because it is cheaper and easily accessible. A smaller reduction in groundwater withdrawals over the next years would amplify the risk of saltwater intrusion.

Furthermore, the level of withdrawals in case of using the full capacity of desalination plants would in fact remain high. The Water Plan even recognizes that the desalination capacity should be raised to 60 hm^3 to balance supply and demand in the future. The lack of transparent decision-making, combined with high energy prices among other issues, raises important questions on the feasibility of desalination as the simple solution to a complex problem. The plan, however, is an attractive solution, since there are socially and politically difficult decisions and actions to be taken to reduce (partly illegal) groundwater withdrawals. Questioning the drivers and gradually changing the incentives has not been the main focus, thus missing the opportunity offered by the preserved access to the groundwater stock in the last thirty years, despite seawater intrusion, to evolve towards a less water-intensive economy.

Regarding the evolution of withdrawals into the future, one of the major worries for farmers are the costs of production (e.g. fertilizers) and market prices in a context of opening the EU market to production, principally from North-Africa. Faced with these uncertainties, one might claim it is better to use groundwater when it can be made productive and that a decrease in withdrawals would impact the economy of the area. This is linked to the view of groundwater development where the current intensive use could allow the development of an area up to a point when groundwater use will reduce thanks to the diversification of the economic sectors (see Chapter 3 on groundwater allocation). Then an 'exit plan' should be proposed to deal with the situation of the loss of the aquifer as a source of water, including a clear attribution of responsibilities. The Campo de Dalías Aquifer could have been the perfect example where this would happen, but this is not the case, as agriculture still sustains the local economy, despite the accumulated economic profits over the last forty years.

IX.5 Synthesis

- The total WF of the Campo de Dalías Aquifer amounts to 147 hm^3. The main use (132 hm^3) is the intensive irrigation of 16,500 ha of greenhouses mainly from September to May. Crops are grown during a long cycle of one crop, or two cycles of the same or different crops, which implies a challenge and the need to design specific methods to quantify the WF compared to 'classic' WF assessments. Furthermore, groundwater is used for other purposes, such as cleaning and soil disinfection.
- While the WF of crops 'at the farm gate' ranges between 35 and 90 L/kg, the Integral WF is comprised between 60 and 160 L/kg due to losses and waste in the supply chain.

- The virtual water flows to foreign countries amount to 66 hm^3 and accounts for 700 million euros.
- The comparative advantage offered by the natural conditions and the access to the EU market implies that the Direct Economic Value is high (10 €/m^3 as an average). However, rising inputs costs and a competition in the market imply uncertainty for farmers, who are not willing to pay more for water, despite this input represents 3.5 % of the costs. Thus, farmers succeeded in their strategy to 'externalise' water cost.
- Groundwater withdrawals are mainly originating from the lower Aquifer System that presents better quality than the upper aquifers, which have been used in the initial stage of development in the area. The groundwater table in the lower system has dropped as a result of intensive groundwater withdrawals, and evidences of seawater intrusions have been observed. Where intrusion has been more direct, withdrawals had to be reduced. Nevertheless, the relative isolation from the sea of the major part of the aquifer allowed keeping pumping with a limited indirect intrusion. It means that withdrawals should now be reduced to limit the replenishment of the aquifer by seawater. It represents a delayed consequence of intensive withdrawals over the last forty years.
- The proposed solution by the Authorities to reduce groundwater withdrawals and reach 'good status' under the WFD is to introduce desalination, together with wastewater reuse. However, the quality of groundwater and the direct cheaper access to this resource may imply farmers keep using it despite agreements on the use of desalinated water. As the cost of desalination will be mainly supported by urban residents, it is questionable whether this cross-subsidy is socially acceptable, since a clear responsibility on the state of the aquifer has not been discussed, particularly in relation with the WFD principles of 'cost recovery' and 'polluter pays'.
- Despite the window of opportunity offered by the relative isolation of the aquifer from the sea, and the benefits obtained from groundwater use over the last forty years, no exit plan has been devised towards a local economy less dependent on groundwater.

Chapter X

CONCLUSIONS AND RECOMMENDATIONS

X. CONCLUSIONS AND RECOMMENDATIONS

Groundwater is repeatedly depicted as an invisible resource, since it can be found underground, out of sight and out of mind; yet its role in sustaining surface water flows and ecosystems is in fact even more unperceived. This thesis has shown that a better knowledge of the resource base, on the one hand, and drivers and destinations for groundwater use, on the other hand, contribute to the identification of trade-offs and potentially to more robust decisions on groundwater use. The case studies have raised numerous issues in terms of allocation, decision-making, rule implementation and monitoring, simultaneously to the description of the flows, footprints and values. The most relevant aspects are reflected in these conclusions.

Contributions relative to the Water Footprint approach

The thesis has introduced advances on the water footprint concept. As a general approach, the usual method to quantify water footprints has been challenged as it assumes the satisfaction of irrigation water needs. Irrigation occurs mostly in water scarce countries, where this assumption hardly holds as water availability is limited, even for groundwater, due to pumping costs, among other factors. Other methodological advances of this thesis are presented in Table 10.1.

More generally, the water footprint is a flexible and multi-scalar indicator of water resources *appropriation* by humans under two axes: the water not delivered back to the river basin, or aquifer, after use, and the attribution of direct and indirect water consumption to any entity of the supply chain or geographical area. Thus, while including the traditional vision of water management, it highlights also the role of consumers as drivers for water resource consumption and succeeded in spurring the interest of the business sector, for both risk assessment and environmental reporting.

A major challenge is to integrate the complexity of water management in the interpretation of water footprints and recommendations. For instance, real options to reduce the water footprint of crops are difficult to assess, as the relation between yield, water consumption and other inputs is complex. Water productivity, the inverse of the water footprint, has been a long debated issue by agronomists. Thus, a view in terms of water footprints can bring interesting ideas as long as simplistic interpretations are avoided. An impact-weighted water footprint, linked to Life Cycle Analysis, identically falls short if it is too directly interpreted, since direct interpretation is more complex, due to a scarcity factor that adds to water productivity issues.

New opportunities for water footprint reduction arise from the concept of 'Integral Water Footprint' proposed in the thesis to associate any entity of the supply chain to the whole water consumption within the life cycle of a product. While the water footprint approach has incentivized companies to work with suppliers, this concept engage them also 'downstream'.

Table 10.1. Main advances of the thesis regarding the water footprint concept and methods and case studies where they have been applied

Topic	Details and comments	Western Mancha Aquifer	Guadalquivir River Basin	La Loma de Ubeda Aquifer	Campo de Dalías Aquifer
Water footprint accounting for agriculture	The traditional methodology focuses on crop irrigation demand and not on the real water consumption. This thesis, considered applied irrigation water.	x	x	X	X
Green Water footprint	Integration of green water consumption in fields out of the cropping season.		x		
	Attribution between a (human) green WF and ecosystems green water consumption.		x		
Dams water footprint	Evaporation from dams generates an additional WF		x		
Agricultural blue water footprint	In greenhouse farming, irrigation is not the only use of water. This should be considered.				X
Water footprint of seawater intrusion	A method based on the period during which the aquifer is made unavailable is proposed.				X
Integral Water Footprint	This is the water consumption in the full life cycle. It reflects that losses of materials in the value chain increase the WF.				X

Full integration of trade-offs from groundwater use in decision-making: a paradigm shift?

Although the impacts on flows and ecosystems have been acknowledged for a long time and sometimes integrated into groundwater management, this has been rather limited to cases where a special dependence on groundwater was identified. This role is disregarded or falsified in the traditional vision of groundwater as a stock, which focuses on recharge or inflows, identified as the resource. It has been evidenced, for instance, in the following situations:

- 'Available groundwater resources' are usually regarded at the scale of the aquifer or groundwater body. Yet the notion of 'availability' should be reserved to the agreed amount at the scale of the river basin.
- Despite the objectives of the WFD, the integration of the environmental dimension in its implementation for groundwater is questionable. Induced recharge is not contemplated

and some criteria are reminiscent of a traditional approach, in spite of a general requirement to integrate the impacts on the environment. This triggers a ‘domino effect’ toward the traditional approach, as revealed by the analysis of Water Plans for Spain.

- In the Western Mancha Aquifer, the conservation of the Tablas de Daimiel wetland depends on a high groundwater table. Yet the focus is usually on groundwater stock and its recovery, despite the case being a textbook example of a groundwater-fed wetland.
- In the La Loma de Úbeda Aquifer, the value of ‘available resources’, which are not evaluated at the river basin scale, is the basis for the legalization of aquifer users.
- When groundwater is presented as a Common-Pool Resource, the view of a stock is reproduced since this approach considers impacts like rising pumping costs and stock availability reduction for other current and future users, and it disregards flow impacts.

The concept of capture allows presenting a view in terms of flow continuity in the watershed, through the identification of the change in flows to and from aquifers, as a result from pumping. ‘Future capture’ has been defined for the first time to illustrate that stock depletion, inherent to any aquifer development, also results in the disruption of flows once pumping is reduced. Thus, withdrawals are the sum of current and future capture. The idea of future capture also suggests the potentiality for aquifer replenishment. This relativizes the implications of stock depletion, often mistakenly identified as ‘non-renewable resources’ or ‘groundwater mining’.

In many situations, groundwater generates flow impacts before the consequences of stock consumption are severe. Decision-making should fully integrate the knowledge on current and future flow impacts. This would highlight the potential conflicting uses or trade-offs in groundwater resource development and contribute to the definition of acceptable thresholds. Furthermore, the complexity of trade-offs assessment and valuation raises doubts on the direct formulation of criteria for ‘efficient / optimal’ use or the ‘sustainability’ of groundwater use. Sustainability constitutes more an ideal, where all the values would combine in harmony and it can be contemplated as a ‘nirvana concept’ that distracts from practical recommendations.

The case studies have presented situations where groundwater withdrawals affect aquifer and basin users and the environment in different manners (Table 10.2). A main result of the thesis is that decision-making processes and institutions for groundwater management should integrate these varying characteristics, actors and values. In the La Loma de Úbeda Aquifer, groundwater availability for aquifer users should be formulated in coordination with all the uses in the river basin, including requirements in terms of environmental flows. In the Western Mancha Aquifer, decisions should integrate the acceptable impacts on the wetland. In the Campo de Dalías, withdrawals affect primarily the own aquifer users through potential loss of quality, even if some ecosystems and local population were originally benefiting from the natural outflows.

Table 10.2. Impacts associated to groundwater use for the three case studies.

Impacts	La Loma de Úbeda Aquifer	Western Mancha Aquifer	Campo de Dalías Aquifer
Downstream impacts	✓		
Ecological impacts	✓	✓	
On connected aquifers	?	✓	
Seawater intrusion	Inland	Inland	✓
Stock impacts	✓	✓	✓
Stock depletion / Future capture	A few years once pumping stops (fractured carbonate aquifers well connected to areas of capture)		

As groundwater governance should be modulated according to the impacts of groundwater use at multiple scales, collective management by the direct aquifer users is not an institution that can be promoted as a general rule. This proposition usually stems from the description of groundwater as a Common-Pool Resource at the scale of an aquifer, and the assumption that coordination will lead to mutual benefits, for users and society, compared to the myopic use. Yet, the users of an aquifer do not have an inherent benefit to conserve all the values linked to groundwater. Nevertheless, the inclusion of aquifer users in the decision-making arena should be certainly promoted, as they are main stakeholders together with other users at basin scale.

The Common-Pool Resource definition is also usual in the economic modelling of groundwater. Yet, by disregarding downstream and ecological impacts, these models cannot claim to represent a general approach. Introducing these impacts would change radically the conditions of 'optimal' use and the conclusions reached. The 'Gisser-Sánchez effect' is a typical example.

Main results for the whole Spain and the case studies

In spite of its pitfalls, the process of WFD implementation has produced valuable data that allowed us to better characterize the use of groundwater in Spain. Total groundwater use in Spain can be estimated to reach 6,700 hm^3 per year as an average, with agriculture as the main use (5,000 hm^3), followed by urban supply (1,400 hm^3) and own supplied industries (300 hm^3). Three river basins, Guadalquivir, Duero and Júcar, concentrate more than half of the withdrawals. On a total of 712 groundwater bodies, 296 have been classified in 'poor status', 166 for quantitative reasons and 229 for qualitative reasons. In the WFD implementation many groundwater bodies have been characterized in detail. The interaction between groundwater and surface water and dependent ecosystems has received much attention in some cases. However, additional effort for the characterization of these interactions should be done in many districts. The consideration of a fixed percentage of inflows for environmental flows is also problematic.

In the Western Mancha Aquifer, the water footprint reached a value of 300 hm^3 on the period 2007-2009, while an effective improvement of the situation of the wetland would require a water footprint not higher than 90 hm^3. The discrepancy between these two figures suggests that an economy based on the intensive use of groundwater in the current order of magnitude is incompatible with wetland conservation. This should be taken into account when policies are designed, particularly when high amounts of money are involved, as it questions their potential for success. Faced with illegal activity, the Special Upper Guadiana Plan (2007-2012), has promoted the buying of rights to reduce the pressure on the wetland and the regularization of illegal vine irrigators, conditioned by a reduction of withdrawals per hectare. This may have contributed to changing the conditions and made sanctions more acceptable by users, increasing compliance, which is a basic aspect in order to meet the agreed level of withdrawals.

Groundwater use in the Guadalquivir River Basin in the last two decades has been marked by the boom in olive groves irrigation, particularly in the river basin headwaters. La Loma de Úbeda Aquifer is one of the most representative examples. Whereas the Authorities have been reluctant to recognize these uses initially, the definition of an available groundwater resource has opened the way to legalization. Whether this will contribute to the limitation of withdrawals to the agreed level will depend of the efficacy of the sanctioning regime, in a conflicting and politicized context. At the scale of the whole basin, olive grove irrigation, among other projects of irrigation extension and modernization, has added pressure on the whole river basin which is now a ‘closed’ river basin. The existence of uses with little economic return implies that water rights reallocation or water markets could allow the more productive users to get access to water. This could be proposed if all the consequences associated are compensated for, particularly in terms of possible intensification of water use, change in place and time of use, and acceptability for the users that are deprived from their right to use water.

In the Campo de Dalías Aquifer, the specific geological configuration, with the aquifer partially isolated from the sea, implies that groundwater withdrawals could continue during many years, in spite of evidences of seawater intrusion. Desalination is now presented as the solution to balance supply and demand. Yet its cost makes it hardly probable that the plant capacity will be fully used. A series of issues can also be raised from an ethical and equity point of view, mainly related to cost recovery, as the responsibility of the different groundwater users has not been debated. It is also arguable that WFD compliance serves as a justification for desalination.

Regarding the comparison of the case studies in terms of water footprints and economic value generated by agriculture, Table 10.3 synthesises the main results. It is noteworthy that, while representing only 2.5 % of the water footprint of groundwater in Spain, the Campo de Dalías generates 23 % of the market value linked to groundwater use for agriculture in Spain[99]. As a comparison, with an area over eight times larger and a water footprint three times higher, the Western Mancha Aquifer generates only 8 % of the market production value of the country. As a result, the Direct Economic Value of water is eight times higher in the Campo de Dalías compared to the Western Mancha and Guadalquivir River Basin.

Table 10.3. Synthesis of the water footprint and the main indicators in the different case studies

		Western Mancha Aquifer	Guadalquivir River Basin (groundwater)	La Loma de Úbeda Aquifer	Campo de Dalías Aquifer
Irrigated area	in ha	125,000	-[a]	40,000	16,900
Agriculture blue water footprint	in hm^3	296	726 / 748[b]	63	103
	share on total for Spain (groundwater) (%)[c]	7.5	18.5	1.5	2.5
	in m^3/ha	2,400	-[a]	1,600	6,100
Total market value	in million euros	377	921	112[d]	1,080
	share on total for Spain (groundwater / total irrigated) (%)[e]	8 / 2.5	20 / 6	not relevant	23 / 7
Direct Economic Value	average (in €/m^3)	1.3	1.3[f]	1.75[d]	10.3
	range of values (in €/m^3)	*0.3 to 3.5*			*8.4 to 13.5*
Land productivity	in €/ha	3,000	-[a]	2,800	62,000

[a] Unknown since a significant share of groundwater use takes place in conjunctive use with surface water. / [b] Relative to groundwater and varying depending on the year. / [c] An efficiency of 0.8 is assumed for the figure obtained in this thesis (5000 hm^3, see Table 4.8). / [d] Average 2005-2011. / [e] Market value of agriculture production in Spain:15,000 million euros; groundwater: 4,700 million euros (De Stefano et al., 2011). / [f]Based on De Stefano et al. (2011) since the market value of irrigated agriculture has not been distinguished specifically for groundwater in this thesis.

Recommendations for the management of complex groundwater socio-ecological systems

Two main measures can ease the pressure on overused aquifers: acting against illegal use and reducing water rights, or other recognized uses. In the case studies, the groundwater regulation in force is not complied with. Large scale illegal use contributes to the local economy, which makes it hard to take action. Faced with the *fait accompli*, public authorities partly recognized this use, like in the Western Mancha and La Loma de Úbeda aquifers. However, as a general rule, the argument of the economic relevance of illegal use should not be used for a blank regularization. Moreover, this is uncertain, whether the agreed groundwater allocations will be complied with. In Campo de Dalías, the problem of illegality has been disregarded so far, in a

[99] This figure should be relativized since the method for the total market value for Spain by De Stefano et al. (2012) has been different to the one by this thesis; the two cycles of crops may not be included.

wait-and-see position of the Authorities as the desalination of seawater, a technical artefact, is perceived as *the* solution.

When enacted uses are excessive, it generally requires negotiating a reduction of use or buying rights or compensating the user, as the deprivation of use means a loss of revenue. This has been illustrated in the case of the Western Mancha Aquifer. Farmers must have a profitable activity and need a secured water tenure to lead this activity. Whether enacted uses should be reduced or illegal use integrated is consequently a matter of context depending on the conditions under which uses are legitimate for society and can be changed.

As regards policy instruments, the definition of a cap on withdrawals is a prerequisite for any instrument that aims at allocating groundwater among the users of an aquifer. For instance, a water market is based on the trading of previously defined water rights and water pricing aims at limiting use to an established amount. Meanwhile, the access to water is necessarily restricted to some users, which implies that the adoption of these instruments is a highly political process, as illustrated by the case studies. Policy instruments must also take account of prevailing conditions of rights, and the potential situation of overuse and illegality, which is hardly the case when they are theoretically formulated. This adds to the arguable definition of productivity and efficiency, since monetary valuations are limited to some aspects of decision-making. The potential intensification of water use should also be addressed, as more productive uses in terms of profits are usually more water consuming, as shown in the Western Mancha. These multiple issues are certainly disregarded due to a preconceived ideological position in favour of markets and pricing, compared to quotas or more direct reallocations, which are associated with a vision of command-and-control by the State (see Barraqué, 2002; Petit, 2002).

Another determining issue regards the economic incentives for groundwater use. The difficulty and costs to both reduce rights – as compensation is necessary – and stop illegal use has its origin in the incentives. The role of subsidies is particularly determinant. The case of olives and vineyards show how subsidies have been a key factor to maintain over-production, while encouraging groundwater use. Action from the authorities to reallocate water can appear complicated and costly, while changes in incentives or markets have direct consequences that are usually more accepted by users. The debate on EU CAP subsidies is currently focused on an agriculture that should have ecological value, while maintaining enough income for farmers. Yet it is unclear how the impacts on water resources will be included compared to factors such as the carbon sequestration by woody plants, like olives and vines.

The baseline situation against which the effects and costs of policies should be evaluated is also an important point. Action is often required to reverse dropping groundwater levels, which jeopardize the future use of the aquifer for the ‘next generations’. In fact, in the absence of values for the environment or at the basin scale, the evolution of the aquifer in case of no action may not be so problematic in many cases since, e.g. there is sometimes a substantial recharge that allows the replenishment of the aquifer in a few years. As rising pumping costs decrease demand, there is also a certain ‘self-regulation’ for the system. In the Western Mancha, when groundwater levels drop, higher pumping costs reduce demand, at least for cereals and potentially for vines or other crops, depending on market conditions. Thus, the relevance of a policy, and its costs, should be weighed against this baseline and the probability of success. It should also be kept in mind that restrictions due to ‘external’ factors or pumping costs are more accepted by users than an ‘administrative’ decision to reduce pumping, although the effects are the same in terms of revenue losses.

An explanation for the tolerance of overuse and the illegal use of groundwater may also lie in the lack of knowledge and mistaken representations, where the view of a renewable resource available for local use prevails. Changing this vision would contribute to the emergence of counterbalancing powers in order to influence the decision-making, e.g. surface water rights holder, who would have their use jeopardized, or environmentalists fully aware of the role of groundwater in the river basin. This idea is intertwined with the approach of the thesis, which aimed at describing the trade-offs of groundwater use, on the premise that decisions are usually based on a limited view. Knowledge diffusion should also be promoted for the actors of the supply chain, particularly consumers who can be considered as a main driver for resource use. The water footprint approach contributes to this view. For instance, consumers are usually unaware of vineyards and olive groves irrigation; increased awareness on this fact, and the related environmental consequences, could lead to a demand for more regulation or specific labels to ensure a limited water footprint or rain-fed production. Making clear the costs and benefits generated by the use of groundwater in Campo de Dalías Aquifer, might also lead to a re-evaluation of the issue of water supply sources financing.

Regarding the monitoring of withdrawals, while this implies a certain cost because of the specific nature of groundwater, this should be facilitated thanks to new technologies (e.g. remote sensing) and the cooperation between farmers and public authorities. The historical opacity and difficulty to have access to data on groundwater should gradually be changed. Finally, compliance with the rules can vary substantially according to the process for their formulation and who is in charge to apply them and control.

As a final note, it can be observed that this thesis has proposed different paths to improve how decisions are taken on groundwater – in general and illustrated by the case of Spain and four specific case studies –, integrating better conflicting values and impacts linked to the use of this resource. Particularly, the transition to a better state for groundwater resources is linked to the inclusion of stakeholders and values that are traditionally disregarded in decision-making. Thus, this thesis has shown that knowledge or, more precisely, the way we look at a social-ecological system is determinant for better decisions and for changes in the management of this system. Considering the impacts of pumping in terms of a change in flows in the watershed has been the first pillar in this thesis toward this change of perspective.

The water footprint is the second main pillar that illustrates the importance of knowledge and change of focus. First, it constitutes a common procedure for water accounting, with an emphasis on the appropriation of water flows, i.e. consumptive use. Second, it associates new stakeholders and entities of the supply chain to the impacts on water resources. These entities are therefore also a lever for change, whether directly, through a change in their consumption pattern or actions to reduce the water footprint in collaboration with an upstream or downstream entity, or indirectly, through an increased awareness and a requirement of regulations.

Finally, the thesis has insisted on the complexity and the political aspects of the regulation of resource use, compared to an approach based on optimality and efficiency that is in fact partial or simplified due to the difficulty of integrating all the values involved. Indeed, all policy instruments imply a change in the repartition of values among the different users of a resource and the question if this ultimately results in a better state for society is a complex issue.

While incentives for groundwater use are mostly established at the national, regional or global scale, e.g. through market prices or energy prices, this thesis has insisted on decisions and drivers at the local scale and on the producer-consumer spectrum in the economy. Thus, it has opened the way to explore numerous options to improve the overall frame to help build better decisions and innovative institutions on groundwater.

A possible follow-up of this work would be to assess or support a process of decision-making that would integrate effectively the role of groundwater as analysed in this thesis, based on the criteria and new concepts proposed. This means a proactive research compared to the case studies assessment in the present work. This would particularly imply exploring innovative institutional arrangements in coordination with the stakeholders, with an attention to rules emergence and compliance and users involvement, rooted in the reality of groundwater dynamics and beyond usual panaceas. Another clear follow-up would be to contribute further to the debates on the water footprint methodology and apply the concept from other perspectives, potentially for different entities of the supply chain, such as companies, in order to identify options for real and accountable reduction in their water resource appropriation.

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APPENDICES

APPENDIX 1.

SEMINARS, MEETINGS AND INTERVIEWS RELATED TO THE CASE STUDIES

I – <u>Western Mancha Aquifer</u>

- Technical seminar on the 'Special Upper Guadiana Plan implementation and irrigation' organized by Consorcio del Alto Guadiana and Castilla-La Mancha University (February 2011).

- Field visit in Western Mancha Aquifer organized in relation to a project from the 'Instituto de Empresa Business School' (Madrid) (March 2011):
  - *Attendance to a focus group with farmers and interviews*
  - *In particular: Interview of Enrique Calleja, Director of Consorcio del Alto Guadiana (the organisation in charge of the Special Upper Guadiana Plan implementation)*

- Public meeting on the draft Water Plan of the Guadiana Basin District organized by the Guadiana Basin Authority (Confederación Hidrográfica del Guadiana) (June 2011).

II – <u>Guadalquivir River Basin and La Loma de Úbeda Aquifer</u>

- Presentation of the results of the study of the Guadalquivir River Basin water footprint at the headquarters of the Guadalquivir Basin Authority and discussion with various senior officers (April 2011).

- Field visit in La Loma de Úbeda Aquifer (July 2011):
  - *Field visit organized by Ignacio Rubiales (Asociación Pozos de La Loma-Úbeda)*
  - *Attendance to focus groups organized by Marta Rica in relation to her doctoral thesis.*

III – <u>Campo de Dalías Aquifer</u>

- Field visit in Campo de Dalías Aquifer 1 (November 2011):
  - *Attendance to focus groups and interviews organized by Marta Rica in relation to her doctoral thesis.*

- Visit in Almería (November 2012):
  - *Meeting to identify the main debates in the Campo de Dalías Aquifer. Participants:*

    -Antonio Canovas (Junta de Andalucía);

    -Hermenegildo Castro (Centro Andaluz de seguimiento y evaluación del Cambio Global);

    -Joan Corominas (ex-Secretario General de Aguas, Junta de Andalucía)

    -Andrés Cuadrado (Comunidad de Regantes "Sierra de Gador");

    -Patricia Domínguez Prats (IGME);

    -Manuel García Quero (Junta Central de Usuarios del Poniente Almeriense);

    -Francisco Javier Martínez Rodríguez (Diputación de Almería);

    -Jerónimo Pérez Parra (Junta de Andalucía);

    -Antonio Pulido Bosch (Universidad de Almería);

    -José López Gálvez (Universidad de Almería)
  - *Interview with Antonio Canovas and Jerónimo Pérez Parra (Junta de Andalucía)*

- Field visit in Campo de Dalías Aquifer 2 (May 2013):
  - *Visit of Las Palmerillas Experimental Station (Fundación Cajamar)*
  - *Individual meetings to collect data and interviews:*

    -Ana Cabrera (Economist, Las Palmerillas Experimental Station)

    -Antonio Canovas (Consejería de Agricultura, Junta de Andalucía)

    -Andrés Cuadrado (President, Comunidad de Regantes "Sierra de Gador")

    -Maria Dolores Fernández (Agronomist, Las Palmerillas Experimental Station)

    -Patricia Domínguez Prats (Hydrogeologist, IGME)

    -Francisco Javier Martínez Rodríguez (Engineer, Diputación de Almería)

    -José Rivera Menéndez (Grupo Ecologista Mediterranea, author of the thesis: La política de colonización agraria en el campo de Dalías : (1940-1990))

    -Maria José Morales (Consejería de Agricultura, Junta de Andalucía)

    -José Antonio Poveda (Secretary, Junta. Central de Usuarios del Acuífero del Poniente Almeriense).

    -Antonio Pulido Bosch (Universidad de Almería)

    -Maria Cruz Sánchez Guerrero (IFAPA, Junta de Andalucía)

    -José Vicente Simón (Director, Oficina Comarcal Agraria La Mojonera)

    -Agencia Andaluza del Agua (delegación de Almería)

APPENDIX 2.

ASSESSMENT OF THE 'RISK OF NO COMPLIANCE' WITH THE ENVIRONMENTAL OBJECTIVE OF THE WATER FRAMEWORK DIRECTIVE. MAY 2005

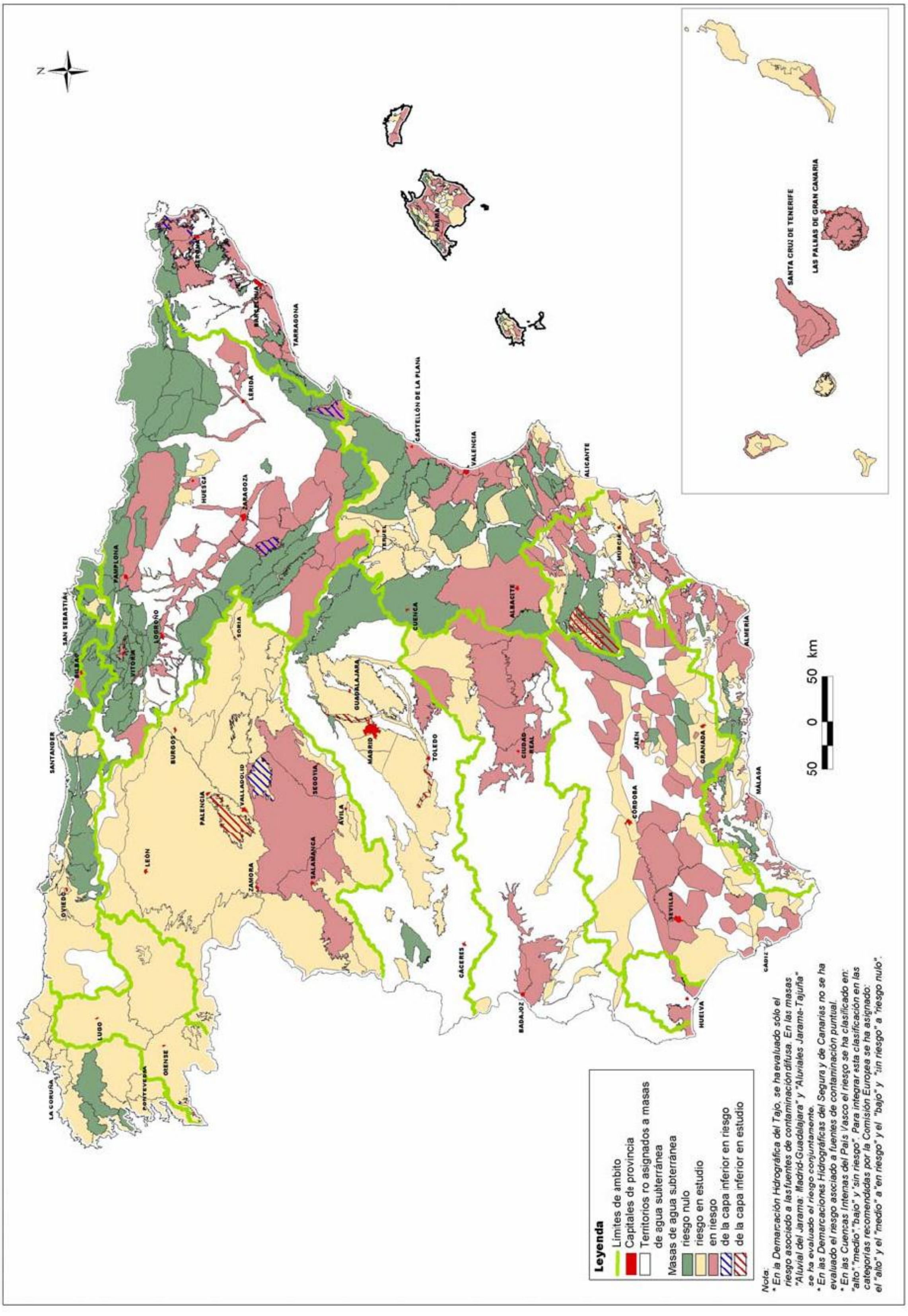
<u>Source:</u> Ministerio de Medio Ambiente (2006) Síntesis de la información remitida por España para dar cumplimiento a los Artículos 5 y 6 de la Directiva Marco del Agua, en materia de aguas subterráneas. Memoria. Madrid, Ministerio de Medio Ambiente, Dirección General del Agua, 85 pp.

<u>I – Synthesis of the assessment of the 'risk of no compliance' with environmental objectives (art. 4 of the Water Framework Directive)</u>

	Demarcación Hidrográfica		Masas de agua subterránea en riesgo					Riesgo en estudio	Riesgo nulo
	Nombre	Número de masas	Químico			Cuantitativo	En riesgo		
			Puntual	Difuso	Intrusión	Extracción			
Cuencas Intercomunitarias	Norte I	6	0	0	0	0	0	6	0
	Norte II y III	34	0	0	0	0	0	12	22
	Bidasoa, Nive Y Nivelle	2	0	0	0	0	0	1	1
	Duero	31	0	3	0	1	3	28	0
	Tajo [1]	24	0	1	0	0	1	18	5
	Guadiana [2]	20	0	9	1	6	11	9	0
	Tinto, Odiel y Piedras	4	0	3	1	0	3	1	0
	Guadalquivir	71	1	21	1	19	35	29	7
	Segura [3]	63	0	1	2	25	25	33	5
	Júcar	79	0	13	8	23	29	26	24
	Ebro	105	11	29	0	1	35	7	63
	SUBTOTAL	**439**	**12**	**80**	**13**	**75**	**142**	**170**	**127**
Cuencas Intracomunitarias	Galicia Costa	18	0	0	0	0	0	15	3
	Cuencas Internas del País Vasco [4]	14	2	0	0	0	2	0	12
	Cuencas Internas de Cataluña [5]	39	23	23	10	10	25	0	14
	Baleares	90	42	36	30	41	42	35	13
	Cuenca Mediterránea Andaluza	67	1	20	11	23	29	23	15
	Canarias [6]	32	0	8	8	15	19	13	0
	SUBTOTAL	**260**	**68**	**87**	**59**	**89**	**117**	**86**	**57**
TOTAL		**699**	**80**	**167**	**72**	**164**	**259**	**256**	**184**

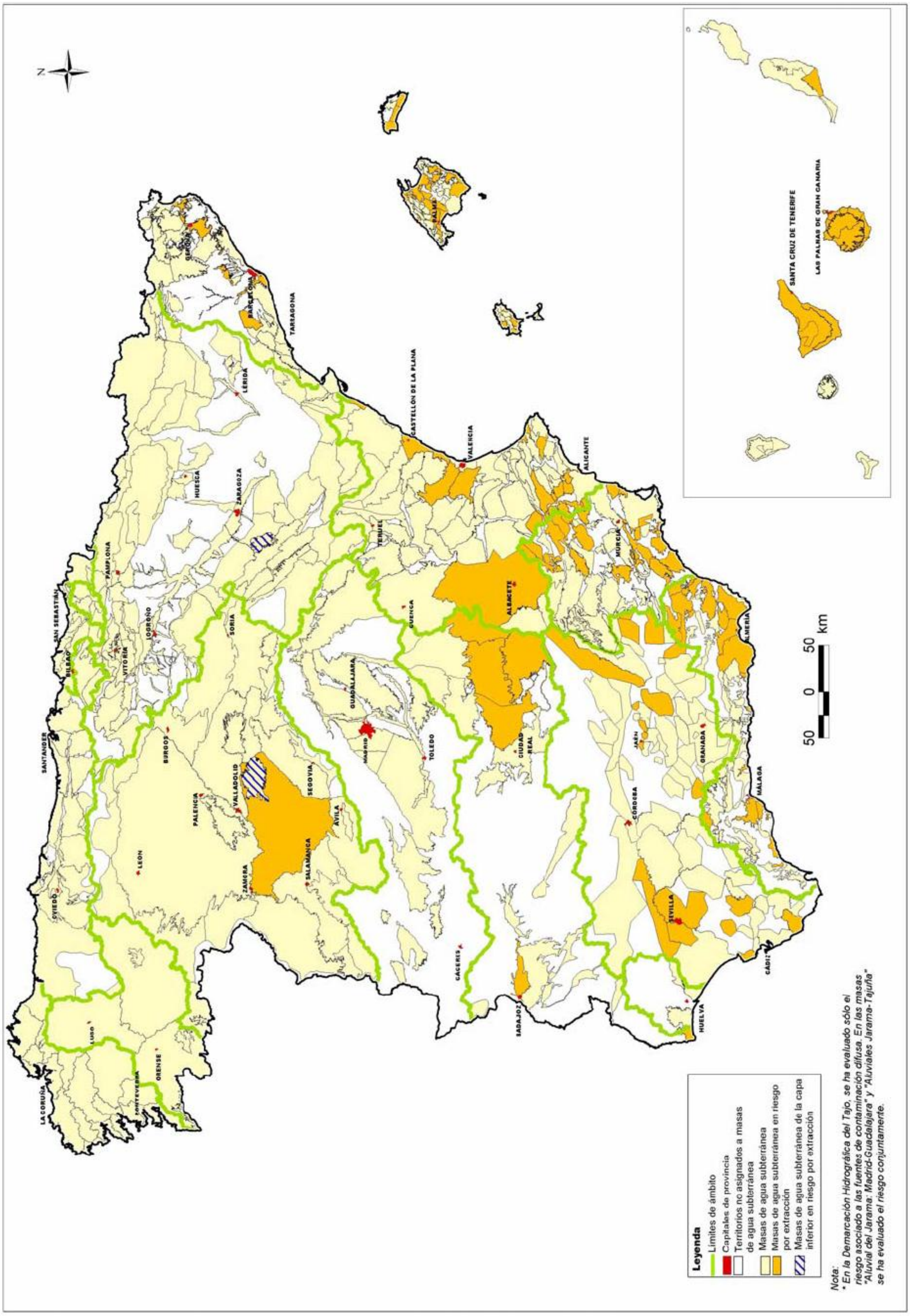
Note: In the left-hand bottom corner of the following maps, differences in the assessments between river basin districts are noted by the authors of the report.

II - Groundwater bodies identified by the river basin districts ‘at risk of no compliance’ with the environmental objectives of the Article 4 of the Water Framework Directive. May 2005



Mapa nº 20.- Masas de agua subterránea identificadas por los Organismos de cuenca en riesgo de no alcanzar los objetivos medioambientales del artículo 4 de la DMA. Mayo de 2005

III - Groundwater bodies identified by the river basin districts ‘at risk of no compliance’ with the environmental objectives of the Article 4 of the Water Framework Directive, due to withdrawals. May 2005



Mapa nº 24.- Masas de agua subterránea **identificadas por los** Organismos de cuenca en riesgo **de** no alcanzar los **objetivos medioambientales del artículo 4** de la **DMA**, **debido a extracción. Mayo de 2005**

APPENDIX 3.

SECTION OF THE 'WATER PLANNING REGULATION' RELATIVE TO THE QUANTITATIVE STATUS OF GROUNDWATER BODIES AND DEFINITIONS OF RELEVANT TERMS (IN SPANISH)

Source: MINISTERIO DE MEDIO AMBIENTE, Y MEDIO RURAL Y MARINO *ORDEN ARM/2656/2008, de 10 de septiembre,por la que se aprueba la instrucción de planificación hidrológica.*

I - Main definitions from Section 5.2.4.1 of the Water Planning Regulation / Definiciones principales introducidas por el Apartado 5.2.4.1 de la Instrucción de Planificación Hidrológica

Índice de explotación: "(…) se obtiene como el cociente entre las extracciones y el recurso disponible. Este indicador se obtendrá con el valor medio del recurso correspondiente al periodo 1980/81-2005/06 y los datos de extracciones representativos de unas condiciones normales de suministro en los últimos años."

Recurso disponible: "(…) se define como el valor medio interanual de la tasa de recarga total de la masa de agua subterránea, menos el flujo interanual medio requerido para conseguir los objetivos de calidad ecológica para el agua superficial asociada para evitar cualquier disminución significativa en el estado ecológico de tales aguas, y cualquier daño significativo a los ecosistemas terrestres asociados. El recurso disponible se obtendrá como diferencia entre los recursos renovables (recarga por la infiltración de la lluvia, recarga por retorno de regadío, pérdidas en el cauce y transferencias desde otras masas de agua subterránea) y los flujos medioambientales, requeridos para cumplir con el régimen de caudales ecológicos y para prevenir los efectos negativos causados por la intrusión marina."

Recursos renovables: "recarga por la infiltración de la lluvia, recarga por retorno de regadío, pérdidas en el cauce y transferencias desde otras masas de agua subterránea"

II - Section 5.2.4.1 of the Water Planning Regulation on 'Assessment of groundwater body quantitative status' / Apartado 5.2.4.1 de la Instrucción de Planificación Hidrológica sobre 'Estimación del estado cuantitativo de una masa de agua subterránea'

La evaluación del estado cuantitativo de una masa o grupo de masas de agua subterránea se realizara de forma global para toda la masa mediante el uso de indicadores de explotación de los acuíferos y de los valores de los niveles piezómetros.

Para cada masa o grupo de masas de agua subterránea se realizará un balance entre la extracción y el recurso disponible, que sirva para identificar si se alcanza un equilibrio que permita alcanzar el buen estado. Como indicador de este balance se utilizará el índice de explotación de la masa de agua subterránea, que se obtiene como el cociente entre las extracciones y el recurso disponible. Este indicador se obtendrá con el valor medio del recurso correspondiente al periodo

1980/81-2005/06 y los datos de extracciones representativos de unas condiciones normales de suministro en los últimos años.

El recurso disponible en las masas de agua subterráneas se define como el valor medio interanual de la tasa de recarga total de la masa de agua subterránea, menos el flujo interanual medio requerido para conseguir los objetivos de calidad ecológica para el agua superficial asociada para evitar cualquier disminución significativa en el estado ecológico de tales aguas, y cualquier daño significativo a los ecosistemas terrestres asociados.
El recurso disponible se obtendrá como diferencia entre los recursos renovables (recarga por la infiltración de la lluvia, recarga por retorno de regadío, pérdidas en el cauce y transferencias desde otras masas de agua subterránea) y los flujos medioambientales, requeridos para cumplir con el régimen de caudales ecológicos y para prevenir los efectos negativos causados por la intrusión marina.

Para determinar el estado cuantitativo se utilizarán también como indicadores los niveles piezométricos, que deberán medirse en puntos de control significativos de las masas de agua subterránea. En los casos en que existan diferencias espaciales apreciables en los niveles piezométricos se realizarán análisis zonales.
Se considerará que una masa o grupo de masas se encuentra en mal estado cuando el índice de explotación sea mayor de 0,8 y además exista una tendencia clara de disminución de los niveles piezométricos en una zona relevante de la masa de agua subterránea.
Asimismo se considerará que una masa o grupo de masas se encuentra en mal estado, cuando esté sujeta a alteraciones antropogénicas que impidan alcanzar los objetivos medioambientales para las aguas superficiales asociadas que puede ocasionar perjuicios a los ecosistemas existentes asociados o que puede causar una alteración del flujo que genere salinización u otras intrusiones.

APPENDIX 4.

DROUGHT LEVEL BY DISTRICT IN THE GUADALQUIVIR RIVER BASIN

Source: MARM (2010) Boletines Hidrologicos. Reserva Hidraulica. Datos de reserva: desglose por embalses.

Management District	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
Salado del Morón	Normal	Normal	Alert	PreAlert	Normal	Normal	Normal	Normal	PreAlert	Alert	Alert	PreAlert
Campiña Sevillana*												
Alto Genil	Normal	Normal	Normal	PreAlert	Normal	Normal	Normal	Normal	Normal	PreAlert	PreAlert	Emergency
Jaén	Normal	Normal	PreAlert	Alert	Normal	Normal	Normal	Normal	PreAlert	Alert	Emergency	Emergency
Hoya de Guadix	Normal	Normal	PreAlert	Alert	PreAlert	Normal	Normal	Normal	Normal	PreAlert	PreAlert	Alert
Alto Guadiana Menor	Normal	Normal	PreAlert	Alert	Normal	PreAlert	Normal	Normal	PreAlert	Alert	PreAlert	Alert
Rumblar	Normal	Normal	Normal	PreAlert	Normal	Normal	Normal	PreAlert	PreAlert	Alert	Emergency	Alert
Guadalmellato	Normal	Normal	Normal	Alert	Normal	Normal	Normal	Normal	Normal	PreAlert	PreAlert	Alert
Bembézar-Retortillo	Normal	Normal	Normal	PreAlert	Normal	Normal	Normal	Normal	Normal	Normal	Normal	Normal
Rivera de Huesna	Normal	Normal	Normal	PreAlert	Normal	Normal	Normal	Normal	Normal	PreAlert	PreAlert	Normal
Viar	Normal	Normal	Normal	Alert	Normal	Normal	Normal	Normal	Normal	Alert	Normal	Normal
Sevilla	Normal	Normal	PreAlert	Alert	Normal	Normal	Normal	Normal	PreAlert	Alert	Alert	Alert
Almonte-Marismas*												
Regulación General	Normal	Normal	PreAlert	Alert	Normal	Normal	Normal	Normal	PreAlert	Alert	Alert	Alert

Note: 'Campiña Sevillana' and 'Almonte-Marismas' have groundwater as their main water source. Thus, there is no restrictions linked to water availability.

APPENDIX 5.

BALANCE OF THE INFLOWS AND OUTFLOWS OF THE THREE GROUNDWATER BODIES COVERING THE WESTERN MANCHA AQUIFER

Source: Martínez-Cortina, L., Mejías-Moreno, L., Díaz-Muñoz, J., Morales-García, R. & Ruiz-Hernández, J.M. (2011) Cuantificación de recursos hídricos subterráneos en la cuenca alta del Guadiana. Consideraciones respecto a las definiciones de recursos renovables y disponibles. Boletín Geológico y Minero, 122 (1), pp.17–36.