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TESIS DOCTORAL

Análisis económico y ambiental aplicado a la prestación del servicio de aguas en núcleos de población de ámbito rural. El caso de Torre Cardela

Presentada por

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El agua te quiere

Pero tú no a ella,

El agua te cuida

Pero tú la maltratas,

El agua te ayuda y tú no la quieres,

Al agua se le cuida

Jara Alguacil Villegas (2023). Concurso de poesía infantil-juvenil del Valle de Lecrín

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1.1 Marco general de la tesis

1.1.1 Introducción

El agua es un elemento imprescindible para el desarrollo de la vida en todos sus aspectos. Además de su importancia ambiental, es también elemento vertebrador de la sociedad y la economía, siendo determinante en la configuración de nuevos contextos ambientales, políticos, sociales y económicos y por tanto un recurso estratégico de primer orden.

Actualmente, la escasez de agua supone un desafío en multitud de países, ya que, incluso en regiones donde es abundante, puede no satisfacer todas las demandas, debido a la multitud de usos y los costes de oportunidad inherentes, o presentar niveles de contaminación que dificulten su uso. Este problema se puede agravar en el futuro si se consideran los distintos escenarios que se plantean, como, por ejemplo, los vinculados al cambio climático y sus impactos en la disponibilidad de agua (Iglesias et al. 2007; Gosling and Arnell 2016; United Nations-Water 2019; Padrón et al. 2020; UNESCO 2020; Yadu et al. 2021; Vicente-Serrano et al. 2022).

La gestión del agua en cuanto a su distribución y a la efectividad de su uso en los diferentes sectores, es fundamental para abordar la escasez al mismo tiempo que se hace frente a una creciente demanda (Ritchie and Roser 2018; Boretti and Rosa 2019; WWAP 2019)

Entre los principales usos del agua destaca el suministro de agua potable para el consumo humano y el saneamiento, ambos integrados en el Ciclo Urbano del Agua (CUA). La trascendencia del agua en multitud de ámbitos exige un abordaje holístico de la investigación. Esta tesis afronta desde diferentes perspectivas una problemática real, con el objeto de contribuir, de forma pragmática, al estudio de retos concretos en torno a este recurso estratégico.

1.1.2 Problemática de la gestión de los servicios del ciclo urbano del agua en pequeños municipios de ámbito rural

La accesibilidad al abastecimiento y saneamiento de agua es un servicio que tiende a ser universal, al menos en el ámbito de los países de la Organización para la Cooperación y el Desarrollo Económico (OCDE), en la que el servicio se presta bajo titularidad pública o privada, como es el caso de España, sometido a diferentes tipos de normativas.

Al margen de las normativas específicas, se articulan varios principios fundamentales en el ámbito del agua delineados por la Unión Europea, que moldean la prestación del servicio y que resulta pertinente exponer para contextualizar la investigación realizada. El primero de estos principios establece que el acceso al agua se reconoce como un servicio de interés general (European Commission 2003), lo que implica que el sector público tiene la responsabilidad de asegurar su disponibilidad de manera universal a precios asequibles (European Commission 2003; European Commission 2004) y con estándares de calidad predefinidos (European Union 2020). Es crucial destacar también la relevancia de la Directiva Marco del Agua (DMA), que en su artículo 9 especifica que, en los servicios vinculados al agua, debe primar el principio de recuperación de costes, abarcando tanto los costes medioambientales como los relativos a los recursos, además del principio de "quien contamina paga" (European Commission 2000).

El CUA está intrínsecamente influenciado por una serie de factores, entre los que destacan el tamaño y la forma de la población, la topografía del terreno, la calidad y procedencia del agua bruta, las características del vertido, el consumo energético y el marco administrativo y normativo. Al considerar éstos, elementos contextuales que afectan al CUA, es relevante destacar dos factores de especial importancia: el tipo de población y el marco institucional.

El tipo de población se relaciona con variables económicas asociadas al tamaño y la dispersión poblacional, lo cual incide significativamente en la optimización de las economías de escala y, por ende, en la rentabilidad financiera de las operaciones. En poblaciones pequeñas, la gestión del CUA implica costos unitarios de producción más elevados en comparación con poblaciones más grandes y compactas (Nauges y van den Berg 2008; Guerrini et al. 2013).

El marco institucional, a su vez, puede desglosarse en dos aspectos: la condición administrativa del CUA y la competencia municipal en España. Destacar que, en municipios con menos de 20 mil habitantes, las diputaciones provinciales también tienen ciertas competencias y, en última instancia, son responsables del servicio (art. 26 LRBRL 7/1985, de 2 de abril).

A partir de la variable poblacional, es posible clasificar el CUA en dos grupos homogéneos en términos de gestión. Por un lado, están los municipios con una población considerable concentrada en uno o pocos núcleos cercanos geográficamente, donde se pueden aprovechar las economías de escala existentes y operar a bajos costos unitarios de producción. Por otro lado, se encuentran los municipios de menor tamaño y con una dispersión poblacional mayor, donde no es posible aprovechar estas economías de escala, lo que se traduce en costos de producción más elevados que deben ser sufragados por el usuario final del servicio o por el gestor, que en la mayoría de los casos es el propio ayuntamiento.

Excluyendo las grandes poblaciones, la gestión del CUA en pequeños núcleos poblacionales representa, en la mayoría de los casos, un desafío considerable desde los puntos de vista económico, administrativo y técnico, lo que convierte a este ámbito en un campo de investigación que demanda soluciones prácticas y efectivas para diversos problemas relacionados con la gestión del agua. En resumen, el gestor del CUA debe abordar cómo proporcionar un servicio de calidad a un costo razonable para el usuario en un contexto menos favorable.

Una particularidad adicional que atañe a las pequeñas poblaciones es la vinculación existente a los usos del suelo, los cuales son principalmente agrícolas y forestales, y sus efectos en las captaciones de agua. La contaminación por nitratos de origen agrícola de las masas de agua para el abastecimiento es cada vez más común, lo que acaba por demandar la implementación de procesos específicos de potabilización en el CUA, obteniendo como resultado un aumento de los costos y la necesidad de mano de obra cualificada (Bijay-Singh y Craswell 2021).

1.1.3 Caso de estudio: El municipio de Torre Cardela

En España se registran un total de 8.131 municipios, de los cuales 4.986 cuentan con menos de 1.000 habitantes, representando el 61,32% del total, mientras que 7.367 tienen menos de 10.000 habitantes, lo que equivale al 90,60% del total de municipios (Instituto Nacional de Estadística 2023). Por ende, teniendo en cuenta su porcentaje, para la planificación de las políticas relacionadas con el agua y el CUA y para una gestión eficaz del servicio municipal de abastecimiento y saneamiento, es imprescindible comprender las problemáticas asociadas a los pequeños y medianos núcleos de población, así como proponer soluciones adaptadas a la escala de su área de servicio. En la provincia de Granada, donde se ubica el caso de estudio, existen 174 municipios con 282 núcleos de población aparte del principal, lo que hace un total de 456 áreas potenciales de servicio para abastecimiento y saneamiento de agua. Esta realidad es representativa de la situación actual en España, que demanda soluciones prácticas para una gestión efectiva del CUA en los entornos rurales.

Torre Cardela es un ejemplo del contexto de gestión del agua en pequeños municipios y, además, de cómo abordar la problemática de la contaminación por nitratos, ya que se encuentra situado en una zona declarada vulnerable a la contaminación por nitratos procedentes de fuentes agrarias designadas en Andalucía (BOJA 2020).

1.1.3.1 Caracterización del problema de la gestión de los servicios del ciclo urbano del agua en Torre Cardela.

El municipio granadino de Torre Cardela se ajusta al patrón de área de servicio del CUA en el que confluyen factores que hacen complejo prestar un servicio de calidad a un coste asequible para el ciudadano. Este municipio reúne las siguientes características:

1. Tiene un pequeño tamaño poblacional de 709 habitantes, por lo que no es posible aprovechar las economías de escala del sector. Según el artículo 9 de la DMA, los costes del servicio deberían recuperarse a través de la factura del agua.
2. La gestión del servicio se lleva a cabo desde las dependencias municipales donde no hay personal especializado ni en cuestiones técnicas, ni de gestión económico-financiera del servicio. Una cuestión

relevante para la investigación, es que, normalmente, cuando la gestión del servicio se lleva desde el propio Ayuntamiento, no hay una contabilidad separada de ingresos y costes de los servicios del CUA. Esto supone que no se conozca la situación económico-financiera, ni el grado de aplicación del principio de recuperación de costes.

3. Adicionalmente, se dan episodios de contaminación difusa por nitratos. En ocasiones se ha sobrepasado el límite de los 50 mg/l marcado por la Directiva (UE) 2020/2184 del Parlamento Europeo y del Consejo de 16 de diciembre de 2020 relativa a la calidad de las aguas destinadas al consumo humano. Podrían ser los propios vecinos del municipio quienes contaminan el agua debido al uso de fertilizantes en labores agrícolas, en las áreas de influencia de los acuíferos de los que se extrae el agua para abastecimiento domiciliario.

1.1.3.2 Antecedentes de la investigación

El municipio de Torre Cardela tuvo una declaración de No Aptitud del agua para consumo el 9/9/1996, superando ésta los valores de concentración de nitratos recogidos en la Directiva 80/778/CEE del Consejo, de 15 de julio de 1980, relativa a la calidad de las aguas destinadas al consumo humano, y en el anexo C del Real Decreto 1138/1990, de 14 de septiembre, por el que se aprueba la Reglamentación Técnico-Sanitaria para el abastecimiento y control de calidad de las aguas potables de consumo público, el cual daba traslado al ordenamiento jurídico español de la anterior directiva. Derogado posteriormente por el Real Decreto 140/2003, de 7 de febrero, por el que se establecen los criterios sanitarios de la calidad del agua de consumo humano, que mantenía el nivel máximo de 50 mg/l en su anexo I, derivado de la directiva 98/83/CE del Consejo, de 3 de noviembre de 1998, relativa a la calidad de las aguas destinadas al consumo humano. A su vez, este ha sido derogado por el Real Decreto 3/2023, de 10 de enero, por el que se establecen los criterios técnico-sanitarios de la calidad del agua de consumo, su control y suministro donde mantiene el mismo nivel límite para la concentración de nitratos en el agua destinada al abastecimiento humano.

Teniendo en cuenta esta circunstancia, el Decreto 70/2009, de 31 de marzo, por el que se aprueba el Reglamento de Vigilancia Sanitaria y Calidad del Agua de consumo humano de Andalucía, que regula la vigilancia y la calidad del agua en el ámbito territorial de la Comunidad Autónoma de Andalucía, obliga al municipio, en el punto e) del artículo 2, a "*Poner en conocimiento de la población y los agentes económicos afectados los incumplimientos y las situaciones de alerta que den lugar a la pérdida de aptitud para el consumo del agua, y las medidas correctoras previstas, en coordinación con la correspondiente Delegación Provincial de la Consejería competente en materia de salud.*" y, en el punto d) del artículo 3, hace responsable a las personas o entidades públicas o privadas gestoras del abastecimiento de "*Poner en conocimiento de la correspondiente Delegación Provincial de la Consejería competente en materia de salud, de otras personas o entidades públicas o privadas gestoras afectadas y del* *municipio en su caso, los incumplimientos y las situaciones de alerta que se produzcan en el abastecimiento, así como la propuesta de medidas correctoras previstas.*"

Por tanto, desde la fecha de la declaración de No Aptitud del agua para consumo humano, la población no debía consumir el agua directamente, lo que conllevaba el consumo de agua embotellada o uso de filtros domésticos.

La declaración de No Aptitud del agua para consumo humano tuvo lugar de forma continuada durante el periodo comprendido entre el 9/9/1996 hasta el 4/4/2012, momento en el que se instaló una planta de osmosis inversa para la eliminación de nitratos presentes en el agua y situar los niveles de concentración en valores inferiores al límite de 50 mg/l, marcado por la normativa. A partir de este momento, con la puesta en marcha de la planta de osmosis inversa, los valores de concentración de nitratos se situaron debajo del límite establecido y el agua pasó a ser apta para el consumo humano sin necesidad de ningún tipo de tratamiento doméstico.

En términos generales, la función de la planta de osmosis inversa es filtrar el agua que entra directamente del acuífero a través de unas membranas aplicando una elevada presión. En su paso por las membranas concéntricas, el agua discurre hacia el interior donde es recogida por un tubo y va perdiendo todos los elementos en suspensión, incluidos los nitratos. En el caso de Torre Cardela, se optó por una planta de osmosis inversa con 8 membranas ESPA2 con capacidad de producción de agua osmotizada de 8 m^3/h. Para generar el flujo de agua hacia el centro de la membrana se requiere de una elevada presión que es realizada por un grupo de presión. De ahí, que el principal insumo de la planta de osmosis inversa sea la energía eléctrica.

En definitiva, en el municipio de Torre Cardela se dan una serie de factores condicionantes que, conjuntamente, propician un escenario que dificulta la posibilidad de conjugar la prestación de un servicio de calidad con la aplicación del principio de recuperación de costes.

1.1.3.3 El Proyecto LIFE-Ecogranularwater

En mayo de 2015, el municipio de Torre Cardela pidió asistencia técnica a la Diputación de Granada para reducir el coste del tratamiento de potabilización del agua. Con la instalación de la osmosis inversa se había resuelto un problema de salud pública. Sin embargo, la problemática había mutado, inicialmente, hacia un problema de costes del servicio de abastecimiento de agua. Ante esta situación, y teniendo en cuenta las características del municipio, la Diputación de Granada creó y coordinó un grupo de trabajo compuesto por la Universidad de Granada, la Universidad de Aalto (Finlandia), Construcciones Otero S.L. y Gedar S.L. En el marco del proyecto LIFE 16 ENV/ES/196 ECOGRANULARWATER, el grupo de trabajo se encargó del diseño e instalación de una planta demostrativa de potabilización de agua con tecnología granular aeróbica secuencial, destinada principalmente a la eliminación de nitratos (Muñoz-Palazón 2020; Hurtado-Martínez et al. 2021; Rosa-Masegosa et al. 2021; Muñoz-Palazón et al. 2023; Hurtado-Martínez et al. 2024), sustituyendo a la planta de ósmosis inversa. La planta demostrativa, debía cumplir con los parámetros de calidad del agua requeridos por la normativa, a fin de abastecer a la población de Torre Cardela con menor coste operativo y, sostenible desde el punto de vista ambiental.

Para abordar este reto se formó un equipo multidisciplinar en el que participaron investigadores de áreas de Ingeniería Civil, Microbiología, Geología y Economía de las universidades de Granada y de Aalto. El resultado de esta tesis es parte del desarrollo llevado a cabo por el equipo encargado de los aspectos económicos y ambientales de la planta piloto. En este contexto, la tesis estudia la problemática mostrada en el equilibrio financiero del servicio de abastecimiento de agua desde distintos enfoques:

- Solución del problema desde el lado de la demanda a través de una elevación del precio del agua,
- Solución desde el lado de la oferta a través de una innovación técnica que permita potabilizar el agua a un menor coste y con un menor impacto ambiental

1.2 Estructura, temática y objetivos de la investigación

La tesis presentada responde a una estructura por agrupación de publicaciones, cumpliendo con los requisitos establecidos en el Programa de Doctorado de Ciencias Económicas y Empresariales de la Universidad de Granada. Además del capítulo de introducción (capítulo 1) y un último capítulo de resultados y conclusiones (capítulo 5), el resto de la tesis está compuesta por tres capítulos centrales (capítulo 2, 3 y 4), que corresponden a tres artículos publicados en revistas indexadas en Journal Citation Reports:

Capítulo 2. Alguacil-Duarte, F., González-Gómez, F., & del Saz-Salazar, S. (2020). Urban Water Pricing and Private Interests' Lobbying in Small Rural Communities. *Water*, *12*(12), 3509. https://10.3390/w12123509

Capítulo 3. Alguacil-Duarte, F., González-Gómez, F., & Romero-Gámez, M. (2022). Biological nitrate removal from a drinking water supply with an aerobic granular sludge technology: An environmental and economic assessment. *Journal of Cleaner Production*, *367*, 133059. https://10.1016/j.jclepro.2022.133059

Capítulo 4. Alguacil-Duarte, F., González-Gómez, F., & Grigoryan, K. (2024). Proposal of an index to evaluate the 'dewaterization' of the urban water cycle and a practical application. *Sustainable Water Resources Management*, *10*(1), 6. https://10.1007/s40899-023-00984-2

En los capítulos 2 y 3 se muestran los resultados de dos investigaciones aplicadas y en el capítulo 4 se muestra una propuesta metodológica.

El orden de secuenciación responde a tres momentos clave de la investigación. A medida que se avanzaba en el desarrollo del Proyecto LIFE, surgieron preguntas que dieron origen a las distintas investigaciones. De este modo se formuló una pregunta en la fase inicial del proyecto, se dio respuesta a una segunda pregunta en la parte final del proyecto y una tercera pregunta surgió a la finalización del proyecto, cuando se pudo hacer balance de la situación, y se observó que había una carencia metodológica por cubrir, que abre la posibilidad a una prometedora línea de investigación.

1.2.1 Primera pregunta formulada al inicio del proyecto: Además de una reducción de costes operativos, ¿podría proponerse un aumento del precio por el servicio de abastecimiento de agua para conseguir una recuperación de costes en el municipio de Torre Cardela?

Cuando la alcaldesa de Torre Cardela envió un correo de auxilio a Diputación Provincial de Granada, aludía a un problema de elevados costes operativos de la planta de ósmosis inversa que no era posible hacer frente. En los momentos iniciales del proyecto, en el que la planta demostrativa estaba todavía en fase de diseño, el equipo encargado del análisis económico y ambiental no podía hacer ningún tipo de contribución al objetivo inicialmente previsto en la memoria del proyecto. Dicho equipo permanecía a la espera de la puesta en marcha de la planta demostrativa, para poder hacer los análisis de impacto económico y ambiental.

En esa primera fase se optó por una vía de trabajo que viniera a complementar el principal *leit motiv* del proyecto. ¿Y si además del problema de costes al que hacía alusión la alcaldesa, había también un problema de ingresos tarifarios? ¿Una elevación en el precio del agua podría mejorar el equilibrio financiero del servicio de abastecimiento de agua hasta el punto de poder hacer frente a todos los costes operativos del servicio?

Para dar respuesta a esta pregunta, se realizó un análisis de valoración contingente entre la población residente en el municipio de Torre Cardela. Básicamente, se pretendía saber cuánto había que pagar más por el agua para mantener un servicio de calidad, al tiempo que se podía lograr el equilibrio en las cuentas del servicio. No solo se trataba de saber qué porcentaje de población estaba dispuesta a pagar más por el servicio de agua, sino que también se pretendía saber cuánto más se estaba dispuesto a pagar. Con esa información se pudieron hacer estimaciones de la tasa de recuperación de costes a la que se podía optar solo con la subida del precio del agua.

En este contexto, también se pensó que la población residente dedicada a la actividad agraria podía desempeñar un papel clave. Se partía de la hipótesis de que la población agraria podía formar un *lobby* que presentara resistencia a nuevos aumentos en el precio del agua para usos residenciales. Por una parte, porque podían estar trasladando al ámbito domiciliario la renuncia a un aumento del precio para usos agrícolas. Y por otra, porque las personas dedicadas profesionalmente a la actividad agraria, podían tener pequeños huertos regados con agua domiciliaria en torno al núcleo urbano de Torre Cardela. Por tanto, se analizó si había un comportamiento diferencial respecto de la disposición al pago por parte de población residente dedicada a la actividad agraria respecto al resto de la población.

Por todo ello, los objetivos de esta investigación fueron:

- Determinar la disposición a pagar más por el agua entre la población residente.
- Determinar hasta qué punto podía conseguirse el objetivo de recuperación de costes por el servicio de abastecimiento de agua a partir de la disposición al pago por parte de la población.
- Determinar si la población residente dedicada a la agricultura tiene una menor disposición al pago que permita el logro del principio de recuperación de costes.

Son cuestiones que quedan resueltas en el segundo capítulo de esta tesis.

1.2.2 Segunda pregunta planteada, a la que pudo darse respuesta en la fase final del proyecto. ¿La nueva planta demostrativa ofrece una mejor *performance* en términos económicos y de impacto ambiental que la planta de ósmosis inversa?

Una vez construida la planta demostrativa, que conseguía la eliminación de nitratos a partir de la tecnología granular aeróbica secuencial, había que resolver la incógnita de cuáles eran las ventajas económicas y ambientales derivadas del cambio de planta de potabilización de agua. Este era el principal objetivo al que había que atender en el marco del proyecto LIFE de acuerdo con los compromisos adoptados en la memoria presentada. En definitiva, se trataba de determinar si la nueva tecnología era una solución al problema de elevados costes del servicio de potabilización de agua en el municipio de Torre Cardela. Y, por extensión, ateniéndose a las premisas de un proyecto LIFE, analizar el impacto ambiental de la nueva planta.

Para hacer frente a estas cuestiones se optó por realizar un análisis comparativo de rendimiento económico e impacto ambiental entre la planta de ósmosis inversa y la planta demostrativa. Para el análisis comparativo de la *performance* y el impacto de ambas plantas se hicieron dos aproximaciones metodológicas.

En primer lugar, se aplicó un Análisis Coste Efectividad (ACE), que incorporara costes ambientales y de oportunidad, para determinar en términos comparativos, el coste de producir un metro cúbico de agua con una y otra tecnología, utilizando el Coste Anual Equivalente determinado por la Confederación Hidrográfica del Guadalquivir (2022).

En segundo lugar, se aplicó la metodología ambiental del Análisis de Ciclo de Vida (ACV) siguiendo las recomendaciones de la Directiva 2013/179/EU (2013) y las normas ISO 14040 (2006) y 14044 (2006), tanto para el estudio ambiental de la planta demostrativa como de la planta por osmosis inversa, y la comparación de ambas tecnologías. La unidad funcional (UF) fue definida como 1 m^3 de agua potable, siguiendo la regla de categoría de producto (PCR, Product Category Rules) UN CPC

6921 “Water distribution through mains”. Para el cálculo de los diferentes impactos ambientales se utilizó el software SimaPro 9.2.0.2. (Pré Consultants, 2021) y el método ILCD 2011 Midpoint+ v. 1.08/EU27 2010, equal weighting (European Commission, 2012), teniendo en cuenta las fases de clasificación y caracterización que definen la norma ISO 14040 (2006).

El alcance de este estudio se limitó a la producción de agua potable, considerando todos los flujos de entrada y salida de materiales y energía hasta que el agua es tratada para consumo humano. Así, se calcularon las cargas ambientales y económicas de la infraestructura y del tratamiento del agua a lo largo de su ciclo de vida en las dos tecnologías.

Los objetivos de esta investigación fueron:

- Determinar, a partir de un Análisis Coste Efectividad, qué planta (tecnología granular vs. ósmosis inversa) puede producir un m^3 de agua a menor coste y en qué magnitud.
- Determinar, a partir de un Análisis de Ciclo de Vida, qué planta (tecnología granular vs. ósmosis inversa) puede producir un m^3 de agua con un menor impacto ambiental y en qué magnitudes.

Son cuestiones que quedan resueltas en el tercer capítulo de esta tesis.

1.2.3 Tercera pregunta planteada, surgida y solucionada a la final del proyecto. ¿Se puede introducir alguna mejora metodológica en el análisis de impacto ambiental para producir un m^3 de agua?

Al finalizar la segunda de las investigaciones, se hizo una reflexión sobre el tipo de análisis de impacto que vienen realizándose en el ámbito del CUA, incluido el realizado en el marco del proyecto que dio origen a esta tesis. Los procesos del CUA requieren de energía eléctrica y para la generación de este a su vez necesita agua, conocido por el término anglosajón *water-energy nexus*. De esta relación subyace una paradoja a evaluar y en la medida de lo posible cuantificar. Este es el origen de la pregunta principal de la investigación desarrollada en el capítulo 4.

La producción de agua potable para abastecer a la población requiere de una serie de insumos en función del tratamiento. De entre estos factores, hay dos que tienen una especial presencia en la estructura de costes y materiales: la energía eléctrica y el agua bruta. Se produce la paradoja, de no tener en cuenta que para generar la energía eléctrica necesaria para producir un m^3 de agua, también se requiere del uso de agua bruta. Por tanto, no tener en cuenta esta circunstancia hace que se infravalore el impacto ambiental y que sobre los recursos hídricos tiene producir o tratar, según la fase, un m^3 de agua. ¿Qué cantidad de agua bruta se necesita para generar la energía eléctrica necesaria para producir un m^3 de agua potable? Obviamente, el elemento determinante en esta relación es el origen de la energía eléctrica.

El consumo de energía eléctrica por parte del CUA genera un impacto en el medioambiente y en los recursos hídricos. Este aspecto se recoge y se valora en el capítulo 3 de esta tesis, donde queda acreditado. La producción de energía eléctrica impacta sobre los recursos hídricos ya que lleva asociada el uso de agua para su generación (Larsen and Drews 2019, Gerbens-Leenes et al. 2020), de ahí el concepto *water-energy nexus*. Este nexo es el que da lugar a la paradoja: para producir, purificar, depurar o transportar agua se necesita energía eléctrica y para generar electricidad se usa, en sentido amplio, agua.

Ambos sectores, el del CUA y el de la energía, entran en competencia por el agua y cuanta más energía eléctrica se utilice en el CUA más será la presión por el recurso. Además, podría darse el caso de que el consumo de energía eléctrica para potabilizar o depurar un volumen determinado de agua, genere el uso de un volumen aún mayor, derivado del consumo eléctrico del proceso. Esta relación tan directa, pero al mismo tiempo subliminal se ha denominado water-energy-water, concepto acuñado por primera vez, que se tenga conocimiento, en la publicación correspondiente al capítulo 4.

Este término pone de relieve el volumen de agua asociado al consumo eléctrico del CUA que actualmente es una incógnita, en primer lugar, porque es un "consumo fantasma" que no tenía denominación y en segundo lugar porque no existía, que se tenga conocimiento, una metodología que reporte de manera directa y sencilla el volumen de agua usado. Derivado del cálculo del indicador en el capítulo 4 también se introduce un nuevo concepto denominado "dewaterization"

Por tanto, los objetivos de esta investigación fueron:

- Proponer una metodología novedosa para el análisis de impacto ambiental que permite determinar cuánta agua se usa para producir un m^3 de agua potable derivado del consumo eléctrico de este proceso.
- Evaluar el impacto que en los recursos hídricos tiene el consumo de energía eléctrica del CUA.

Estos objetivos quedan resueltos en el cuarto capítulo de la tesis.

1.3 Justificación e interés de la investigación

La importancia tanto del agua en general como del CUA en particular es incuestionable, más aún cuando se barajan escenarios de aumento del consumo y escasez de agua (Ercin and Hoelstra 2014; Kummu et al. 2016; Jaramillo and Nazemi 2018).

La problemática concreta que se aborda en este documento tiene varios factores, como son, altos costes de explotación, exigencias de recuperación de costes, municipios pequeños con gestión directa del CUA, personal poco cualificado, y acuíferos contaminados por nitratos. La tesis presentada se aproxima inicialmente a esta problemática desde los aspectos socioeconómicos y ambientales para evaluar la planta demostrativa realizada en el proyecto LIFE ECOGRANULARWATER (LIFE16

ENV/ES/196). Se trata por tanto de una aportación inédita desde el plano científico a todos los grupos de interés ligados al sector del agua para avanzar en la efectividad del servicio.

El gran número de municipios con características similares al de Torre Cardela, en cuanto a la gestión del CUA en España, dotan a esta investigación de un gran potencial de replicabilidad y transferencia de los resultados y soluciones propuestas a lo largo de esta tesis, sirviendo de modelo de aplicación directa o para abrir nuevas líneas de investigación.

Respecto a la contaminación por nitratos, a nivel representativo, hay que destacar que según el Ministerio para la Transición Ecológica y el Reto Demográfico (MTERD, 2020), el 17,51% de los puntos de muestreo de las captaciones subterráneas incluidos en el Sistema de Información Nacional de Agua de Consumo (SINAC), superarían el valor medio anual de 50 mg/l.

Según el informe de la Comisión al Consejo y al Parlamento Europeo sobre la implementación de la Directiva 91/676/CEE (COM 2021 1000 final), se observa una problemática recurrente a nivel europeo relacionada con la contaminación por nitratos en las aguas. Se destaca que un número significativo de estaciones de monitoreo de agua registran concentraciones superiores al límite máximo establecido de 50 mg de nitratos por litro. Esta situación se manifiesta de manera particular en países como Malta, Alemania, Luxemburgo, España, Portugal y en la región de Flandes en Bélgica.

El informe también subraya que, a pesar de haber transcurrido tres décadas desde la adopción de la Directiva 91/676/CEE, varios países europeos continúan enfrentando problemas de calidad del agua en todo su territorio. Estos países incluyen a la región de Flandes en Bélgica, la República Checa, Dinamarca, Alemania, Finlandia, Hungría, Letonia, Luxemburgo, Malta, los Países Bajos, Polonia y España. Esta situación resalta la persistencia y la magnitud del desafío que representa la gestión de la contaminación por nitratos provenientes de la actividad agrícola en Europa. Esta problemática no solo es europea, sino que escala a nivel mundial ligada principalmente al uso de fertilizantes (Craswell 2021) y sigue en continuo aumento (Abascal et al. 2022).

Al margen de los expuesto en el párrafo anterior, una posible reconsideración a la baja de los umbrales aceptables para la concentración de nitratos en el agua destinada a abastecimiento podría aumentar la necesidad del tratamiento del agua en España y toda la Unión Europea. Como ha ocurrido en el Real Decreto 47/2022, de 18 de enero, sobre protección de las aguas contra la contaminación difusa producida por los nitratos procedentes de fuentes agrarias, donde se considera que las aguas afectadas por nitratos son aquellas que tienen una concentración de 37,5 mg/l. Actualmente, el límite establecido para la concentración máxima de nitratos es de 50 mg/l, sin embargo, en Estados Unidos la Agencia de Protección Ambiental considera que la concentración máxima admisible para agua destinada a abastecimiento es de 10 mg/l[1]. En un escenario de bajada de los niveles exigidos de concentración de nitratos, se debería disponer de tecnologías que pudieran dar respuesta a las necesidades establecidas en las normativas. Sin tecnología que haga viable esta bajada de niveles no sería realista legislar en este sentido, de ahí la importancia de la implantación de la planta demostrativa y sus resultados.

Estos dos efectos, el incremento de la contaminación por nitratos y la disminución del límite de concentración establecido por la normativa, de forma aislada o conjunta, muestran de forma inequívoca que la tendencia relacionada con este tipo de problemática va a continuar en aumento.

El análisis económico y ambiental de las nuevas tecnologías desarrolladas es crucial para su validación. En ocasiones, existen nuevas tecnologías que presentan una ventaja económica y se implantan exclusivamente bajo ese criterio, sin tener en consideración las externalidades negativas, que por ejemplo en el CUA podrían agudizar más aún la problemática de escasez de agua y a nivel ambiental, generar un incremento de las emisiones a la atmosfera, entre ellas las de gases de efecto invernadero. La implantación de una tecnología nueva puede ser aparentemente más beneficiosa desde el punto de vista económico y ambiental, sin embargo, lo único que se está haciendo es trasladar los impactos cambiando su naturaleza, su escala temporal o su localización geográfica, pasando estos desapercibidos. La metodología del Análisis de Ciclo de Vida permite detectar y evitar estos cambios, obteniendo una valoración holística del impacto ambiental que junto con el impacto económico mostrará el verdadero desempeño de una instalación.

A raíz de lo expuesto, se puede apreciar que se trata de una investigación eminentemente aplicada, y de ahí deriva la "Mención Industrial" a la que se opta con esta tesis. En esta tesis se evalúa la problemática en torno a la planta demostrativa instalada en el marco del proyecto LIFE ECOGRANULARWATER (LIFE16 ENV/ES/196) para avalar científicamente la idoneidad de la propuesta, desde una perspectiva multidisciplinar y holística, para el municipio de Torre Cardela. Con este cometido, se da solución a esta problemática concreta, produciendo un avance tangible en la gestión del agua. Los análisis y resultados obtenidos en el municipio de Torre Cardela desarrollados en los capítulos 2 y 3 son perfectamente replicable a otros municipios, de hecho, este era uno de los objetivos de la investigación. La metodología desarrollada en el capítulo 4 puede ser exportada a todo tipo de municipios o instalaciones del CUA, como se muestra en la publicación.

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[1] United States Environmental Protection Agency. Estimated Nitrate Concentrations in Groundwater Used for Drinking https://www.epa.gov/nutrientpollution/estimated-nitrate-concentrations-groundwater-used-drinking

A modo de síntesis genérica de lo anteriormente expuesto, los factores que sostienen el interés y justifican la investigación son los siguientes:

1. Importancia del recurso hídrico:

El agua es un recurso escaso, vital y estratégico, lo que subraya la trascendencia de investigar su uso y gestión.

2. Problemática de gestión del CUA en pequeños núcleos de población:

La problemática del CUA en comunidades pequeñas es de relevancia estatal y europea, lo que justifica la necesidad de abordarla mediante investigación desde diferentes perspectivas.

3. Contaminación del agua por nitratos:

La contaminación del agua de consumo humano a causa de los nitratos, no solo afecta a nivel local y europeo, sino también global, de ahí la necesidad de profundizar en medidas para resolver los problemas derivados de sus efectos.

4. Evaluación multidisciplinar de nuevas tecnologías:

Se requiere una evaluación multidisciplinar de las nuevas tecnologías para validar su desempeño ambiental y económico, especialmente en relación a los recursos hídricos, que son la materia prima principal del CUA.

5. Replicabilidad y transferibilidad de aplicaciones prácticas y conclusiones:

Las aplicaciones prácticas y conclusiones derivadas de la investigación tienen un alto potencial de replicación y transferencia, lo que puede contribuir a la solución de problemáticas similares en otras regiones y contextos, justificando así la relevancia y la utilidad de la investigación.

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Anexo I: Imágenes

Imagen 1: Planta piloto de tecnología granular aeróbica

Imagen 2: Planta demostrativa con tecnología granular aeróbica secuencial

Imagen 3: Detalle de los gránulos

Imagen 4: Planta de osmosis inversa de Torre Cardela

Imagen 5: Folleto proyecto LIFE Ecogranularwater pag.1

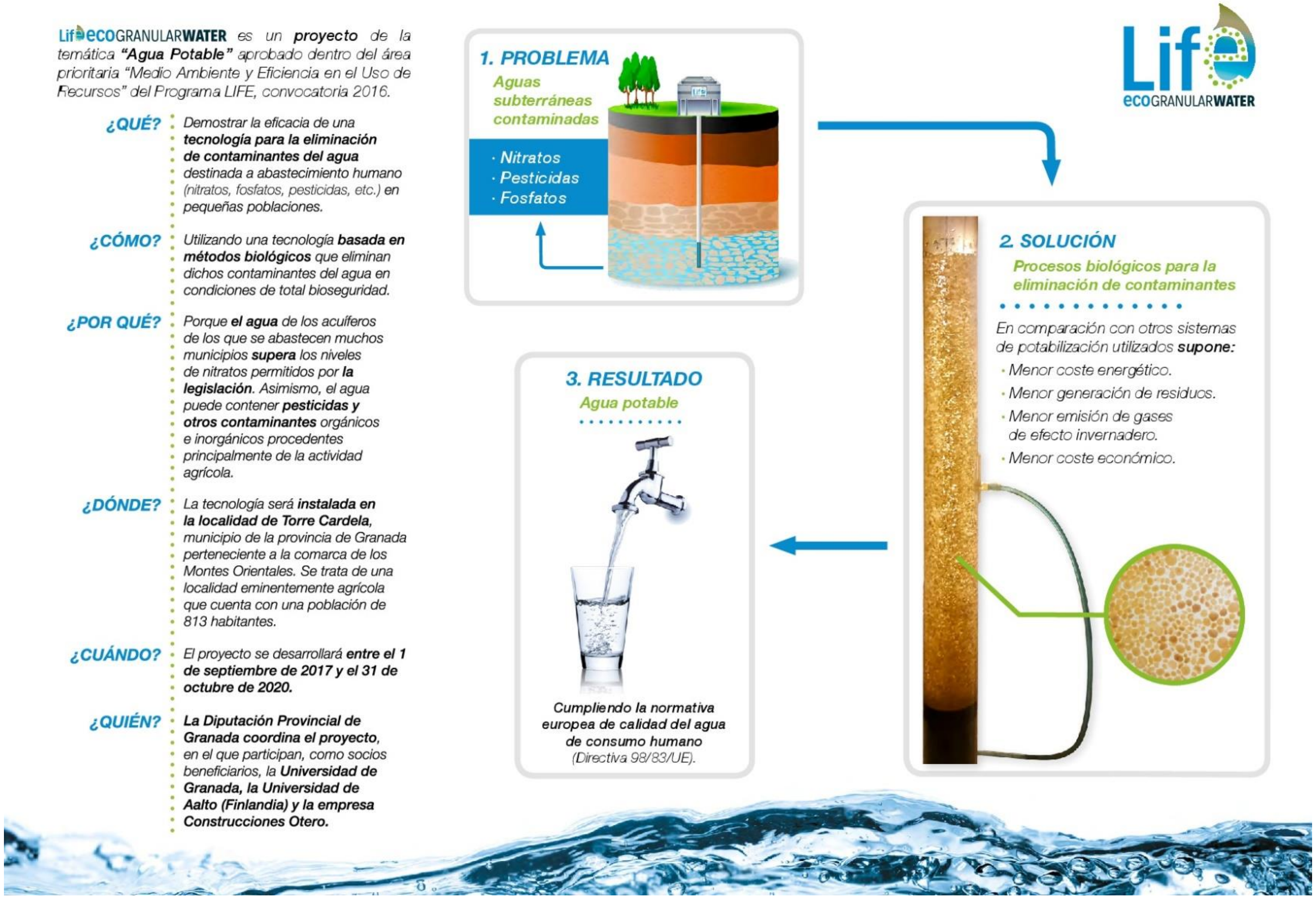
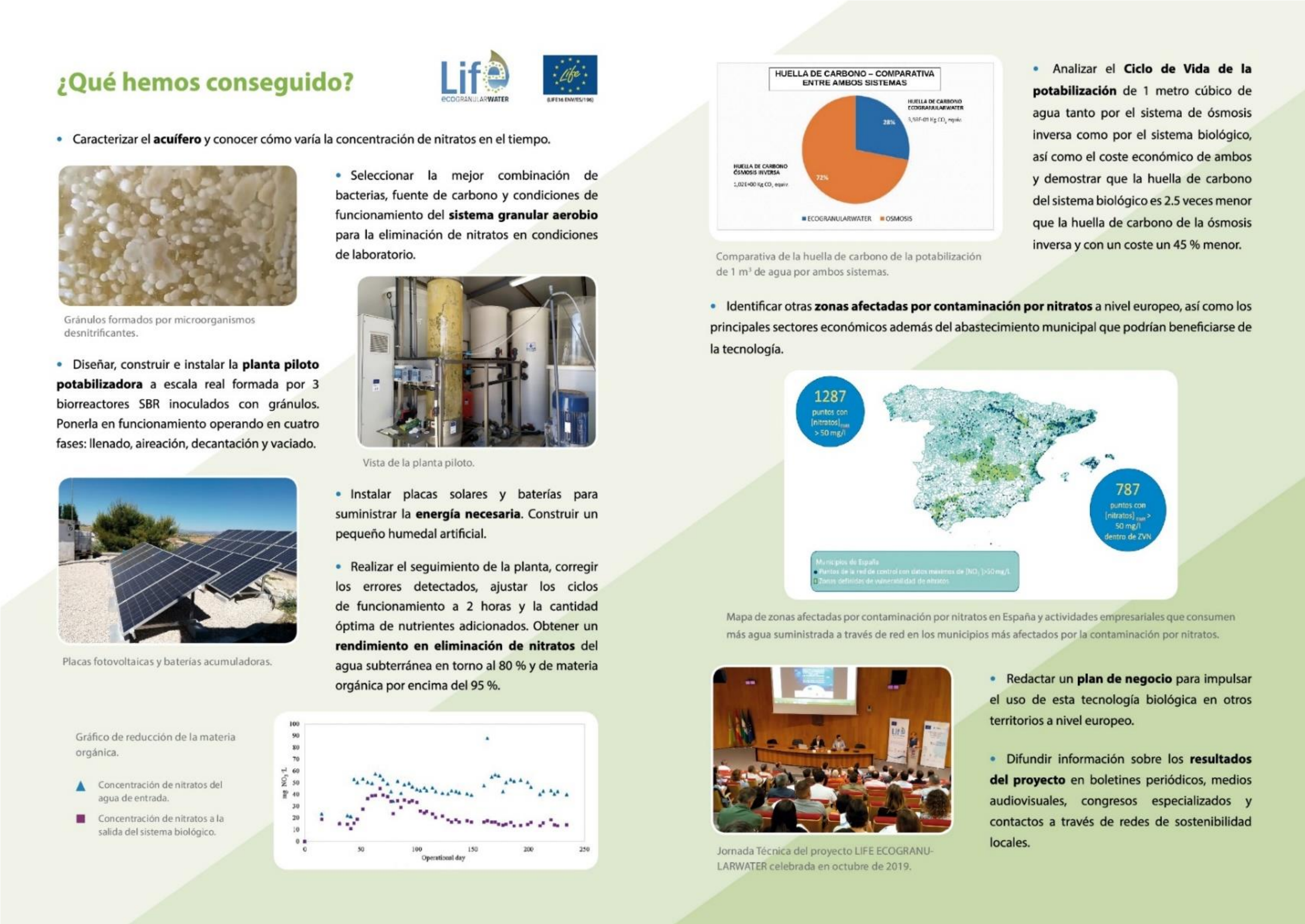


Imagen 6: Folleto proyecto LIFE Ecogranularwater pag. 2

Imagen 7: Folleto jornada final del proyecto LIFE Ecogranularwater pag. 1

Imagen 8: Folleto jornada final del proyecto LIFE Ecogranularwater pag. 2



¿Qué hemos conseguido?

- Caracterizar el **acuífero** y conocer cómo varía la concentración de nitratos en el tiempo.

Gránulos formados por microorganismos desnitrificantes.

- Seleccionar la mejor combinación de bacterias, fuente de carbono y condiciones de funcionamiento del **sistema granular aerobio** para la eliminación de nitratos en condiciones de laboratorio.

Vista de la planta piloto.

- Diseñar, construir e instalar la **planta piloto potabilizadora** a escala real formada por 3 biorreactores SBR inoculados con gránulos. Ponerla en funcionamiento operando en cuatro fases: llenado, aireación, decantación y vaciado.

Placas fotovoltaicas y baterías acumuladoras.

- Instalar placas solares y baterías para suministrar la **energía necesaria**. Construir un pequeño humedal artificial.

- Realizar el seguimiento de la planta, corregir los errores detectados, ajustar los ciclos de funcionamiento a 2 horas y la cantidad óptima de nutrientes adicionados. Obtener un **rendimiento en eliminación de nitratos** del agua subterránea en torno al 80 % y de materia orgánica por encima del 95 %.

Gráfico de reducción de la materia orgánica.

▲ Concentración de nitratos del agua de entrada.

■ Concentración de nitratos a la salida del sistema biológico.

Comparativa de la huella de carbono de la potabilización de 1 m³ de agua por ambos sistemas.

- Analizar el **Ciclo de Vida de la potabilización** de 1 metro cúbico de agua tanto por el sistema de ósmosis inversa como por el sistema biológico, así como el coste económico de ambos y demostrar que la huella de carbono del sistema biológico es 2.5 veces menor que la huella de carbono de la ósmosis inversa y con un coste un 45 % menor.

- Identificar otras **zonas afectadas por contaminación por nitratos** a nivel europeo, así como los principales sectores económicos además del abastecimiento municipal que podrían beneficiarse de la tecnología.

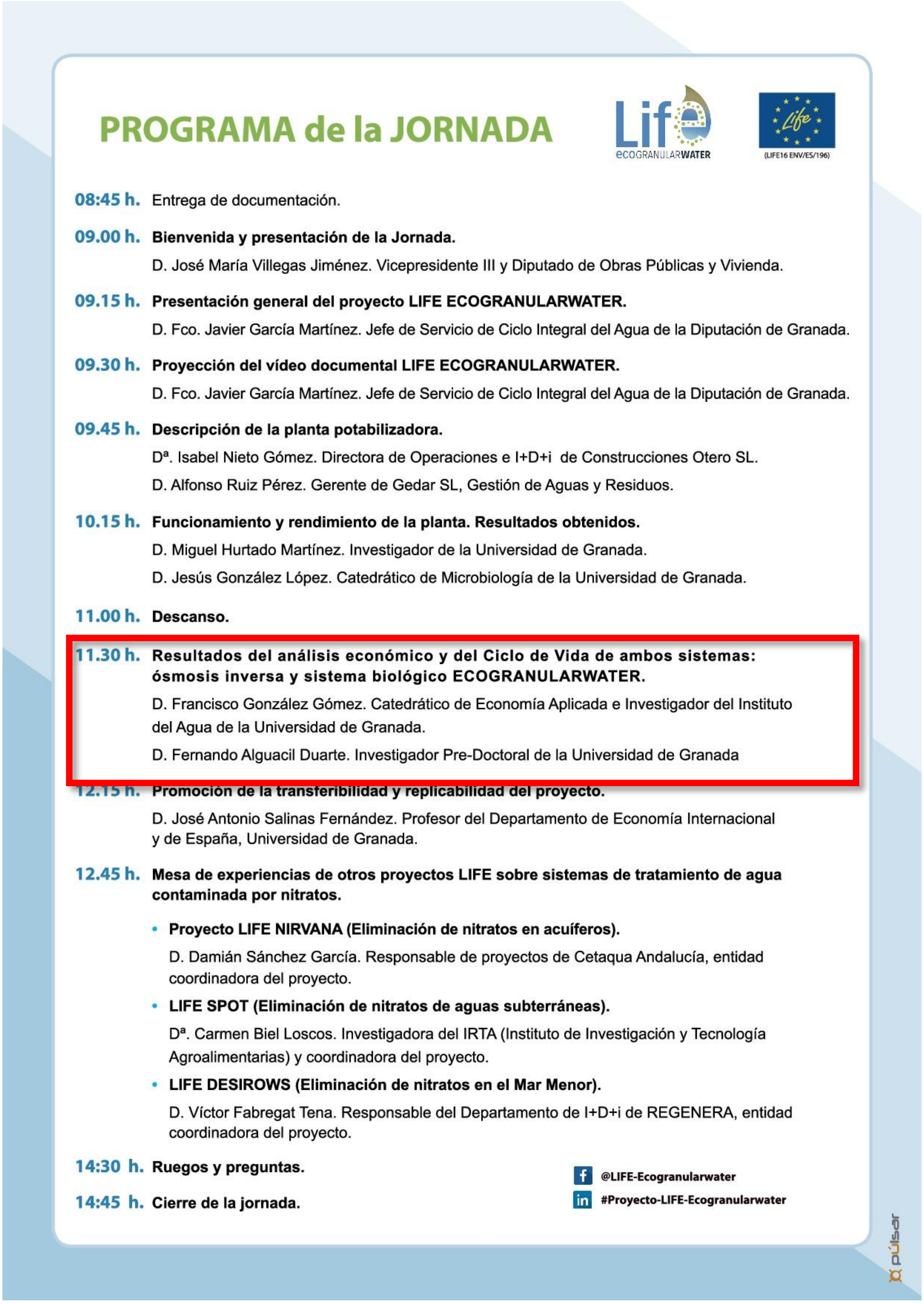
Mapa de zonas afectadas por contaminación por nitratos en España y actividades empresariales que consumen más agua suministrada a través de red en los municipios más afectados por la contaminación por nitratos.

Jornada Técnica del proyecto LIFE ECOGRANULARWATER celebrada en octubre de 2019.

- Redactar un **plan de negocio** para impulsar el uso de esta tecnología biológica en otros territorios a nivel europeo.

- Difundir información sobre los **resultados del proyecto** en boletines periódicos, medios audiovisuales, congresos especializados y contactos a través de redes de sostenibilidad locales.

Imagen 9: Folleto jornada final del proyecto LIFE Ecogranularwater pag. 3



PROGRAMA de la JORNADA

Life ecogranularwater

(LIFE16 ENV/ES/196)

08:45 h. Entrega de documentación.

09.00 h. **Bienvenida y presentación de la Jornada.**
D. José María Villegas Jiménez. Vicepresidente III y Diputado de Obras Públicas y Vivienda.

09.15 h. **Presentación general del proyecto LIFE ECOGRANULARWATER.**
D. Fco. Javier García Martínez. Jefe de Servicio de Ciclo Integral del Agua de la Diputación de Granada.

09.30 h. **Proyección del vídeo documental LIFE ECOGRANULARWATER.**
D. Fco. Javier García Martínez. Jefe de Servicio de Ciclo Integral del Agua de la Diputación de Granada.

09.45 h. **Descripción de la planta potabilizadora.**
Dª. Isabel Nieto Gómez. Directora de Operaciones e I+D+i de Construcciones Otero SL.
D. Alfonso Ruiz Pérez. Gerente de Gedar SL, Gestión de Aguas y Residuos.

10.15 h. **Funcionamiento y rendimiento de la planta. Resultados obtenidos.**
D. Miguel Hurtado Martínez. Investigador de la Universidad de Granada.
D. Jesús González López. Catedrático de Microbiología de la Universidad de Granada.

11.00 h. **Descanso.**

11.30 h. **Resultados del análisis económico y del Ciclo de Vida de ambos sistemas: ósmosis inversa y sistema biológico ECOGRANULARWATER.**
D. Francisco González Gómez. Catedrático de Economía Aplicada e Investigador del Instituto del Agua de la Universidad de Granada.
D. Fernando Alguacil Duarte. Investigador Pre-Doctoral de la Universidad de Granada

12.15 h. **Promoción de la transferibilidad y replicabilidad del proyecto.**
D. José Antonio Salinas Fernández. Profesor del Departamento de Economía Internacional y de España, Universidad de Granada.

12.45 h. **Mesa de experiencias de otros proyectos LIFE sobre sistemas de tratamiento de agua contaminada por nitratos.**

- **Proyecto LIFE NIRVANA (Eliminación de nitratos en acuíferos).**
  D. Damián Sánchez García. Responsable de proyectos de Cetaqua Andalucía, entidad coordinadora del proyecto.
- **LIFE SPOT (Eliminación de nitratos de aguas subterráneas).**
  Dª. Carmen Biel Loscos. Investigadora del IRTA (Instituto de Investigación y Tecnología Agroalimentarias) y coordinadora del proyecto.
- **LIFE DESIROWS (Eliminación de nitratos en el Mar Menor).**
  D. Víctor Fabregat Tena. Responsable del Departamento de I+D+i de REGENERA, entidad coordinadora del proyecto.

14:30 h. **Ruegos y preguntas.**

14:45 h. **Cierre de la jornada.**

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Imagen 10: Folleto jornada final del proyecto LIFE Ecogranularwater pag. 4

CAPITULO 2. Does the agricultural population constitute a lobby against raising the price of water for residential uses? Case study in Southern Spain

A preliminary version of this chapter was disseminated as a working paper for the LIFE ECOGRANULARWATER (LIFE16 ENV/ES/196) project. The key final results have been published in the paper Does the agricultural population constitute a lobby against raising the price of water for residential uses? Case study in Southern Spain. *Water 12* (12), 3509. https://doi.org/10.3390/w12123509

2.1 Abstract

It is difficult for small municipalities to ensure their urban water cycle complies with the principle of cost recovery established in the European Union (EU) Water Framework Directive. Unlike more populous municipalities, small municipalities face higher average production costs. However, at least in Spain, the price of water is, on average, lower in small municipalities. We question whether the low price of water in rural areas is due, at least in part, to people linked to agriculture, i.e., do farmers constitute a special interest group that hinders increases in the price of water? The main hypothesis was tested with data taken from Torre-Cardela, a municipality in southern Spain with close to 800 inhabitants. In the research a contingent valuation analysis was carried out to analyze respondents' willingness to pay in the event of a hypothetical increase in the price of water to help cover the service costs. Contrary to expectations, the study yields no evidence that the agricultural population is more resistant to price rises than the rest of the citizens surveyed. In fact, results show that people involved in the agricultural sector would be willing to accept a hypothetical increase in water tariffs in between 15% and 25% over the current tariff, while for the rest of the population this same increase would be lower (in between 9% and 20%).

2.2 Keywords

Water price; rural areas; contingent valuation; willingness to pay; lobby; water framework directive; comparative analysis

2.3 Introduction

There is a growing concern about resource management, in the face of the threats of climate change and increased water demand for different uses (Duan et al., 2020a; Duan and Takara, 2020) The EU Water Framework Directive (WFD) aims to ensure efficient and sustainable management of water resources. To achieve this, it calls for the application of economic principles and instruments (Unnerstall, 2007). According to Article 9 of the WFD, Member States should place more emphasis on the principle of cost recovery and water-pricing policy to achieve better management of water resources (European Environmental Agency, 2013).

In terms of residential water supply, the application of the cost recovery principle means that users must finance the total costs of the urban water cycle. The population must therefore cover the full cost of having a high quality water service and to contribute to the conservation of the environment. Article 9.4 of the WFD states that Member States shall not violate the principle of cost recovery. However, the same article also contemplates the possibility of non-compliance, provided that neither the purposes nor the achievement of the objectives of the WFD are compromised. It is thus acknowledged that the application of efficient water prices can have a negative impact on social welfare. The strict implementation of the cost recovery principle would compromise affordability, mainly in low-income groups and rural communities (Reynaud, 2016).

Specifically, one of the reasons for non-compliance with the complete recovery of costs in the urban sphere is the principle of equity. It is generally accepted that complete cost recovery is not possible in small municipalities, mainly due to the higher average costs. Such municipalities cannot take advantage of the economies of scale and density existing in the industry (González-Gómez and García-Rubio, 2008; Walter et al., 2009; Witte and Marques, 2010; Cetrulo et al., 2019). A case in point is that of Spain since it is common for Spanish river basin plans to include justifications for breaching the principle of cost recovery of the urban water cycle in rural areas. Such justifications include territorial cohesion in depressed areas, social reasons or the inability to exploit economies of scale.

Recently, the Government of Spain has expressed concern about the situation regarding water services in rural areas, particularly in terms of the quality of service provision and the financial deficit associated with the urban water cycle. In meetings promoted by the Ministry for Ecological Transition throughout 2019, prior to the drafting of a Green Paper on Water Governance in Spain, the management of the complete urban water cycle in small municipalities was denoted a core topic. It is a problem on a large scale since 77% of the 8116 Spanish municipalities have fewer than 3000 inhabitants (INE, 2019). In these meetings, the need to create optimal areas of exploitation was highlighted in order to make better use of economies of scale and reduce unit production costs.

A complementary measure could be to increase revenue by raising the price of water. The price of water is comparatively low in Spain (IWA, 2016; Global Water Intelligence, 2018). In addition, within Spain, water prices are, on average, lower in small municipalities (García-Valiñas et al., 2013), which means that less revenue is generated per cubic meter of water consumed in the home. The low price of water may be due, at least in part, to the fact that complete urban water cycle services are not provided in these municipalities, or that the provision of these services is of lower quality. However, it may also be due to greater difficulty in updating water prices in rural areas. In fact, it is not uncommon to find current rates that were approved in the 1990s and still quoted in pesetas, the official currency before the introduction of the euro.

In the absence of national regulation and monitoring agencies, small rural municipalities may still be setting excessively low water prices. Within the framework of Public Choice Theory (Mueller 2003), there are different explanations for why the prices set may be lower than socially desirable, mainly in municipalities where the management of the service is handled by the local council itself. In this research, we focus on one explanation: the existence of lobbyists. Specifically, we question whether the population linked to agricultural activity forms a pressure group that helps keep water prices low in rural areas. If this is indeed the case, for the sake of economic efficiency and cost recovery public authorities should implement educational and awareness-raising training campaigns targeted at this group, in order to reduce their resistance to possible increases in the price of water for residential uses.

To determine whether the population engaged in agricultural activity helps keep water prices low, the methodological approach used is contingent valuation analysis (Carson, 2012). The declared preferences will allow us to determine whether people linked to agriculture are more reluctant to pay to finance the public water service. The existence of such a relationship would be an indication that this group constitutes a lobby that hinders the setting of water prices at levels that contribute to cost recovery. The research has been carried out with data from 468 interviews conducted in Torre-Cardela, a municipality in southern Spain that runs large deficits in its urban water cycle accounts. Contrary to expectations, the findings show that the agricultural population is not more reluctant than the population as a whole to pay more to help ensure the recovery of service costs. In fact, those people in the agricultural population who were willing to pay more for water stated that they would be prepared to assume a greater increase in the amount of the water bill than the rest of the population.

The paper is structured as follows. After this introduction, the second section presents some theories to explain why water prices for residential uses may be abnormally low in rural areas. The third section explains the empirical strategy used in the investigation. The fourth section shows the main results obtained. Finally, the paper concludes with a discussion, conclusions and recommendations section. Why might prices be abnormally low in rural areas? An explanation within the framework of Public Choice Theory

2.4 Why Might Prices Be Abnormally Low in Rural Areas? An Explanation within the Framework of Public Choice Theory

The management of the urban water cycle in Spain falls under the jurisdiction of the municipalities. However, the local government may assign the management of the service to a public company, a private company or a public-private joint venture. In small municipalities it is relatively more common for the municipality to directly manage the urban water cycle (González-Gómez et al., 2014; AEAS, 2018; González-Gómez et al., 2011). Indeed, the management is carried out by the town council in approximately 75% of the municipalities with fewer than 2000 inhabitants.[2]

Additionally, pricing is the responsibility of the local government. The local government decides when the prices are reviewed, the design of the tariff and the price level. In Spain there is no regulation that establishes uniform pricing criteria. Nor is

[2] Own estimates based on data extracted from the Survey of Infrastructures and Local Equipment: https://ssweb.seap.minhap.es/descargas-eiel/ (accessed on 16 November 2020).

there a regulatory authority in Spain such as the Water Services Regulation Authority (Ofwat) in England and Wales or a water observatory such as ONEMA (L'Office national de l'eau et des milieux aquatique) in France. In Spain, only the approval of the regional administration is necessary for the new price proposals requested by local governments. In practice, this approval is no more than an administrative procedure (García-Rubio et al., 2015; Picazo-Tadeo et al, 2020).

Therefore, as regards the water pricing decision, there is an excessive concentration of competencies in local governments, as well as a practical absence of control over the decision taken in the municipalities. This concentration of competencies can lead to low-priced water in rural areas, where the municipality normally handles the management. According to García-Valiñas et al (2013) and González-Gómez and García-Rubio (2018), when the management is carried out by the municipality, prices are lower than when the service is outsourced. This is probably an ideal scenario for some public sector failures; it would explain, at least in part, why prices in rural areas are lower than would be socially desirable.

According to Public Choice Theory, consumers of a good, in this case water users, aspire to make the price as low as possible (Stigler, 1971). In rural areas, the agricultural population can act as a lobby influencing local governments' water pricing decision. According to Becker (1985), the size of the group is an important factor in the pressure put on local politicians. In small rural areas, a high percentage of the population is engaged in agricultural activity, so it could be an influential lobby.

Traditions and customs in Spain may have contributed to the agricultural population's role as a pressure group (Guillet, 1997). Historically, the use of water has been free for agriculture. Currently, water for agricultural uses is heavily subsidized (Calatrava et al., 2015; Toan, 2016). Moreover, until a few decades ago, water for residential uses was collected directly from rivers and wells (González-Gómez et al., 2012; Señan, 2007). Hence, the population in rural areas is used to paying little or nothing for water. In addition, the fact that farmers can easily compare what they pay for water for agricultural uses and what they pay for residential uses can also contribute to the population's greater resistance to paying more for water for residential uses.

This resistance to paying more for water in rural areas can also be reinforced by the political opportunism of the current local government. According to Nordhaus (1975), political agents act rationally when they deliberately manipulate the economy to achieve electoral advantages. Although the literature analyzing political opportunism focuses on fiscal instruments, other instruments may also be used, such as the levels of regulated prices (Agénor and Asilis, 1997; Özatay, 2007; Dubois, 2016). Regarding the public water service, the local politician would ideally seek to keep a paradoxical promise to set low prices and improve infrastructure at the same time (Felgendreher and Lehmann, 2016). Water price increases are unpopular, which can influence local government decision-making (Picazo-Tadeo et al, 2020; Klien, 2015). Predictably, this political interference in the pricing of water is greater when the water service is managed by the city council (Klien, 2014a). In addition, there may be more interference in small population centers, since these institutions are less developed and media scrutiny is weaker than at national or regional level, or even than in larger municipalities (Mandon and Cazals, 2019). Additionally, it could be expected that this effect would be more noticeable in small municipalities where there is closer contact between politicians and the population, and the relationships between them are shaped by family and friendship ties.

In addition, the expected effect of lobbying on the price of water, which will also be facilitated by political opportunism, can be maintained over time if the service costs are permanently financed through grants from other administrations. Generous subsidies compensate for diseconomies of scale and tend to perpetuate situations of inefficiency (Sørensen, 2006). In the Spanish case, provincial councils, regional governments and the national government provide complementary subsidies for the activity of small rural municipalities. According to Agency Theory, in the context analyzed, the local authority has more information on costs and production processes than the subsidizer, which could be used to the advantage of the former. Additionally, as Peacock [39] maintains, the perception of subsidies allows the cost structure to move upwards. Grants could eliminate incentives for efficient management, which would aggravate the financial balance issues affecting the service.

Therefore, in the described model, both the lobby composed of the agricultural population and the political agent seeking re-election will have an interest in keeping water prices low. Thus, the population of medium and large municipalities could be subsidizing water in rural areas, without such subsidies being justified by the size of the municipality or, ultimately, socially desirable.

In this context, we focus our attention on one of the explanations proposed. We question whether the agricultural population really is reluctant to pay more for water, which would suggest that the group acts as a lobby to keep the price of water low. In part, this would explain the low price of water in small rural population centers. However, the relationship may not be so clear cut; it is possible that the traditional vision of water users in the agricultural field is outdated, or that they do not form a well-organized group. In light of the above, we test the following hypothesis:

H_1: The agricultural population is more reluctant to pay more for water for residential uses

If the hypothesis is confirmed, it would suggest that there is need to apply measures in the rural environment that allow the price of water to rise, while minimizing public opposition. If the hypothesis is rejected, it would suggest that there is a need for a more in-depth examination of the relationship between political opportunism and low water prices, as well as the possibility of incorporating measures that reduce local politicians' power to set the prices of water for residential uses in rural areas.

2.5 Empirical Strategy

2.5.1 Case study

In order to test the main hypothesis, research was carried out in Torre-Cardela, a small municipality in southern Spain. Torre-Cardela has 782 inhabitants and its main economic activity is based on agriculture. It is located in the Guadalquivir River Basin, a highly water-stressed area. Water for urban uses is extracted from wells and has a high nitrate concentration due to the use of fertilizers in agriculture. Until 2012, when a water treatment plant using reverse osmosis technology was built, water from the public supply was not drinkable.

As mentioned above, the management of the urban water service is carried out by the municipality. According to the 2017 local government budgets, the tariff and non-tariff revenues only allow 40% of the total costs of the service to be recovered. The average price of water for residential uses is 1.56 euros/m3; this price includes supply, sewerage and treatment. The price is low in comparison with the published data for the whole of Spain, which also includes large and medium-sized municipalities. According to INE (2018), the price of water for residential uses in 2016 was 1.95 euros/m3. According to the Spanish Association of Water Supply and Sanitation (AEAS, 2018), in 2016 the price of a cubic meter of water was 2.34 euros.

The question we ask ourselves is whether there is any relationship between the agricultural population and resistance to paying more for water for residential uses. This association would explain, at least in part, why rural areas show greater difficulty in complying with the principle of cost recovery. This explanation would be complementary to the fact that small municipalities cannot take advantage of the economies of scale of the industry.

2.5.2 Methodology

2.5.2.1 Contingent Valuation, protest responses and censoring

Traditionally, economists have addressed the valuation of water resources using stated preference methods, among which the Contingent Valuation Method (CVM) stands out (see, for example, [Byambadorj and Lee, 2019; Gschwandtner et al., 2020; Rodriguez-Tapia et al., 2017]). CVM is a survey-based approach used to place a monetary value on public and environmental goods that are not commonly bought and sold in the marketplace (Carson, 2000). However, the absence of a market does not necessarily imply the absence of value. Hence, in order to determine the value of such goods, a representative sample of the population is usually asked about their willingness to pay (WTP) for an improvement in the quality or quantity of this public good, and less often about their willingness to accept (WTA) an amount of compensation for a reduction in the supply or quality of the good (Carson et al. 2003). Whether WTP or

WTA is the appropriate measure depends on how property rights to the environmental good are allocated (Carson, et al., 2001).

Nevertheless, despite its prevalence over recent decades in the non-market valuation context, CVM is not without its flaws. For example, Johnson and Whitehead (2000) point out that for many policy issues CVM surveys generate a considerable number of zero responses. True zeros are responses in line with expressions of economic preference whereby the respondent rejects the amount offered either because of income constraints or just because the good that is being valued is of no value to the respondent. On the other hand, protesters are respondents who, although their true WTP is greater than zero, reject the amount offered because they object to some aspects of the valuation scenario, such as lack of information or the payment vehicle, or even because of ethical beliefs (Jorgensen and Syme 2000; Meyerhoff and Liebe, 2006).

It can be deduced from the above that the definition and treatment of protest zero responses is a major concern in the CVM literature. Indeed, since the pioneering work by Lindsey (1994), it seems that no clear consensus has been established about the most appropriate procedure for differentiating between true zero WTP and protest responses and, crucially, about how to treat protest responses in the subsequent analysis. Regarding the treatment of protest responses, it is common practice in the CVM literature to delete them from the sample on the grounds that they are illegitimate choices, i.e., they do not represent true economic values (Jongersen et al., 1999). In this respect, Szabó (2011) claims that, when valuing unfamiliar and complex environmental goods, the use of deliberative techniques from political sciences significantly reduces the rate of protest responses while increasing the validity of the stated preference methods.

Censoring protest responses may not be the correct procedure since it can lead to a non-response bias or selection bias (Soliño, 2010), i.e., when the preference structure of the group of protesters systematically differs from the rest of the sample. If this is the case, these individuals are self-selecting when protesting. Accordingly, the estimated WTP values will not be representative of the entire population and it will be not possible to extrapolate them and make accurate inferences about the target population based on the sample (Bonnichsen and Olsen, 2016).

2.5.2.2 Econometric specifications

First, we want to test whether or not people involved in agricultural activities are more likely to protest. This is done by fitting a maximum-likelihood bivariate probit model with sample selection. Accordingly, two binary dependent variables are modeled jointly as a function of some explanatory variables using the “heckprob” command in Stata 15 software (StataCorp, 2017). The variable y_j^{probit} takes the value of 1 if the respondent protested and 0 otherwise, while $Y_j^{selection}$ takes the value of 1 if the respondent is somehow involved in the agricultural industry (the respondent owns agricultural land or works in the agricultural industry, or both), and 0 otherwise.

Therefore, the two binary outcomes may be correlated, i.e. the probability of protesting may be correlated with involvement in the agricultural industry.

The probit model with sample selection (Van de Ven and Van Pragg 1981) assumes that there is an underlying relationship, known as a latent equation:

$$y_j^* = x_j\beta + \varepsilon_{1,j} \quad (1)$$

where x_j is a vector of variables that explain the protest decision and β are the parameters to be estimated. But we observe only a binary outcome (probit equation) such that:

$$y_j^{probit} = y_j^* > 0 \quad (2)$$

The dependent variable, however, is not always observed. It is observed if:

$$y_j^{selection} = z_j\gamma + \varepsilon_{2,j} > 0 \quad (3)$$

where z_j is a vector of variables that explain involvement in the agricultural industry and γ the parameters to be estimated, and:

$$\varepsilon_1 \sim N(0,1)$$

$$\varepsilon_2 \sim N(0,1)$$

$$corr\ (\varepsilon_1, \varepsilon_1) = \rho \quad (4)$$

When $\rho \neq 0$, i.e. when the two error terms are correlated, then the standard probit model will produce biased results. The "heckprobit" procedure is intended to correct for selection bias and to provide consistent, asymptotically efficient estimates for all the parameters in the model.

Second, again fitting a bivariate probit model with selection, we investigate the potential non-response bias than can arise when protest responses are removed from the sample. In this case, the two binary outcomes to be jointly explained are the decision whether to participate or not in the hypothetical market created and the decision whether to protest or not, as in Ramajo-Hernández and Saz-Salazar (2012) and Saz-Salazar et al. (2016). Therefore, let d_j be a dichotomous variable that takes the value of 1 if the respondent decides to participate in the market, and 0 otherwise, while d_j^* represents the latent dependent variable of the participation equation. Hence:

$$d_j^* = A_j\beta + x_j\gamma + \varepsilon_{1,j} \quad (5)$$

where A_j is the payment offered to the individual and, in this case, x_j is a vector of variables that explain the decision to participate, while β and γ are the parameters to be estimated. But again we observe only a binary outcome (probit equation) such that:

$$d_j^{probit} = d_j^* > 0 \quad (6)$$

Hence, the dependent variable is observed if:

$$y_j^{selection} = z_j\delta + \varepsilon_{2,j} > 0 \quad (7)$$

where z_j is a vector of variables that explain the protest decision and δ the parameters to be estimated, and:

$$\varepsilon_1 \sim N(0,1)$$

$$\varepsilon_2 \sim N(0,1)$$

$$corr\ (\varepsilon_1, \varepsilon_1) = \rho \qquad (8)$$

2.5.3 Survey design and data collection

In contingent valuation studies the use of focus groups and pilot surveys is crucial to test the comprehension of the information provided in the questionnaire and also to develop information on the offer amounts (bids) used to elicit WTP (Boyle, 2017). A pilot survey is a small-scale test of the draft survey materials and implementation process (Champ, 2017) while a focus-group is a research technique that collects data through group interaction on a topic determined by the researcher (Morgan, 1996). In this particular case, the group discussion, led by a skilled moderator, was conducted several times with 8 to 10 participants. For the pilot survey a group of 50 households were surveyed, approximately 10% of the final sample. The final survey instrument was administered in 2019. Since the municipality is small, the researchers knocked on the door of all the homes. In the end, 468 completed questionnaires were obtained. Users were informed during the interview that the research was funded by the European Union and had the approval of the local government, and that the results could inform further action taken; the intention was to convey that the research was rigorous and to emphasize the importance of the responses (Poe and Vossler, 2011).

The questionnaire was divided into three parts. The first contained questions to determine the respondents' concern about the state of the environment. Respondents were also asked about their perception of the quality of the municipal water service. These questions were useful for two reasons. First, they served as an introduction to the valuation scenario; second, some of the information obtained in this section was subsequently used as a predictor of the WTP.

In the second part of the questionnaire, respondents were shown the assessment scenario to determine their WTP to maintain the financial balance of the municipal water service. To reinforce the credibility of the hypothetical market constructed and to prevent the free ride behavior typical of voluntary payments (Carson, 1997) the payment mechanism proposed was an increase in the water bill paid on a quarterly basis since respondents were familiar with it. The elicitation method used was the single-bounded dichotomous choice format (Bishop and Heberlein, 1979) since it closely mimics market situations while being incentive compatible (Arrow et al., 1993). However, the respondents were first asked a binary question with the aim of determining whether or not they were in the market, thus allowing a Spike model to be estimated (Kriström, 1997).

Based on the results from the pre-test of the survey and following the model for optimal bid selection proposed for Cooper (1993), for the discrete choice question 8 different bids were considered (€3, €6, €9, €12, €15, €18, €21 and €24). As pointed out by Schläpfer (2008), in determining the number of values the challenge is to achieve the proper balance between probing a sufficiently wide range of cost figures and keeping these cost figures within a credible range.

More specifically, they WTP questions were asked as follows:

Question 1: Currently, the amount you pay for water is below the real cost of the service provided by the Town Council. In fact, the Town Council only covers approximately 40% of the costs of the service with the income it collects from the water bill. In order to contribute to the economic sustainability of the service and to increase water quality, would you be willing to pay more in your water bill? Please answer "yes" or "no" considering what you currently pay for your water bill, that you have a limited available budget, and that at some point the public administration may also ask you to pay to finance other public services. Question 2: Considering your positive answer to the previous question, would you willing to pay an extra amount of € A in your quarterly water will in order to benefit from the proposed improvement in water quality? Yes, No, Don't know.

In contingent valuation surveys, where follow-up questions are used, "anchoring" or starting point bias occurs when the bid mentioned in the initial question has a noticeable effect on the subsequent response (Green et al., 1998; Herriges and Shogren, 1996; Veronesi et al. 2011). This is usually the case of the double-bounded dichotomous choice format, hence that, in order to avoid this problem, in this study we have opted for the single-bounded dichotomous choice format. In this respect, as respondents were informed that the town council covers about 40% of the cost of the water service, it could be though that respondents could have anchored their valuation to this information. However, we believe that is not the case since no information about the cost of the service in euros was provided to the respondents.

For the purpose of our investigation, it was key to determine how many people were not willing to pay more for the water bill, even after being informed of the financial imbalance of the service, as well as the reason for their non-willingness to pay. A true zero reflected non-willingness to pay due to an insufficient level of income. A protest response indicated that the non-willingness to pay was due to the conviction that the service should be financed through public subsidies, or simply because respondents believed they had the right to enjoy the service without having to contribute more to ensure its financial balance.

The questionnaire concluded with a third block of validation questions, intended to obtain socioeconomic, attitudinal and behavioral information about the individual (table 2.1). For example, respondents were asked about political ideology, membership of environmental associations, family size, sex, level of personal and family income, and educational level. This last block of the questionnaire contained another question that was key to the investigation: the respondent was asked if their income depended directly on farming. In order to test the main research hypothesis, those in the protest response group who were linked to agricultural activity were identified as the pressure group.

Table 2.1 shows the variables constructed from the information collected through the questionnaire, which are used in the estimations reported in the results section.

Table 2.1 Explanatory variables and summary statistics

Variable	Definition
INCOME	1 if respondent's income is equal or greater than €900, 0 otherwise.
H_ECONCERN	1 if the respondent is highly environmentally concerned, 0 otherwise. Highly environmental concerned means that the respondent, on a scale from 1 to 5 stated a value equal or greater than 4 when asked about his concern for the nvironment.
EXPENSIVE	1 if the respondent when asked about the price of water using a scale from 1 to 5 (1=very cheap and 5 = very expensive) stated a value equal or greater than 4, 0 otherwise.
TURBIDITY	Respondent's perception of the turbidity of the water from the tap before the construction of the current treatment plant on a scale from 1 to 5 (1 = never; 5 = always).
SMELL	Respondent's perception of the smell of the water from the tap before the construction of the current treatment plant on a scale from 1 to 5 (1 = never; 5 = always).
BAD_TASTE	Respondent's perception about the bad taste of the water from the tap before the construction of the current treatment plant on a scale from 1 to 5 (1 = never; 5 = always).
BOTTLED_W	1 if the respondent states that she drinks bottled water, 0 otherwise.
SATISFIED	1 if the respondents, when asked about her satisfaction related to the quality of the water from the tap, stated a value > 5 on a scale from 0 to 10 (0 = totally dissatisfied; 10 = totally satisfied), 0 otherwise.
WQ_CONCERN	Respondent's level of concern about water quality on a scale from 1 to 5 (1 = not at all concerned; 10 = extremely concerned).
CHILDREN	Number of children in the family unit
INACTIVE	1 if the respondent is economically inactive (student, pensioner and housewife), 0 otherwise.
RIGHT_WING	1 if the respondent has right-wing ideology, 0 otherwise.
INVOLVE_AGR	1 if the income of the interviewees depend directly on farming, 0 otherwise.

2.6 Results

2.6.1 WTP estimates

Almost 60% of the respondents stated that they were not willing to pay any extra money in their water bill in order to guarantee water supply reliability while maintaining quality standards. This is a fairly high rate of zero responses although considerably lower than the 77%, 65% and 75% obtained respectively by Kriström (1979), Dziegielewska and Mendelsohn (2007) and Lee and Yoo (2016). Even though some zero bids are a true reflection of individuals' preferences—40% of all zero responses in this research—others may be motivated by protest behavior, as mentioned before. The usual way of differentiating between a true zero WTP and a protest response is to present those respondents that are not willing to pay with a set of debriefing questions (Meyerhoff and Liebe, 2006). Accordingly, Table 2.2 shows the reasons behind a ‘‘no’’ WTP response. The proportion of protest responses was 35.7%, which is close to the upper limit of what is considered acceptable in CVM studies: from 20% to 40% according to Carson (1991). The main reasons for protesting were that “it is my right to expect clean water without paying extra” (16.4%) and that “the local government should fund the proposed policy aimed at improving water quality” (12.4%).

Table 2.2 Reasons for a “No” WTP response

Reason	Number (%)
True zero responses	
I can not afford to pay (income constraints)	100 (21.4)
Water quality is not the most important problem	13 (2.8)
Protest responses	
The local government should fund the proposed improvement in water quality	58 (12.4)
It is my right to expect clean water without paying extra	77 (16.4)
Lack of trust in the local government, to much waste	32 (6.8)

Note: percentages are calculated over the full sample (468 interviews)

For the purpose of the research, it is interesting to determine the protest response rate in each of the identified groups: those linked to agricultural activity and those without links to agriculture (Table 2.3). The results of the survey show that around 36% of the interviewees gave a protest response when asked about willingness to pay. In addition, contrary to the hypothesis raised, it was found that the protest response rate is not higher in the group of people whose income depends on agricultural activity.

Table 2.3 Percentage of protest responses among respondents according to their link to the agricultural industry.

	All responses (1)	Protest responses (2)	(2)/(1) %
Involved in agricultural industry	293	104	35.49 %
Non involved in agricultural industry	175	63	36.00 %
All the sample	468	167	35.68 %

Table 2.4 shows the coefficients of the parametric models estimated in order to determine how much the average respondent is willing to pay to enjoy the welfare increase resulting from improved water supply reliability and quality. As we used two dichotomous choice questions in the valuation scenario, it is feasible to estimate a Spike model (del Saz-Salazar and García-Menéndez, 2003) in addition to the more usual probit model. Therefore, respondents who answered “yes” to the first dichotomous question, i.e. indicating that they were in the market for this environmental good, were asked a second dichotomous question, offering them a bid. As expected, the higher the bid offered, the lower the probability of accepting it, i.e. the percentage of “yes”

responses is monotonically decreasing[3]. The probit model yields a mean WTP for the whole sample of €3.02, while the Spike model—which assigns a positive probability to zero responses, unlike the previous model—yields a higher mean WTP (€7.09), as in Saz-Salazar and García-Menéndez (2003). In both cases, protest responses have been excluded from the sample since, as will be shown below, there is no selectivity problem caused by excluding protesters. Nevertheless, when protest bids are included, the probit model yields an estimated mean WTP of −€2.17. This result is explained by the high rate of zero responses obtained and by the fact that the probit specification allows WTP to be negative. Considering the current tariff paid by the average city dweller (€34.8), the proposed policy would imply a hypothetical increase in water tariffs of between 8.7% and 20.4%, depending on which positive mean WTP estimate is chosen.

The same mean WTP estimates are now calculated for those respondents that are involved in the agricultural industry. In this case, the mean WTP is between €5.13 and €8.70, depending on the model chosen. Now the hypothetical increase of tariffs would lie in between 14.7% and 25% since these WTP estimates are respectively 70% and 23% higher than the corresponding estimates for the whole sample. Hence, contrary to expectations, it seems that respondents involved in the agricultural industry have a higher WTP..

[3] This implies that the coefficient of the bid should be negative and statistically significant. However, this does not hold for the Spike model since in this particular case, as noted by Kriström (1997), this coefficient (beta) should be positive, i.e. the marginal utility of money must be positive in order for the mean to exist. Thus, mean WTP is given by the following expression: ln [1 + exp (alpha)] / beta.

Table 2.4 Estimated models and mean WTP.

	Including protest responses				Excluding protest responses			
	All the sample		Involved in agricultural industry		All the sample		Involved in agricultural industry	
	Probit	Spike	Probit	Spike	Probit	Spike	Probit	Spike
Constant [a]	0.1912 (-1.30)	-0.8802*** (-8.63)	-0.0848 (-0.47)	-0.8643*** (-6.72)	0.2387 (1.39)	-0.1679 (-1.44)	0.4256** (1.98)	- 0.1504 (-1.02)
Bid (A) [a]	-0.0694*** (-5.91)	0.0733*** (8.23)	-0.0690*** (-4.91)	0.0595*** (6.06)	-0.0790*** (-5.88)	0.0863*** (8.62)	-0.0829*** (-5.06)	0.0713*** (6.33)
Mean WTP (€) and 95 % confidence interval [b]	-2.75 [-9.95 - 1.14]	4.74 [3.46 - 6.00]	-1.23 [-10.02 - 2.97]	5.90 [3.79 - 8.01]	3.02 [-1.72 - 5.79]	7.09 [5.36 - 8.83]	5.13 [0.14 – 7.94]	8.70 [5.87 - 11.51]
Log likelihood	-182.9290	-349.4914	-125.1067	-218.8886	-147.2424	-273.6019	- 97.8220	- 171.0007
LR chi2(1) [c]	39.38	67.79	26.81	36.77	39.01	74.24	28.64	40.02
Prob > chi2	0	0	0	0	0	0	0	0
Pseudo R^2	0.098	-	0.097	-	0.117	-	0.128	-
N	468	468	293	293	301	301	189	189

[a] t-statistic in parentheses.

[b] Krinsky and Robb (95%) confidence interval for WTP measures (number of repetitions 40,000). See Hole (2007) for a detailed explanation of how these confidence intervals are calculated.

[c] For the Spike model instead of the LR chi2(1) it is shown the Wald chi2(1).

***, ** significant at 1 % and 5%, respectively.

2.6.2 Selection models and WTP determinants

In addition to comparing WTP estimates, this study explains the probability of being willing to pay the proposed increase in water tariffs as a function of different variables. In doing so, using two bivariate probit models with selection, we test whether or not people involved in agricultural activities are more likely to protest, and we also test for the presence of sample selection bias. The explanatory variables used and their main descriptive statistics are shown in in Table 2.1, while the two models are shown in Tables 2.5 and 2.6, respectively.

Regarding the first model (Table 2.5), the LR test for independent equations (χ^2 = 2.20 and not statistically significant) shows that it is not possible to reject the null hypothesis that the two equations are in fact independent. It can thus be seen that the respondents involved in the agricultural industry are no more likely to protest than the other respondents.

The selection equation shows that the probability of being involved in the agriculture industry (INVOLVE_AGR) is negatively and significantly related to the respondent's income (INCOME), her environmental awareness (H_CONCERN), her perception that the water from the tap is cloudy (TURBIDITY) and being economically inactive (INACTIVE). On the other hand, having a right-wing ideology (RIGHT_WING), the number of children in the family unit (CHILDREN) and the belief that water is expensive (EXPENSIVE) are positively and significantly related to this outcome.

The protest equation (second column) shows that, as expected, the higher the respondent's income, the lower the probability of protesting. On the other hand, right-wing respondents, those that think water is expensive, and those with more children have a higher probability of protesting. Finally, the only organoleptic property of water that is related to the probability of protesting is its turbidity. However, the negative result is counterintuitive since we would have expected a positive relationship, i.e., the higher the turbidity of the water, the higher the probability of protesting.

Table 2.5 Probit selection model and protesting determinants.

Variable	Selection equation (Involve_agr =1)	Protest equation (Protest = 1)
CONSTANT	1.9564*** (3.21)	-0.1539 (-0.23)
INCOME	-0.7561** (-2.07)	-0.9891** (-2.39)
INNACTIVE	-0.8023 (-1.93)*	-0.5131 (-1.19)
RIGHT_WING	0.8147** (2.25)	0.7648** (2.31)
CHILDREN	0.2681*** (3.56)	0.1557** (2.18)
H_ECONCERN	-0.8391* (-1.74)	0.2623 (0.25)
EXPENSIVE	0.6755*** (2.45)	0.4860* (1.79)
TURBIDITY	-0.3870* (-1.73)	0.1384 (0.69)
SMELL	-0.0961 (-0.64)	0.0020 (-0.01)
BAD_TASTE	-0.1127 (-0.84)	-0.2514* (-1.85)
BOTTLED	-0.0746 (-0.31)	
Number of observations	156	
Log likelihood	-141.9109	
Wald Chi-squared (9)	23.59***	
LR test of indep. eqns. (ρ = 0)		
Chi-squared (1)	2.20	

Note: Z- statistic between parentheses. For the model to be well identified, the selection equation should have at least one variable that is not in the probit (participation) equation. * 10% significance level, ** 5% significance level and *** 1% significance level.

We now check for the presence of sample selection bias as a result of excluding protest responses from the sample. The validity of the WTP estimates will only be unaffected by the exclusion of these response if there is no sample selection bias. Otherwise, depending on the sign of the sample selection bias, the estimates will be upward or downward biased (Calia and Strazzera, 2001). As shown in Table 2.6, the correlation ρ between the error terms in both equations accounts for the presence of selection bias in the estimates of the parameters of the model. In this particular case, the LR test for independent equations ($\chi 2 = 0.10$) is not statistically significant, so it is not possible to reject the null hypothesis that that the decisions on whether to protest and on whether to enter the market are not independent. Accordingly, protest responses can be removed from the sample and WTP estimates will not be biased.

Table 2.6 Probit selection model and participation determinants.

Variable	Selection equation (Do not protest =1)	Participation equation (Enter = 1)
CONSTANT	-1.3347* (-1.78)	-2.4004** (-2.11)
INCOME	0.7077** (2.32)	0.2671*** (3.09)
RIGHT_WING	-0.6500** (-2.20)	
H_ECONCERN	0.5459 (1.37)	
EXPENSIVE	-0.5005** (-2.20)	-0.4772 (-1.34)
TURBIDITY	0.0531 (0.33)	0.7206*** (3.32)
SMELL	-0.0797 (-0.62)	-0.0449 (-0.29)
BAD_TASTE	0.1154 (0.96)	-0.2357 (-1.53)
WQ_CONCERN	0.0582 (0.67)	0.1571 (1.48)
SATISFIED	0.0681 (0.29)	-0.8007*** (-2.67)
CRISIS	0.2997** (2.35)	
Number of observations	207	
Log likelihood	-192.196	
Wald Chi-squared (8)	26.77***	
LR test of indep. eqns. (ρ = 0)		
Chi-squared (1)	0.01	

Note: Z- statistic between parentheses. For the model to be well identified, the selection equation should have at least one variable that is not in the probit (participation) equation. * 10% significance level, ** 5% significance level and *** 1% significance level.

The selection equation shows that the higher the respondent's income, the lower the probability of protesting. However, respondents that have a right-wing ideology and that think that tap water is expensive have a higher probability of protesting, as in the previous model estimated. In addition, respondents that claim to have been more affected by the recent financial crisis, which in the particular case of Spain had a severe, lasting impact resulting in increased inequality, are more willing to protest. Finally, neither of the organoleptic properties of water affects this probability.

Regarding the participation equation, results show that the higher the respondent's income, the higher the probability of entering the market, as expected. On the other hand, those respondents that stated that they were satisfied with the current quality of the water from the tap are less willing to enter the market as they do not consider it necessary to implement the proposed policy aimed at improving water quality.

2.7 Discussion, conclusions and recommendations

The Government of Spain is seeking to improve water management in Spain. In the last quarter of 2018, it opened discussions with different actors with the aim of preparing a Green Paper on Water Governance to serve as the basis for the future drafting of a new National Hydrological Plan. (Ministerio para la Transición Ecológica y el Reto Demográfico, 2020)

Among other issues under discussion in this process, there has been particular concern about the urban water cycle in small municipalities (García-Rubio and González-Gómez, 2020). Many small population centers are still far from achieving full cost recovery. In the consultations carried out, emphasis has been placed on promoting the aggregation of municipal processes to reduce production costs. However, the possibility of raising the price of water in rural areas has been left out of the debate.

The starting point of this research is the idea that financial imbalances in rural municipalities can also be reduced by raising the price of water. In this context, we question whether people linked to agriculture form a lobby that tends to curb possible increases in the price of water. If this were the case, it would be advisable for public authorities to implement specific actions aimed at overcoming the possible resistance of the agricultural population.

However, contrary to expectations, the research carried out does not allow us to confirm the hypothesis that the agricultural population behaves significantly differently when faced with a hypothetical increase in the water bill. In fact, the percentage of protest responses against the possibility of raising the price of water is practically the same among all respondents. Indeed, the results show that the low rate of support for an increase in the water bill is mainly due to interviewees' budget constraints. Low income is a determinant of higher probability of true zero, protest responses and less willingness to pay in case of market entry.

In any case, it is striking that, among those willing to pay more to cover the costs of the water service, and given equal purchasing power, the agricultural population shows a greater willingness to pay. This may be due to the fact that, in a scenario of water scarcity, both the marginal utility of water and the value assigned to water are greater for As water is of great value to them in their main economic activity, this group is likely to transfer that value to the residential area.

The first conclusion that can be drawn from the results obtained is that, at least regarding water for residential uses, there is no evidence that people linked to agriculture form an interest group representing an obstacle to raising water prices for residential uses in rural areas. A second conclusion is that, despite the low proportion of users willing to pay more for water in rural areas, which is mainly due to low income levels, there is some scope for raising the price of water for residential uses. The findings of this study suggest that any such raises would be modest, but would contribute to improving cost recovery.

The main recommendation is that public authorities should study increases in the price of water in each municipality. However, considering the average income in rural areas, particularly in agricultural activity, any increases in the price of water should be accompanied by discounted rates for low-income families. In Spain, low-income discounts can be found in almost all large municipalities, but not in small municipalities (García-Rubio et al., 2015; López-Ruiz et al., 2020). Additionally, in order for water price increases to be accepted in low-income settings, information, communication and education campaigns should be implemented (Felgendreher and Lehmann, 2016),

A limitation of the research is that the main hypothesis has been tested using data from a single municipality. It would thus be desirable to replicate the research in other population centers. In any case, the results obtained provide sufficient evidence to support the recommendation to study possible increases in the price of water in each municipality. On the other hand, an interesting line of research could focus on local government politicians as the main actor. On the demand side, we conclude that a low level of income, rather than links to agricultural activity, is the main limiting factor in urban water prices; on the supply side, it is worth asking whether local politicians contribute to keeping water prices low in order to enjoy the support of their local electorate. Local politicians may be the main obstacle to water price increases in in rural areas. Kayaga et al. (2018) recently identified a paradoxical situation in Ethiopia, whereby citizens are willing to pay more to enjoy a quality public water service while governments appear unwilling to charge citizens more for water, a situation that could also occur in developed countries.

Finally, although contingent valuation provides a good basis for informed decision-making when environmental values are involved as it is the case in this study, we cannot overlook that the estimated WTP values are somewhat “uncertain” because they are heavily dependent on the assumptions made about the implicit consumer preferences and the different empirical models used in the WTP inference process (Bengochea-Morancho at al., 2005)

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CAPITULO 3. Biological nitrate removal from a drinking water supply with an aerobic granular sludge technology: An environmental and economic assessment

A preliminary version of this chapter was disseminated as a working paper for the LIFE ECOGRANULARWATER (LIFE16 ENV/ES/196) project. The key final results have been published in the paper Biological nitrate removal from a drinking water supply with an aerobic granular technology. Environmental and economic assessment. *Journal of Cleaner Production, 367,* 133059. https://doi.org/10.1016/j.jclepro.2022.133059

3.1 Abstract

Nitrate pollution of groundwater, mainly from agricultural applications, is a widespread water quality problem in many countries. The aim of this study was to identify and compare the main environmental impacts and costs associated with removing nitrate from groundwater under a conventional treatment technology and a technology based on a biological treatment process (denitrification using aerobic granular sludge). The analysis focused on the first real-world experience of an industrial-scale implementation of said technology, applied to the drinking water supply in a small town in Spain. The methods selected for the environmental and economic evaluations were life cycle assessment and cost-effectiveness analysis, respectively. The drinking water treatment technologies under study were a conventional reverse osmosis plant and a plant using a biological treatment called ecogranularwater. The analysis of these two drinking water production processes was divided into two phases: structure and water treatment. This study demonstrates that the biological technology produces drinking water in a more environmentally-friendly, cost-effective way, and with lower energy costs. The greatest environmental impacts from the reverse osmosis technology occurred in the water treatment phase due to the high levels of energy consumption (up to 1.68 kWh m^{-3} higher than with the ecogranularwater technology). In the structure phase, the biological technology contributed more than reverse osmosis in all impact categories, with values ranging from 91% in freshwater ecotoxicity to 98% in stratospheric ozone depletion. The cost of producing 1 m^3 of water was estimated as 43% lower with the ecogranularwater technology compared to reverse osmosis. In order to further lessen the environmental impact of the biological technology, efforts should be focused on reducing and optimizing energy use and making improvements to the design of the structure. This biological technology proved a good alternative for small and medium-sized municipalities with problems of nitrate-polluted water. The decision to apply technological innovations in drinking water treatments to remove nitrates can be supported by environmental and economic studies.

3.2 Keywords

Life cycle assessment, Nitrate-polluted water, Drinking water, Environmental footprint, Cost-effectiveness analysis, Biological treatment

3.3 Introduction

The constant release of chemicals into water bodies is leading to a deterioration in the quality of our water resources. These chemicals include emerging water contaminants such as pharmaceutical product residues, as well as pollutants from dyes, fertilizers and pesticides, all of which pose serious problems to human health and the environment (Ali et al., 2018; Basheer, 2018; Basheer and Ali, 2018). Groundwater is one of the main sources of drinking water in most countries, especially in arid and semiarid climates. Groundwater pollution, caused by the release of chemicals such as nitrogen into the ground, is a major issue these days. Nitrogen (N) losses lead to increased nitrate concentrations in aquifers and the eutrophication of rivers, lakes, and transition surface waters (Smith, 2003; Smith and Schindler, 2009). Since 1990, the nitrogen surplus in Spain has been growing, mainly due to inadequate irrigation and fertilization management (EEA, 2006). Agriculture is responsible for potentially high levels of N loss, whether through nitrate leaching (Pratt, 1984; Ramos et al., 2002; Thompson et al., 2007; Guo et al., 2010), gaseous losses or large quantities of N left in the soil after the crop harvest, which can be washed out by the rains falling between consecutive crops. Some studies point to the reduction and optimization of fertilization as the most efficient way to improve the environmental performance of crop production (Romero-Gámez and Suárez-Rey, 2020).

Certain areas in Spain are classified as zones that are at risk of agricultural nitrate pollution (Nitrate Vulnerable Zones or NVZs), according to the European Union directive on measures to prevent aquifer contamination by fertilizers (Council of the European Communities, 1991). This directive contains mandatory measures relating to agricultural practices, aimed at reducing nitrate pollution of groundwater and surface water. It establishes a limit for the concentration of nitrates in drinking water of 25 milligrams per litre (mg L^{-1}). The present study is focused on an area designated an NVZ, specifically, the municipality of Torre-Cardela (north-central area of Granada province, Andalusia, Spain), where water for human consumption is taken exclusively from the aquifer.

In Torre-Cardela, as in many other municipalities in Spain, the management of the municipal water service gives rise to both environmental and economic problems. Regarding the environmental problems, in addition to the fact this area is classified as an NVZ (BOJA, 2020), the Torre-Cardela reverse osmosis plant consumes 1.42 m^3 of raw water to produce 1 m^3 of water, meaning that 42% of the treated water is being rejected. This is a serious issue in a Mediterranean region facing water scarcity, recurring droughts and threats from climate change. On top of this is the high energy consumption required for the process (around 2 kilowatt-hour (kWh) per m^3 produced) and, to a lesser extent, the chemicals used in the process. The economic problem is basically that the financial costs of the service, that is, the investment costs and the operating and maintenance costs are not being covered. Moreover, environmental and opportunity costs are not considered. According to our own estimates based on data provided by the town council, only 60% of the financial costs of the service are covered through the revenue generated by urban water tariffs.

In response to these problems, a new alternative for treating drinking water in the study area was developed and optimized. The new treatment is based on aerobic granular sludge technology (nitrate removal by means of heterotrophic microbial metabolism in granular biomass). This biological technology applies the lowest concentration of carbon source needed to remove nitrates at concentrations higher than

25 mg L^{-1}, which determines the eukaryotic and prokaryotic microorganisms involved in the denitrification process, as well as the granular stability (Hurtado-Martínez et al., 2021a). Aerobic granular sludge systems require 25-50% less floor space than a conventional wastewater treatment system, use 25-40% less electricity than a conventional activated sludge system, and their operating costs are 20-25% lower (Sarma et al., 2017). Biological drinking water treatment systems are considered environmentally-friendly, cost-effective systems because the processes are carried out by heterotrophic or autotrophic microorganisms. However, biological technologies may also present some problems due to their operating conditions and the initial investment or maintenance costs of the bioreactors (Ahmed et al., 2017). Various different strategies have been applied in biological wastewater treatment technologies to achieve a successful nitrate removal rate, such as the use of external carbon sources in anoxic reactors (Panepinto et al., 2013).

Denitrification of groundwater to ensure safe nitrate levels can be done through either separation- or removal-type technologies. Separation technologies include ion exchange, nanofiltration, electrochemical reduction, and reverse osmosis (Rezvani et al., 2019). However, the generation of brine as a secondary waste product and high operational costs are disadvantages compared to removal-type methods, which completely remove nitrates by converting them to dinitrogen. Removal-type methods can be based on chemical and/or biological processes. Some of the chemical and physical technologies used to remove nitrates and other contaminants (e.g., pharmaceutical residues or heavy metal ions) from water include metal nanoparticles (Adeleye et al., 2016; Ali et al., 2018, 2019; Shahat et al., 2018), electrodialysis (Martínez et al., 2017), ion exchange adsorption (Awal, 2019; Kamel et al., 2019), reverse osmosis (Epsztein et al., 2015), chemical reduction (Eljamal et al., 2020a, 2020b) and catalytic and electrocatalytic reduction (Siciliano, 2015; Hashim et al., 2017). However, chemical and physical processes have high start-up and operating costs, as well as high energy requirements, which is an obstacle to their implementation in small population centres.

Reliable environmental assessment tools are therefore needed to determine the level of environmental sustainability of these processes used to produce water for human consumption. The life cycle assessment (LCA) concept is used to analyse the environmental impact of industrial products or production process, as well as for the evaluation of treatment processes (Ortiz et al., 2007; Vince et al., 2008). Several studies have used the LCA methodology to assess the environmental impacts of different water treatment technologies, especially for making an objective comparison between alternative and conventional desalination processes (Muñoz and Rodríguez, 2008; Vince et al., 2008; Tarnacki et al., 2011; Zhou et al., 2011; Qiu and Davies, 2012; Lawler et al., 2015). Previous studies have concluded that the choice of water treatment chemicals and the energy source are critical elements in the LCA of the production of drinking water from groundwater and fresh surface water since energy consumption and chemical dosing have high environmental impacts in different processes. Thus, several authors find that the construction and infrastructure of the plants have less of an impact than the operational phase (Friedrich et al., 2001; Raluy et al., 2005; Stokes and Horvath, 2006; Bonton et al., 2012).

The environmental impacts of production processes usually remain in the background, with economic profitability being prioritized; however, this framework no longer reflects the scientific and social reality. It is important to carry out a rigorous comparative assessment of different production systems in terms of their environmental and economic impact. Society is becoming increasingly aware of environmental concerns and sustainability in general. There is thus a need for integrated environmental and economic LCA of drinking water systems (Bonton et al., 2012). The design and implementation of cost-effective, environmentally-friendly production processes is a fundamental challenge that must be tackled to ensure the sustainability of agricultural systems and ecosystem services. Therefore, the aim of this work is to design a sustainable, high-quality product and service to meet societal demand.

To that end, the present study conducts a comparison of the environmental impacts and costs associated with two technologies for removing nitrate from groundwater: 1) a physical-chemical technology (reverse osmosis, RO) and 2) a biological technology (ecogranularwater, EGW). To compare the two technologies, LCA and an economic impact analysis are carried out. To the best of our knowledge, the new biological plant under study—a pilot plant that supplies nitrate-free water to a Spanish municipality—is the first of its kind built on an industrial scale. Furthermore, the present study is the first environmental and economic analysis to include the most relevant elements of both technologies. A major contribution of this study is that it provides detailed information on nitrate removal technologies for use in municipalities classified as NVZs, facing serious environmental and economic problems related to the management of the municipal water service.

3.4 Materials and methods

3.4.1 Environmental impact assessment

LCA was used to evaluate the environmental footprint of drinking water production, in accordance with the European Commission recommendation on the use of common methods to measure and communicate the life cycle environmental performance of products and organizations (2013/179/EU) and the standards ISO 14040 and 14044 (2006). LCA is a very useful methodology for measuring the environmental performance of a process and/or product, since it allows us to evaluate the associated environmental loads and to determine the impact on the environment of its use of resources, materials and energy inputs, and emissions. The main function of LCA is to support decision-making about a process and/or product, and more specifically, to offer an understanding of its possible environmental consequences. In addition, LCA can inform the implementation of environmental improvement strategies. There are four main stages in an LCA study according to ISO 14040: a) goal and scope definition, b) inventory analysis, c) impact assessment and d) interpretation of the results. This work includes the mandatory phases (classification and characterization) and the optional phase (normalization) of impact assessment, as defined by the ISO standard.

3.4.1.1 Goal and scope definition

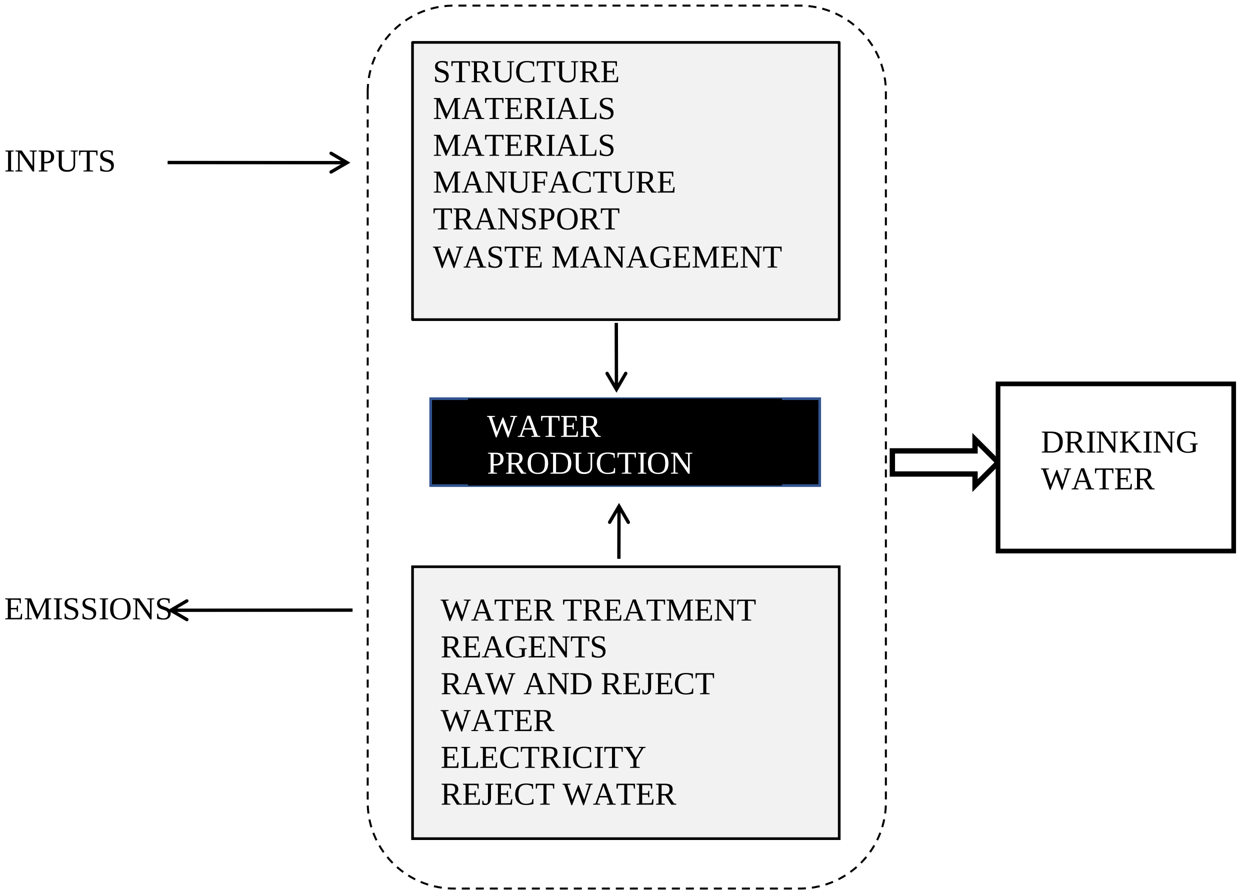
The main aim of this study was to calculate and compare the environmental footprint of two drinking water production technologies. More specific objectives were: (i) to conduct a life cycle inventory (LCI) with all the flows and processes involved in the selected drinking water production plants; (ii) to identify the processes and phases that produce the most significant environmental issues; and (iii) to design and propose to users a more efficient, environmentally-friendly production system, while at the same time suggesting possible strategies for mitigating the environmental impact.

The scope of this study was the production of drinking water, considering all the input and output flows of materials and energy up to the point where the water has been treated for human consumption. According to the Product Category Rule (PCR) for water distribution through the mains, the unit of analysis is the production of 1 m^3 of water fit for human consumption. Therefore, the functional unit (FU) was defined as 1 m3 of drinking water.

The study compared two drinking water treatment technologies: a conventional plant with RO water treatment and a new plant using a biological treatment called ecogranularwater (EGW). The analysis focused on both the structure of each plant and the water treatment process. The system boundary was set at the intake to both plants without considering the pumping from the raw water tank, since this configuration may differ depending on where the facilities are located and could distort the results in relation to energy consumption and thus the impacts. Therefore, the LCA focused exclusively on the production of 1 m3 of water fit for human consumption, produced by a biological process and by RO.

The environmental analysis of the production of drinking water by the two technologies included the following elements: the transport, manufacture and waste management of the materials used in the infrastructure of the two technologies; the chemicals used in the production of the water; the raw water consumed in the production of 1 m3 of drinking water; the reject water in the RO technology only, since no water is rejected in the EGW technology; the energy consumed during the production process; and the emission of chemicals into the reject water. The analysis did not include the bacteria rejected in the sand filter of the EGW technology since it was deemed to have little or no environmental impact compared to the other elements included, as reported in previous studies (Muñoz and Rodríguez Fernández-Alba, 2008). Figure 1 is a flow chart of the water production technologies considered, structured in two phases (plant infrastructure and water treatment), each showing the processes and flows. This chart facilitates inventory analysis, impact analysis and interpretation of the study results.

Fig. 3.1 Flow diagram of the phases analysed in each drinking water production technology.



3.4.1.2 Life inventory analysis

The data on the two drinking water production technologies evaluated were collected through different suppliers, literature, experimental data, and direct measurements, and in some cases were tested to check the accuracy of the data. The life cycle inventory (LCI) included representative local data from the two production plants in the municipality of Torre-Cardela, Granada (Spain), the geographical coordinates of which are 37°30' N, 3°21' W.

The inventory data on the RO technology were collected from the town council's billing information on technology inputs and direct readings of the rotameters, electricity meter and chemical dosing systems. For the infrastructure, direct measurements were made of the materials used. For the EGW technology, the inventory was based on direct readings of electricity consumption provided through the monitoring carried out in this plant, information on the chemicals was obtained from the concentrations applied in the production process, and the information on water used was sourced from the measurement of the water in the cleaning filter. Infrastructure data were taken from direct measurements in the plant.

The inventory data collected span the period from 2019 to 2021. Data collected for each phase were related to materials and energy consumption, emissions into the water and atmosphere, and the waste generated. All data for the environmental analysis were obtained from the Ecoinvent database v. 3.7.1. (Ecoinvent, 2021), including data on the manufacture of materials needed to install the infrastructure, manufacture of chemicals, electricity production mix, and materials and waste transport and disposal. The emissions of chemicals into the RO reject water were measured directly in the laboratory.

The LCA involved the abovementioned data collection to quantify the relevant inputs and outputs during the life cycle of the drinking water production process of each technology. The main characteristics of these technologies are shown in Table 3.1.

Table 3.1 Main characteristics of selected drinking water production technologies.

	REVERSE OSMOSIS	ECOGRANULARWATER
Drinking Water (m3 h-1)	4	1.6
Production period (hours)	1	2
Raw Water (m3 h-1)	1.42	1.02
Reject water (m3 h-1)	0.42	0.02
Electricity consumed (KWh m-3)	2.05	0.37

The analysis of the drinking water production process was divided into two phases (structure and water treatment) to facilitate data compilation, the assessment, and the interpretation of the results. The detailed quantitative data for all the materials and processes related to the structure and the chemicals included in the water treatment are summarized in Tables 3.2, 3.3, 3.4, 3.5 and 3.6.

3.4.1.2.1 Structure

The materials used in the manufacture of the elements of the RO and EGW technologies were included in this phase of the analysis (Tables 3.2 and 3.3, i.e., high density polyethylene (HDPE), steel, polycarbonate (PC), polypropylene (PP), GFRP, PVC etc. used in pipes, frames, clamps, filters, membranes, tanks, pumps, sensors, and reactors. The materials used in both structures were raw materials, so the analysis also accounted for manufacturing processes such as blow moulding, metal product manufacturing and plastic extrusion. In addition, this phase of the analysis included the extraction and transport of each material by truck to the production sites, as well as the management and transport by truck of waste materials (metal and plastic) to landfill and recycling centres (Tables 3.2 and 3.3) (BOJA, 2012).

Table 3.2 Materials and processes for the Reverse Osmosis technology structure included in the life cycle inventory.

Materials	Elements	Quantity*	Unit
PVC	Pipes	50.49	kg
Galvanized steel	Structure and frame	173.11	kg
Stainless steel	Clamps and pressurizer	191.30	kg
HDPE	Filters and tanks	13.68	kg
GFRP	Vessels	64.10	kg
PP	Membranes	23.04	kg
Materials	**Processes**	**Quantity***	**Unit**
PVC	Extrusion, plastic pipes	50.49	kg
Steel	Manufacturing, metal working	364.41	kg
HDPE	Blow moulding, tanks	12.18	kg
Transport of materials to the production sites	freight, lorry 7.5-16 metric ton	33.10	tkm
Materials	**Waste**	**Quantity***	**Unit**
PVC	recycling centre	50.49	kg
Galvanized steel	recycling centre	173.11	kg
Stainless steel	recycling centre	191.30	kg
HDPE	recycling centre	13.68	kg
GFRP	landfill	64.10	kg
PP	recycling centre	23.04	kg
Transport of waste to the recycling centre	freight, lorry 7.5-16 metric ton	28.8	tkm
Transport of waste to landfill	freight, lorry 7.5-16 metric ton	2.72	tkm

*total amount of material without considering life span

PVC: polyvinylchloride; HDPE: high density polyethylene; GFRP: glass fibre reinforced polyester; PP: polypropylene; tkm: ton per kilometre

Table 3.3 Materials and processes for the ecogranularwater technology structure included in the life cycle inventory.

Materials	Elements	Quantity*	Unit
PVC	Pipes and dosing system	31.93	kg
Brass	Pipes	3.30	kg
Rubber	Pipes and clamps	5.64	kg
Galvanized steel	Frame, tramex	298.00	kg
Stainless steel	Pumps, reactor 1 and blowers	145.83	kg
Zinc plated steel	Clamps	4.45	kg
Carbon steel	Sand filter	80.00	kg
PC	Sensor and transmitters	4.28	kg
HDPE	Tanks	42.00	kg
GFRP	Reactor 2 and 3	1700.00	kg
PMMA	Reactor 1	27.53	kg
Materials	Processes	Quantity*	Unit
PVC	Extrusion, plastic pipes	31.93	kg
Steel	Manufacturing, metal working	528.28	kg
HDPE	Blow moulding, tanks	29.90	kg
Transport of materials to the production sites	freight, lorry 7.5-16 metric ton	544.41	tkm
Materials	Wastes	Quantity*	Unit
PVC	recycling centre	31.93	kg
Brass	recycling centre	3.30	kg
Steel	recycling centre	528.28	kg
HDPE	recycling centre	42.00	kg
Rubber	landfill	5.64	kg
PC	landfill	4.28	kg
GFRP	landfill	1700.00	kg
PMMA	landfill	27.53	kg
Transport of waste to the recycling centre	freight, lorry 7.5-16 metric ton	37.77	tkm
Transport of waste to landfill	freight, lorry 7.5-16 metric ton	73.84	tkm

*total amount of material without considering life span

PVC: polyvinylchloride; HDPE: high density polyethylene; GFRP: glass fibre reinforced polyester; PC: polycarbonate; PMMA: methyl methacrylate; tkm: ton per kilometer

The time periods considered for the environmental assessment of the frame materials (average life span for the remaining plastic and metal materials) are shown in Table 3.4.

Table 3.4 Life span of structure materials used in the Reverse Osmosis and Ecogranularwater technologies (years).

REVERSE OSMOSIS

Materials	Life span (years)
PVC	50
Stainless steel	100
HDPE	50
PRFV	120
Galvanized steel	50

ECOGRANULARWATER

Materials	Life span (years)
PVC	50
Brass	50
Rubber	30
Galvanized steel	70
Zinc plated steel	60
PC	25
PMMA	40
Stainless steel	100
Carbon steel	60
Steel	65
HDPE	60
PRFV	120

PVC: polyvinylchloride; HDPE: high density polyethylene; GFRP: glass fibre reinforced polyester; PP: polypropylene; PC: polycarbonate; PMMA: methyl methacrylate

3.4.1.2.2 Water treatment

The RO technology inputs included in this phase were the chemicals used (polycarboxylates and hydrochloric acid manufacturing) (Table 3.5), the raw water needed to generate the permeate water and the energy consumed in the water treatment process (Table 3.1). The chemical composition of the water from discharge was determined, and the emissions to water were included in the analysis (Table 3.5).

Table 3.5 Chemical compound added to raw groundwater and emissions to reject water in the Reverse Osmosis technology per FU (m-3).

Chemicals	g/m^3
Polycarboxylates	32.9
Hydrochloric acid, HCl	27.5
Water emissions	g/m^3
Nitrate, NO^{3-}	5.23E+01
Chlorides, Cl^-	1.52E+01
Calcium, Ca	8.77E+01
Magnesium, Mg	2.10E+01
Fluorides, F^-	2.68E-01
Nickel, Ni	1.03E-03
Potassium, K	9.47E-01
Silicon, Si	6.59E+00
Sodium, Na	7.82E+00
Sulfates, SO_4^{2-}	4.98E+01

The drinking water production process of the EGW technology involves the biological removal of nitrates by aerobic denitrifying bacteria that form a granular sludge in a sequencing reactor. Granular aerobic systems for drinking water treatment are based on the ability of microorganisms to degrade pollutants such as nitrate in groundwater. The operation of these systems relies on the metabolism of different microorganisms, including denitrifying bacteria such as Pseudomonas. These bacteria can carry the nosZ gene in their genome, which encodes nitrous oxide reductase, an enzyme responsible for the conversion of nitrous oxide (N2O) to dinitrogen (N2) (Eljamal et al., 2020b). Achieving an optimal nitrate removal process in an oligotrophic medium such as groundwater requires the addition of an external carbon source to the system, as it enables the complete denitrification process to occur correctly. There are many carbon sources that can be used, including sodium acetate, a compound that has proven effective in the biological nitrate removal process (Hurtado-Martínez et al., 2021b).

The C:N ratio is of vital importance in the removal process, and it has to be adjusted according to the nature of the water to be treated (Hurtado-Martinez et al., 2021a). In the case of nitrate-contaminated groundwater with 50 to 100 mg NO3- L-1, a lower C:N ratio will lead to the formation of smaller and denser granules, enabling efficient removal in compliance with the European Water Framework for drinking water. For higher concentrations of nitrate-contaminated groundwater (>120 mg NO3- L-1), C:N needs to be higher to ensure better removal. In this case, the size of the granules will be larger but the distance from the outer layer to the core of the granules allows a strong gradient of oxygen and nutrients, as occurs with a low C:N. The optimal mode of operation depends entirely on the in situ groundwater conditions, although general guidelines can be given; for example, the C:N ratio should range from a minimum of 1 to a maximum of 4 (Hurtado-Martínez et al., 2021a).

The main system inputs analysed were the chemicals used to maintain the bacterial communities present in the biological reactor: namely, sodium acetate (C2H3NaO2), potassium chloride (KCl), magnesium sulfate heptahydrate (MgSO4·7H2O), potassium dihydrogen phosphate (KH2PO4) and potassium monohydrogen phosphate (K2HPO4) (Table 3.6). There was practically no reject water in this process: the only reject water was that generated in the washing of the sand filter, which was done once a day in this pilot plant. It is estimated that in a plant on a larger industrial scale, an even smaller volume of reject water would be produced. Given the negligible volume of reject, it was discharged into a small wetland area created for that purpose; hence, the impact is not significant. The raw water needed to generate the permeate water and the energy consumed were also included in the analysis (Table 3.1).

Table 3.6 Doses of chemicals compounds added to groundwater in the ecogranularwater technology per FU (m-3).

Reactive	g/m^3
Sodium Acetate, $C_2H_3NaO_2$	100
Potassium chloride, KCl	2.6
Magnesium sulfate heptahydrate, $MgSO_4{\cdot}7H_2O$	6.3
Potassium monohydrogen phosphate, K_2HPO_4	6.1
Potassium dihydrogen phosphate, KH_2PO_4	1.5

3.4.1.3 Life Cycle Impact Assessment (LCIA)

The production burdens associated with drinking water production were calculated and evaluated using the LCIA methodology. The simaPro software v. 9.2.0.2 (PRé Sustainability, 2021) was used to model the systems and evaluate the environmental impacts, considering the classification, characterization and normalization stages set out in ISO 14040 (2006), which specifies the general framework, principles, and basic requirements to carry out an LCA. The LCIA was performed using a midpoint approach. The method used for the classification, characterization and normalization of the inputs and outputs of the inventory was the ReCiPe Midpoint (H) (Huijbregts et al., 2017). The six midpoint impact categories included in this study are shown in Table 3.7. These environmental impacts were chosen because of their relevance in energy and industrial processes, and in accordance with the PCR for water distribution through the mains included in the International Environmental Product Declaration (EPD) Systems (Environdec, 2021).

Table 3.7 Selected environmental impacts and units of measurement.

Impact category	Units
Carbon footprint	kg CO_2eq
Stratospheric ozone depletion	kg CFC-11 eq
Ozone formation, Terrestrial ecosystems	kg NO_x eq
Terrestrial acidification	kg SO_2 eq
Freshwater Eutrophication	kg P eq
Freshwater Ecotoxicity	kg 1,4-DCB

eq: equivalent; CO_2: carbon dioxide; CFC: chlorofluorocarbon; NOx: nitrogen oxides; SO2: sulfur dioxide; P: phosphorus; DCB: dichlorobenzene

3.4.2 Economic impact assessment

The RO plant began operating in 2012 and from the beginning the service ran at a deficit, with tariff revenues only covering around 60% of the financial costs of the service. As the municipal drinking water service was managed directly by the Torre-Cardela town council itself, the remaining 40% was subsidized using other sources of municipal income. Therefore, it did not comply with the principle of cost recovery established in the Water Framework Directive (EU, 2000).

For the economic impact analysis, a cost-effectiveness analysis was carried out. Cost-effectiveness analysis is an instrument used in River Basin Plans to design the action programme to be applied in each river basin. It is the method most used to choose the policy measures aimed at ensuring the good ecological status of water bodies, as indicated by the Water Framework Directive (Berbel et al., 2011). Cost-effectiveness analysis is used to estimate the monetary cost needed to achieve a water policy objective measured in physical terms. This method can be used to identify the measures that enable the same objective to be achieved at a lower cost.

In this case, the cost of producing 1 m^3 of water with each of the technologies was compared. In a first stage, only the financial costs of the service were considered. In a second stage, environmental and resource costs were also included, as stipulated by the Water Framework Directive. The operating costs considered were personnel costs, the costs of reagents (Table 3.8) and energy costs. In addition, the costs of the membranes and filters used in RO were included in the analysis.

Table 3.8 Detail of the costs for the use of reagents.

REVERSE OSMOSIS			
REAGENTS	g/m^3	COST (€/kg)	UNIT COST (€/m^3)
Polycarboxylates	32.97	12	0.3956
HCl	27.50	1	0.0275
TOTAL			0.4231

ECOGRANULARWATER			
REAGENTS	g/m^3	COST (€/kg)	UNIT COST (€/m^3)
Sodium acetate (CH_3COONa)	100	3.27	0.330
Magnesium sulfate heptahydrate ($MgSO_4 {*} 7H_2O$)	6.3	1.49	0.010
Potassium monohydrogen phosphate (K_2HPO_4)	6.1	3.64	0.020
Potassium dihydrogen phosphate (KH_2PO_4)	1.5	2.48	0.003
Potassium chloride (KCl)	2.6	2.02	0.010
TOTAL			0.367

To calculate the investment cost per m³ of water, the equivalent annual cost (EAC) of the investment was estimated using the following expression (Confederación Hidrográfica del Guadalquivir, 2015):

$$EAC = M\left[\frac{i(1+i)^n}{(1+i)^n - 1}\right]$$

M: Initial investment for the construction of the plant.

i: Discount rate.

n: Useful life of the structure.

The calculation of the EAC was based on a discount rate of 2% and a useful life of 20 years. In the case of the RO technology, the initial investment was 64,500 euros, while in the biological plant it was 60,000 euros.

In addition, the estimation of the cost of the investment per m³ of water was based on an average consumption of 133 litres per person per day. This is the average daily consumption of water per person in Spain according to data from the Spanish Statistics Institute (2020).

In a second stage, as mentioned above, environmental costs and resource costs were incorporated into the analysis. In comparative terms, the main environmental impact to consider was from the reject water generated in the production process. The impact was practically negligible in the case of the pilot EGW plant: the percentage of reject water was only 2% and due to its composition it does not have a negative impact on the environment. However, the high volume of reject water generated in the RO technology does have a negative environmental impact since it is brine. The estimation of this environmental cost followed a preventive approach: specifically, the cost of treating 1 m3 of wastewater was calculated. In this study, the value proposed was 0.31 €/m3 (Moral Pajares et al., 2019).

The cost of the resource was represented by the alternative use value of the raw water rejected in the production process. Considering this opportunity cost makes sense in a hydrographic basin subject to high water stress. The Guadalquivir River Basin Authority estimates the opportunity cost at 0.354 €/m3 (Confederación Hidrográfica del Guadalquivir, 2022).

3.5 Results

3.5.1 Environmental impact assessment

Table 3.9 shows the main impacts of the production of 1 m3 of drinking water with the two water treatment technologies under analysis. RO produced higher environmental impacts than the new EGW technology for all impact categories, with differences ranging from 1.66E-03 kg NOx eq. in the ozone formation category to 6.18E-02 kg 1.4-DCB in the freshwater ecotoxicity category (Table 3.9a). These differences can be attributed to the higher electricity consumption in the water treatment phase in RO (up to 1.68 kWh m-3 higher than with the EGW technology), although the EGW technology produced a much smaller volume of drinking water per hour (Table 3.1). Freshwater ecotoxicity was the category with the highest impacts for both technologies, followed by the carbon footprint category in RO and by freshwater eutrophication in EGW (Table 3.9b).

Table 3.9 Comparison of the main impacts of the production of 1 m3 of drinking water with the two water treatment technologies: a) characterized indicator results; b) normalized indicator results.

Impacts per m^3 FU		RO	EGW
a)			
Carbon footprint	kg CO_2 eq	8.53E-01	3.76E-01
Stratospheric ozone depletion	kg CFC11 eq	4.10E-07	2.14E-07
Ozone formation, Terrestrial ecosystems	kg NO_x eq	2.71E-03	1.05E-03
Terrestrial acidification	kg SO_2 eq	4.88E-03	1.57E-03
Freshwater eutrophication	kg P eq	3.61E-04	1.13E-04
Freshwater ecotoxicity	kg 1,4-DCB	8.31E-02	2.13E-02
b)			
Carbon footprint		1.07E-04	4.70E-05
Stratospheric ozone depletion		6.85E-06	3.57E-06
Ozone formation, Terrestrial ecosystems		1.52E-04	5.90E-05
Terrestrial acidification		1.19E-04	3.83E-05
Freshwater eutrophication		5.55E-04	1.74E-04
Freshwater ecotoxicity		3.30E-03	8.46E-04

In both technologies, the water treatment phase contributed more to all impact categories than the structure phase, due to the application of chemical doses and energy consumption during the treatment processes (Tables 3.5 and 3.6). Table 3.10 shows the most relevant elementary flow, compartment, main life cycle phase, and main processes in RO and EGW for each impact category. Emissions related to inputs in the water treatment phases, mainly due to chemical compounds added and electricity consumed, were the main contributors to the impact categories studied. The carbon footprint impact was predominantly driven by carbon dioxide emissions to air caused by the electricity consumed in the RO treatment and the application of organic chemical compounds in the EGW treatment. Stratospheric ozone depletion was determined by dinitrogen monoxide emissions to air due to inputs such as the electricity consumed in RO and inorganic chemical compounds added to raw groundwater in the EGW treatment. The ozone formation category was primarily driven by nitrogen oxide emissions to air (mainly caused by the electricity consumed) and the application of organic chemical compounds, in RO and EGW technologies, respectively. Sulfur dioxide emissions to air and phosphate and copper emissions to groundwater from the electricity consumed during the water treatment process in both technologies were major pollutants in the terrestrial acidification, freshwater eutrophication and freshwater ecotoxicity categories, respectively.

Table 3.10 Most relevant elementary flow, compartment, main life cycle phase, and main processes for each impact category.

Impact category	Elementary flow	Compartment	Main LC phase	Main process RO	Main process EGW
Carbon footprint	Carbon dioxide, CO_2	Air	Water treatment	Electricity (92.71%)	Chemicals, organic (55.64%)
Stratospheric ozone depletion	Dinitrogen monoxide, NO_2	Air	Water treatment	Electricity (89.65%)	Chemicals, inorganic (40.44%)
Ozone formation, Terrestrial ecosystems	Nitrogen oxides, NO_x	Air	Water treatment	Electricity (95.55%)	Chemicals, organic (50.33%)
Terrestrial acidification	Sulfur dioxide, SO_2	Air	Water treatment	Electricity (96.20%)	Electricity (54.83%)
Freshwater Eutrophication	Phosphate, PO_4^{3-}	Groundwater	Water treatment	Electricity (91.18%)	Electricity (53.36%)
Freshwater Ecotoxicity	Copper, Cu	Groundwater	Water treatment	Electricity (94.44%)	Electricity (67.79%)

Figure 3.2 shows the contributions to all impact categories of the water treatment phases of the RO and EGW technologies. EGW had lower environmental impacts than RO for all impact categories. The greatest impacts of the RO water treatment phase were registered in the acidification (76%), eutrophication (76%) and ecotoxicity (80%) categories, due to the emissions to air and groundwater, mainly caused by the electricity consumed (Table 3.10). Organic and inorganic chemical compounds added in the water treatment phase in the EGW technology (Table 3.6) were the inputs accounting for its greatest impacts in the carbon footprint and stratospheric ozone categories (contributing approximately 31%).

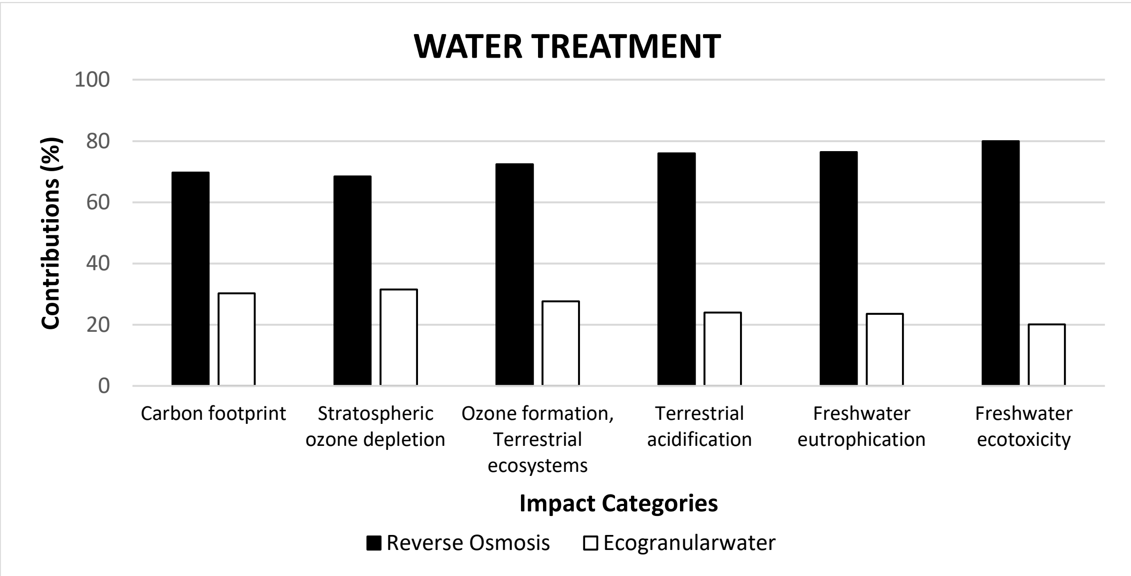


Fig. 3.2 Contributions of the technologies to selected impact categories for the water treatment phase.

Conversely, in the structure phase, EGW technology contributed more than RO for all impact categories, with values ranging from 91% in freshwater ecotoxicity to

98% in stratospheric ozone depletion. The large amount of materials used in the installation of the EGW plant was the main cause of the high impacts in all categories (Table 3.3). Specifically, GFRP—the material used in biological reactors 2 and 3—made the greatest contribution in terms of the emissions produced in most categories, with maximum values of 97% and 70% in the stratospheric ozone depletion and ozone formation categories, respectively.

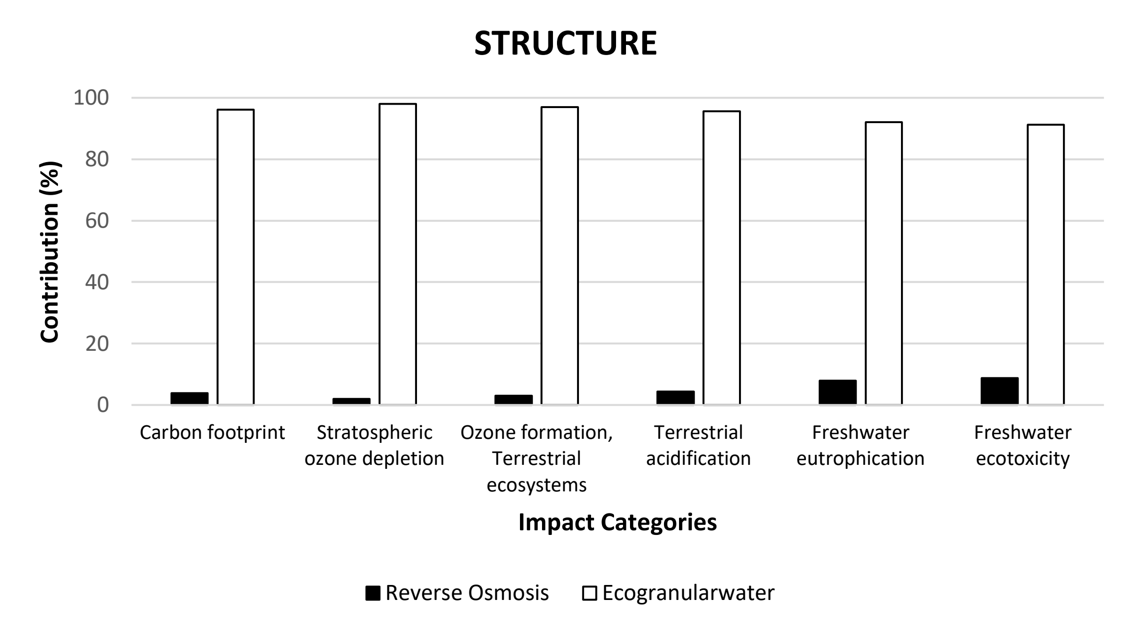


Fig. 3.3 Contributions of the technologies to selected impact categories for the structure phase.

3.5.2 Economic impact assessment

The first stage of this assessment entailed estimating the financial costs associated with the production of 1 m3 of water with the two technologies (Table 3.11). The EGW technology required more human resources, implying higher staff costs. However, it had lower energy consumption and lower costs for reagents. Additionally, there was no cost associated with the use of membranes. Taken together, the financial costs of producing 1 m3 of water were 30% lower than the costs of producing the same volume with RO.

Table 3.11 Operating costs associated with the production of 1 m3 of water.

Items	€/m^3		Percentage structure (%)	
	RO	EGW	RO	EGW
Staff	0.0893	0.2228	7.69	27.45
Energy	0.3856	0.0701	33.19	8.64
Chemicals	0.4231	0.3675	36.41	45.28
Membranes	0.1117	0.0000	9.61	0.00
EAC of the treatment plant	0.1522	0.1512	13.10	18.63
TOTAL	1.1619	0.8116	100.00	100.00

EAC: Equivalent Annual Cost

In a second stage, the environmental and resource costs were incorporated into the analysis (Table 3.12). As mentioned above, these concepts reflect the costs due to the raw water rejected in the production process. In the case of the EGW technology, the volume of reject water was negligible and had no environmental impact. However, the high percentage of reject water in RO means substantial environmental and opportunity costs, widening the differential in the cost of producing 1 m^3 of water with the two technologies. Following the indications of the Water Framework Directive, the cost of producing 1 m^3 of water was estimated as 43% lower with the EGW technology than with RO.

Table 3.12 Operating costs and structure costs associated with the production of 1 m^3 of water.

Items	€/m^3		Percentage structure (%)	
	RO	EGW	RO	EGW
Staff	0.0893	0.2228	6.20	27.15
Energy	0.3856	0.0701	26.78	8.54
Chemicals	0.4231	0.3675	29.39	44.78
Membranes	0.1117	0.0000	7.76	0.00
EAC of the treatment plant	0.1522	0.1512	10.57	18.43
Opportunity cost	0.1464	0.0009	10.17	1.10
Environmental costs	0.1315	0.0000	9.13	0.00
TOTAL	1.4398	0.8206	100.00	100.00

EAC: Equivalent Annual Cost

3.6 Discussion

Different physical, chemical and biological water treatment technologies have been developed to solve the problem of groundwater contamination. Notable examples of physical and chemical systems include adsorbent material systems (Awual et al., 2015; Awual, 2019; Awual et al., 2019), electrodialysis (Martínez et al., 2017) and reverse osmosis (Epsztein et al., 2015). Biological treatments include submerged biofilters (Zeng et al., 2019) and aerobic granular systems. These granular systems do not require any type of support on which to develop the biomass, as the granules are made up solely of biomass (Hurtado-Martínez et al., 2021b).

This biological technology was selected for the pilot due to its specific characteristics: it is an inexpensive technology, easy to implement in small municipalities and environmentally friendly.

Environmental impact assessment

The EGW technology had lower environmental impacts per m3 than RO in all impact categories. Specifically, the carbon footprint, stratospheric ozone depletion, ozone formation, acidification, eutrophication and ecotoxicity impact categories were lower by 39%, 31%, 44%, 51%, 52% and 59%, respectively. The results showed that the highest environmental impacts with the RO technology occurred in the water treatment phase (Table 3.9) due to energy consumption in this phase (Table 3.1). Our results are in accordance with those of Muñoz and Rodríguez Fernández-Alba (2008), Vince et al. (2008), Qiu and Davies (2011), Tarnacki et al. (2011), Zhou et al. (2011) and Lawler et al. (2015). Ortiz et al. (2007) concluded that replacing fossil fuels with renewable energy will substantially reduce the environmental load. Achieving a shift in the Spanish energy mix toward more renewable energies and certified high quality electricity supply may be one way of reducing the environmental impact of water treatment plants (Vince et al., 2008; Meneses et al., 2010). Tarnacki et al. (2011) highlighted the need for research on energy efficiency, use of renewable energy sources or use of waste heat. The water treatment phase in the EGW technology produced high impacts in the carbon footprint, stratospheric ozone depletion and ozone formation categories due to the inorganic and organic chemical compounds added in this phase (Table 3.6); therefore, other chemicals should be tested to find ones that generate smaller environmental impacts.

The water treatment phase was far more important than the structure phase in RO, with the former accounting for 99.9% of the impact in all categories, while EGW registered values of 98% in most categories. Similar results were found by Raluy et al. (2005) and Muñoz and Rodríguez Fernández-Alba (2008), who concluded that the operational phase of water production in desalination plants was responsible for 98-99% and 96-99%, respectively, of the overall life cycle impact. Likewise, Bonton et al. (2012) indicated that the impacts of the operational phase were 3 to 9 times greater than those of the construction phase. Therefore, the main contribution to the overall environmental impact of RO and EGW technologies came from the water treatment (Table 3.9), while the structure phase had an almost negligible impact in comparison. This finding is in line with Friedrich (2001) and Raluy et al. (2005), who reported minor impacts in the construction phase, with values of less than 5% and 15%, respectively.

EGW technology contributed more than RO in the structure phase for all impact categories (Figure 3.3). This was due to the greater weight of the frame, requiring more materials, mainly in the biological reactors (Table 3.3). Therefore, recycled materials and/or materials with a longer useful life (mainly for plastic materials) should be employed in the structure.

3.6.1 Economic impact assessment

It was estimated that to meet the cost recovery objective, rates would have to rise by 60% (Alguacil-Duarte et al., 2020). A complementary contingent valuation analysis was carried out to estimate the population's willingness to pay to help ensure the financial balance of the service. In the best-case scenario, it was estimated that the population would only accept a 20% increase in the price of water and that the resulting cost recovery rate would be 71%. With the EGW technology, considering only the financial costs (and not the environmental and resource costs, which are not accounted for in the municipal budget), it was estimated that the cost recovery rate would be 85%. In the case of Torre-Cardela, implementing a progressive rate increase while also installing EGW technology for water treatment would enable the town council to get closer to the target of cost recovery.

The scenario under study in the municipality of Torre-Cardela is not an exception in Spain. It is just one of many municipalities with a small population, dealing with a financial deficit in the provision of public water services (García-Rubio and González-Gómez, 2020). To a large extent, this deficit emerged because the town was unable to harness the important economies of scale associated with the industry. But it is also the result of the excessive investment of public funds targeted at modernizing and improving the water service seen since the second half of the 1990s (García-Rubio and González-Gómez, 2020). Many of these investments were made without any consideration that, once the infrastructure had been built, the municipalities would have to cover the costs of the service. Moreover, it should be noted that the resident population has, on average, low purchasing power. They are rural population centres with low average incomes. This combination of factors means that the water service in many municipalities is implicitly subsidized. The income obtained from other budget items ends up financing part of the costs for the water supply service.

3.7 Conclusions

The study focused on a standardized and widely used technology, reverse osmosis (RO), and a new biological nitrate removal technology based on aerobic granular sludge, named ecogranularwater (EGW). To compare the two technologies, an environmental impact study was carried out, using the life cycle assessment technique, as well as an economic impact study using a cost-effectiveness analysis. This new biological treatment has been implemented for the first time on an industrial scale through a pilot plant that supplies nitrate-free water to a Spanish municipality.

The RO technology registered higher environmental impacts than the EGW technology in all impact categories, while the cost of producing 1 m3 of water was lower with the EGW technology. This was because RO requires higher energy use in the water treatment phase than the biological technology does. The use of renewable energy sources could be an effective to reduce the environmental impacts. Efforts to further lessen the environmental impact of the EGW technology should seek to reduce the impact of the structure using recycled materials and/or materials with a longer useful life.

Our study showed that:

- The EGW technology produces drinking water in a more environmentally-friendly, cost-effective way, and with lower energy costs.
- The EGW technology is more appropriate than conventional technologies in rural municipalities with nitrate pollution problems.
- The new biological technology contributes to the transition to a green economy and complies with European legislation regarding drinking water quality.
- The decision-making about the technological innovations needed in drinking water treatments to remove nitrates can be supported by environmental and economic studies.

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CAPITULO 4. Proposal of an index to evaluate the 'dewaterization' of the urban water cycle and a practical application

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4.1 Abstract

Analyses of the relationship between water and energy do not account for the fact that the energy used in the urban water cycle is a consumer of water. To ensure the efficient use of water resources, the operator must know the raw water use associated with the energy input of the urban water infrastructure. The main contribution of this research is the proposal of an index that measures how much raw water is consumed by the energy used to produce one cubic metre of water. The resulting index is a decision-making tool that enables the sustainable use of water resources. This article first explains the index, which is called the Water Footprint of the Urban Water Cycle. It then provides examples of how to apply the proposed method; among other applications, it can be used to establish a classification of energy sources based on their relative consumption of raw water, according to the electricity generation mix in each service area. The proposed method is useful for operators, policymakers and other stakeholders, enabling them to make decisions that contribute to the "dewaterization" of the urban water cycle.

4.2 Keywords

Dewaterization, WFUWC, Water-Energy Nexus, Water Footprint, Urban Water Cycle.

4.3 Introduction

This article presents a novel method for quantifying the interdependence between water and energy in the urban water cycle (UWC). The proposed index, which is called the Water Footprint of the Urban Water Cycle (WFUWC), measures the volume of water used in the analysed facilities due to the electrical energy consumed. This connection is referred to as the water-energy-water nexus (see Figure 4.1).

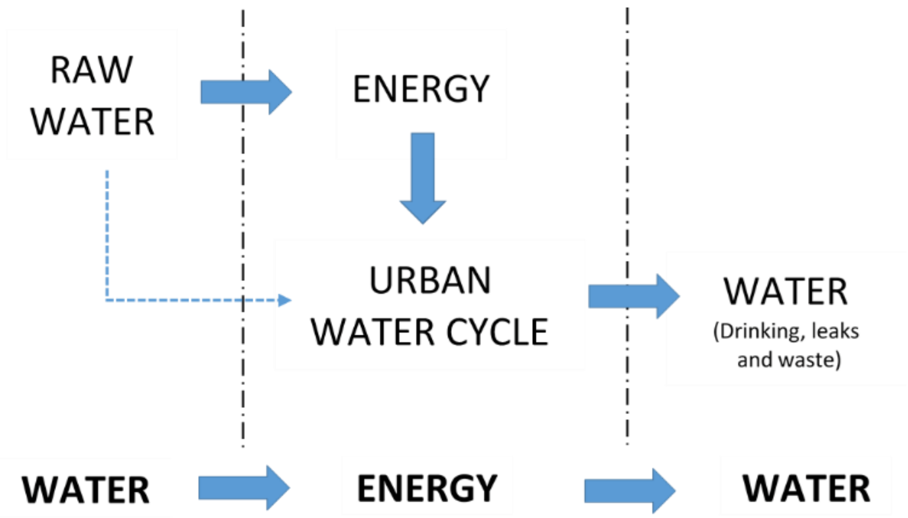

Fig. 4.1 The relationship between water and energy in the urban water cycle.

Previous studies have clearly demonstrated and explained in detail the relationship between water and energy (Hoekstra and Mekonnen 2012; Heshmati et al. 2015; He et al. 2019; Gerbens-Leenes et al. 2020). Nevertheless, the lack of simple tools to measure it means that it is not assessed. Such a tool is therefore needed for the management and planning of water resources (Javadinejad et al. 2019b), energy sources and related infrastructure.

The two basic elements of the method proposed here are the water used in electricity generation and the electricity consumed in the production of water. Energy use has a major environmental impact (Dincer 1999). Global energy consumption accounts for two-thirds of greenhouse gas emissions (European Environment Agency 2017) and is the primary cause of climate change (Heshmati et al. 2015; Javadinejad et al. 2019a). Over the years, various authors have identified and analysed other impacts linked to the use of energy (Dincer 1998; Dincer 1999; Bilgen 2014), including those specifically linked to the UWC (Venkatesh and Bratteboe 2011; Amores et al. 2013).

The UWC is energy-intensive and the main source of that energy is electricity (Kenway et al. 2008; Venkatesh and Bratteboe 2011; Lemos et al. 2013; Elias-Maxil et al. 2014; Loubet et al. 2014; Oppenheimer 2014; Wakeel and Chen 2016; Al-Omari et al. 2022). In the specific case of wastewater treatment plants (WWTPs), electricity has been identified as the main cause of their environmental impact (Gallego et al. 2008; Shao and Chen 2013; Zappone et al. 2014; Capodaglio and Olsson 2020), accounting for as much as 95.85% of the total impact (Pasqualino et al. 2009; Li et al. 2013; Morera et al. 2016).

Recent studies have employed different methodologies and scopes of analysis to spotlight the relationship between water and energy (Hamiche et al. 2016; Dai et al. 2018; Fayiah 2020; Gerbens-Leenes et al. 2020; Helerea et al. 2023). However, unlike other studies that analyse the impact of the UWC in terms of the volume of water used for each unit of energy consumed, the novel method proposed here goes one step further in examining the water-energy nexus, by determining the Water Footprint (WF) for each unit volume of water involved in the functioning of the UWC.

Efforts to improve the efficiency of water resource management in the UWC have traditionally focused on reducing network losses and encouraging consumers to use water more efficiently. A novel aspect of the proposed index is that it measures the volume of water that is used (consumed or polluted) by the UWC itself, highlighting the role played by the operation of the UWC in saving water.

The calculation of the proposed index starts with the WF (Hoekstra and Hung 2002; Hoekstra 2003; Hoekstra and Chapagain 2007; Aldaya et al. 2011; Hoekstra and Mekonnen 2012), which is the volume of freshwater appropriated to produce a product. In this case, the product is the electrical energy used in the UWC. The electricity generation mix determines the impact of the energy used (Amores et al. 2013) and therefore the WF. In a joint analysis, the aim of reducing the WF may come into conflict with the goal of cutting carbon emissions (Mekonnen et al. 2016; Bello et al. 2018). In this respect, the proposed index is a decision-making tool that can be used to help reconcile the two objectives. The second step of the calculation requires data on the electricity consumption per unit volume of water treated in the facilities.

Another novel contribution of this article is the concept of "dewaterization", which can be understood as the process of reducing the use of water in an economic activity. Achieving dewaterization is essential to help balance the uses of water resources.

For illustrative purposes, we demonstrate how the method can be applied to two types of UWC facilities: WWTPs and a reverse osmosis (RO) plant. The examples show the effect of the different electricity mixes on the calculation of the proposed index and its evolution over a 10-year period.

4.4 Methodological proposal

Below we present the proposed method for estimating the impact of the UWC on water resources, based on the WF of the electricity generation mix and the energy consumption per unit volume of water treated in the UWC or part of its facilities. The end result is an index called the Water Footprint of the Urban Water Cycle (WFUWC).

The starting point of the proposed method is the annual electricity generation mix in the country or region where the UWC facilities are located. Based on this information and applying the method developed by Mekonnen et al. (2015), we calculate the WF of the different energy sources that make up the electricity generation mix. From that point on, the rest of our method is entirely novel. As such, it represents the main contribution made by this research to the current body of knowledge.

The second step is to collect data on the flows of water involved in the processes of abstraction, water treatment, transport, distribution, use, sewage collection and wastewater treatment. Exactly what data are collected will depend on whether the aim is a partial or full evaluation of the UWC.

The last data requirement is the electricity consumption of the analysed facilities. To evaluate a future scenario in which the facilities are not yet in operation, the consumption will have to be estimated.

4.4.1 WF of electricity generation

Each source of electricity has a different WF, which we calculate using the method proposed by Mekonnen et al. (2015). The estimation of the total WF in m^3/year, corresponding to the fuel supply and construction stages—together regarded as the supply chain—plus the operational stage, is formulated by Mekonnen et al. (2015) as follows (Eq.1):

$$WF = WFsupplychain + WFoperation \quad (1)$$

Where: WF is the water footprint of electricity and heat production (WF in m^3 per year), $WF_{supplychain}$ is the water footprint of the supply chain and $WF_{operation}$ is the operational water footprint.

The WF corresponding to the electricity generated from fossil fuels, nuclear energy and biomass is calculated by Mekonnen et al. (2015) as follows (Eq.2):

$$WFe,total[f] = WFh,f\,[f] + FEE[f] + (WF_{e,c}[f] + WF_{eo}[f])xE[f] \quad (2)$$

Where: $WF_{h,f}$ [f] is the water footprint per thermal unit of energy ($m^3TJ_h^{-1}$), FEE [f] is the annual consumption of fuel "f" needed to produce electricity (TJ_h per year), $WF_{e,c}$[f] is the water footprint linked to the construction of the power plant per unit of electricity produced over the useful life of the plant ($m^3TJ_h^{-1}$), $WF_{e;o}$[f] is the water footprint corresponding to the operation of the plant per unit of electricity produced by fuel "f" ($m^3TJ_h^{-1}$), and E[f] is the annual production of electricity from fuel "f" (TJ_h per year).

Since all the other renewable energies apart from biomass are not fuel-based sources, the WF is calculated using the following expression (Eq.3) given by Mekonnen et al. (2015):

$$WFe,total[r] = (WFc[r] + WFo[r]xE[r] \quad (3)$$

Mekonnen et al. (2015) use these operations to obtain the WF data for each source of energy, as shown in Table 4.1:

Table 4.1 The global consumptive WF per unit of electricity output for different energy sources, with reference to the regional specifications established by Mekonnen et al. (2015).

Fuel	$Wf_{e(m^3 TJe^{-1})}$=Fuel supply+ Construction+ Operation
Coal	79–2100
Lignite	93–1580
Conventional oil	214–1190
Unconventional oil (oil sand)	419–1340
Unconventional oil (oil shale)	316–1830
Natural gas	76–1240
Shale gas	81–1270
Nuclear	18–1450
Firewood	48000–500000
Hydropower	300–850000
Concentrated solar power	118–2180
Photovoltaic	6.4–303
Wind	0.2–12
Geothermal	7.3–759

In our case, we calculate the WF of the electricity generation mix using Eq. 4, below, which takes the WF data from Table 4.1 for each of the energy sources, their production and the annual amount of electrical energy generated:

$$WF_t(m^3/_{TWh}) = \frac{\sum_{F=1,e=1}^{n}\left[E_F(TWh)\cdot WF_e(m^3/_{TWh})\right]}{E_a(TWh)} \quad (4)$$

Where: WFt is the total water footprint of the electricity mix of the analysed country or region; E_F is the energy generated from each of the sources in the mix; WF_e is the water footprint of each of the energy sources (Mekonnen et al. 2015) and E_a is the total energy produced annually in the analysed country or region.

4.4.2 Electricity consumption ratio in the UWC

We use the water flow data from the UWC facilities and the electricity consumption data to calculate a ratio indicating the amount of electrical energy used per unit volume of water involved (extracted, pumped, purified, supplied, collected, etc.) in the process under analysis.

The source of the data may vary depending on the objective and scope of the calculation of the WFUWC index: primary data, such as direct measures of energy consumption; secondary data, based on the installed capacity of a facility; or a tertiary source of data, such as the consumption ratios of similar facilities (Mizuta and Shimada 2010; Guo et al. 2014; Trapote et al. 2014; Gu et al. 2017).

4.4.3 WFUWC index

Using the data on the WF of the electricity generation mix of the country or region under study, together with the data on energy consumption per unit volume of water treated in the whole UWC or part thereof, we calculate the WFUWC index using Eq. 5:

$$WFUWC_i\left(m^3/_{m^3}\right) = WF_t\left(m^3/_{TWh}\right)\cdot\sum_{i=1}^{n} C_i\left(TWh/_{m^3}\right) \quad (5)$$

Where: $WFUWC_i$ indicates the WF of each cubic metre of water processed in facility i (facility or the entire UWC); WF_t represents the total water footprint of the electricity mix of the analysed country or region; and C_i is the electricity consumption per cubic metre of the analysed facilities.

To help interpret the result of Eq. 5, we provide a classification of the WFUWC index values (Table 4.2).

Table 4.2 Range of values for the WFUWC index

Classification	l/m^3
Excellent	<10
Very good	10-25
Good	25-50
Fair	50-100
Poor	100-250
Very bad	>250

4.5 Example of an application of the proposed method

To illustrate how the WFUWC index (Eq. 5) works and how it might be useful, we apply it to two types of UWC facilities. We thus calculate the WFUWC index for six WWTPs and for an RO plant. Given that both types of facilities are aimed at improving the quality of treated water, the index captures the depletion of water resources due to a treatment used to improve water quality.

4.5.1 WF for the electricity mix in Spain

To calculate the WF of the electricity generation mix in Spain, we use Eq. 4, which consists of two main parameters: the electricity generation data (European Commission, Directorate-General for Energy 2020); and the WF presented in Table 4.1 for each of the energy sources (Mekonnen et al. 2015).

For this case, only the electricity generated in Spain is included in the analysis, as the average share of imports in the analysed period is minimal (5.24%). Additionally, imports are offset by electricity exports during this period. If this were not the case, the WF of electricity imports could be calculated and weighted according to their percentage share in the mix.

In Spain in 2019, coal, lignite, oil, natural gas, nuclear, hydropower, solar thermal, solar photovoltaic and wind accounted for 97.59% of all energy sources. Some energy sources are omitted from the calculation (European Commission, Directorate-General for Energy 2020)—namely, solid biofuels and renewable wastes, biogases, liquid biofuels, tide, wave and ocean and non-RES waste—as they are not directly addressed in the paper by Mekonnen et al. (2016). We also omit shale gas, firewood and geothermal as they are not used for electricity generation in Spain, according to published data (European Commission, Directorate-General for Energy 2020).

To ensure a sufficiently representative figure for the WF of the electricity mix, we use electricity data from the last 10 years (2010-2019). This addresses the potential issue of specific annual variations caused by factors such as economic conditions, fuel prices, weather conditions, international conflicts or the level of energy dependence, which may significantly influence the result.

Table 4.3 shows the WF of each of the energy sources used in the electricity mix in Spain, the total electricity generated annually, and in the last row, the WF per unit of electricity generated (Eq. 4). The WF data are in l/kWh to provide a figure that is suitable for the scale of this analysis.

Table 4.3 WF (m3) for the electricity mix in Spain.

Fuel	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Coal	7.16E+07	1.19E+08	1.55E+08	1.12E+08	1.22E+08	1.43E+08	1.03E+08	1.27E+08	1.06E+08	3.55E+07
Lignite (brown coal)	2.64E+06	8.22E+06	6.21E+06	4.61E+06	6.03E+06	6.66E+06	3.79E+06	5.27E+06	3.57E+06	1.91E+06
Conventional oil	4.59E+07	4.07E+07	4.24E+07	3.81E+07	3.91E+07	4.77E+07	4.68E+07	4.36E+07	4.01E+07	3.57E+07
Unconventional oil (oil sand)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Unconventional oil (oil shale)	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Natural gas	9.87E+07	8.93E+07	7.64E+07	6.07E+07	5.02E+07	5.54E+07	5.55E+07	6.72E+07	6.12E+07	8.74E+07
Shale gas	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Nuclear	1.95E+08	1.82E+08	1.93E+08	1.78E+08	1.80E+08	1.80E+08	1.84E+08	1.83E+08	1.75E+08	1.84E+08
Firewood	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Hydropower	1.14E+10	8.23E+09	6.04E+09	1.03E+10	1.07E+10	7.85E+09	9.97E+09	5.27E+09	9.20E+09	6.72E+09
Concentrated solar power	3.56E+06	6.61E+06	1.37E+07	1.69E+07	1.76E+07	1.78E+07	1.75E+07	1.84E+07	1.62E+07	1.02E+07
Photovoltaics	2.20E+06	2.60E+06	2.54E+06	2.52E+06	2.63E+06	2.67E+06	2.64E+06	2.79E+06	2.49E+06	4.24E+06
Wind	1.91E+05	1.85E+05	2.14E+05	2.40E+05	2.25E+05	2.13E+05	2.11E+05	2.12E+05	2.20E+05	2.40E+05
Geothermal	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
Total WF (m^3)	1.18E+10	8.68E+09	6.53E+09	1.07E+10	1.12E+10	8.30E+09	1.04E+10	5.72E+09	9.61E+09	7.08E+09
Total electricity generation (TWh)	296.69	288.27	291.69	279.15	272.65	274.16	268.24	268.79	267.43	266.54
Electricity generation WF (l/kWh)	39.78	30.11	22.40	38.26	40.95	30.27	38.71	21.27	35.93	26.56

Figure 4.2 presents the data from Table 4.3, distinguishing between the renewable energy sources (firewood, hydropower, concentrated solar power, wind, and geothermal) and the non-renewable energy sources (coal, lignite, conventional oil, unconventional oil, natural gas, shale gas, nuclear). In addition, the annual WF data for electricity generation are represented in bars along with the corresponding trend line.

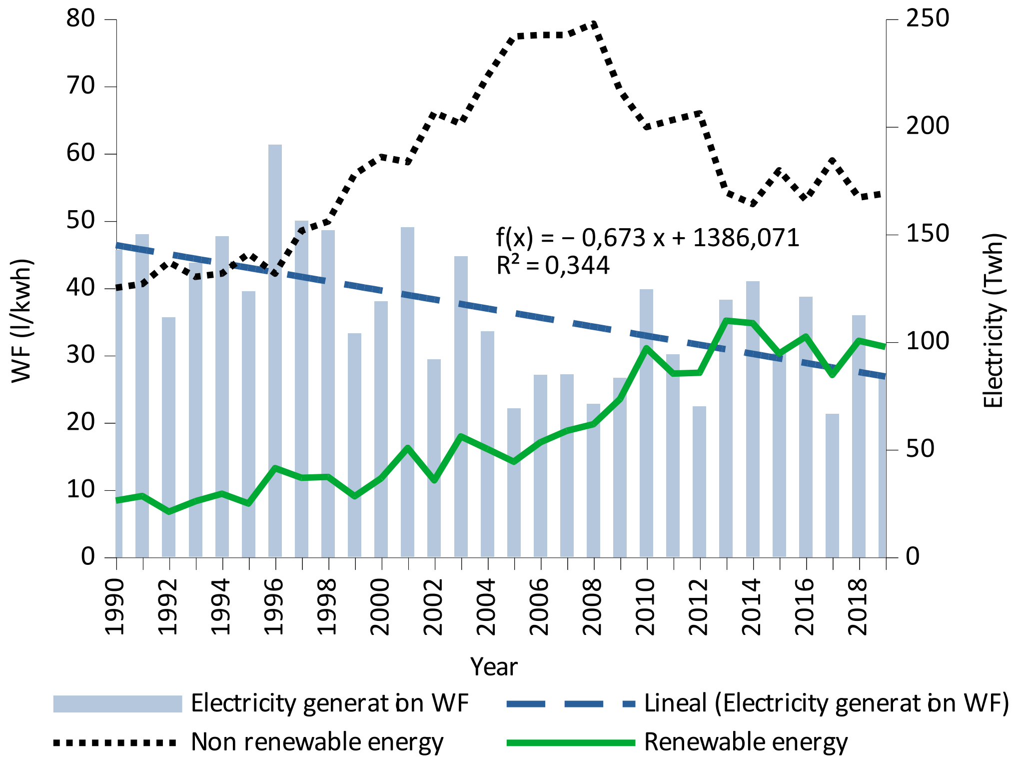

Fig. 4.2 Electricity generation mix in Spain and the evolution of its WF.

4.5.2 Electricity consumption ratio figures

Annual operational data have been collected on both electricity consumption and the flow of treated water for six WWTPs (Table 4.4) and an RO plant.

In the case of the WWTPs, the values lie within the range of energy intensity reported in other articles that analyse the use of energy by this type of facility in Spain (Trapote et al. 2014) and in various other countries (Panepinto et al. 2016; Wakeel et al. 2016).

Table 4.4 WWTP features.

WWTP	WWTP1	WWTP2	WWTP3	WWTP4	WWTP5	WWTP6
Type	Trickling filter	Trickling filter	Sequencing batch reactor (SBR)	Stahlermatic batch reactor	Extended aeration	Biodisc
Observations	-	-	-	-	-	Influent pump station
Population equivalent (design)	1092	5460	249	1130	526	2146
Population equivalent (current)	741	5444	193	860	196	1872
Electricity consumption ratio (kWh/m³)	0.4016	0.2956	0.15097	0.6743	2.491	0.322

As for the RO plant, the data are from a small standard plant which has two osmotizers with a production flow of 8 m^3/h. Under current operating conditions, consumption is 2.05 kWh/m^3, which lies within the standard consumption range for these facilities (Al-Karaghouli and Kazmerski 2012; Dashtpour and Al-Zubaidy 2012).

4.5.3 Calculation of the WFUWC index

Using the data on the WF per kWh of the electricity generation mix in Spain (Table 4.3) in l/kWh and electricity consumption ratios of the WWTPs (Table 4.4), we apply Eq. 5 to calculate the WFUCW index. The results are shown in Table 4.5.

Table 4.5 WFUWC for the WWTPs.

	WFWUC (l/m³)									
WWTP	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
WWTP1	15.98	12.09	9.00	15.37	16.45	12.16	15.55	8.54	14.43	10.67
WWTP2	11.76	8.90	6.62	11.31	12.11	8.95	11.45	6.29	10.62	7.85
WWTP3	6.10	4.62	3.43	5.87	6.28	4.64	5.93	3.26	5.51	4.07
WWTP4	26.82	20.30	15.10	25.80	27.61	20.41	26.11	14.34	24.23	17.91
WWTP5	99.09	75.00	55.79	95.32	102.01	75.41	96.44	52.98	89.51	66.17
WWTP6	12.82	9.70	7.22	12.33	13.20	9.76	12.48	6.85	11.58	8.56

Figure 4.3 depicts the WFUWC index of each of the WWTPs (Table 4.5).

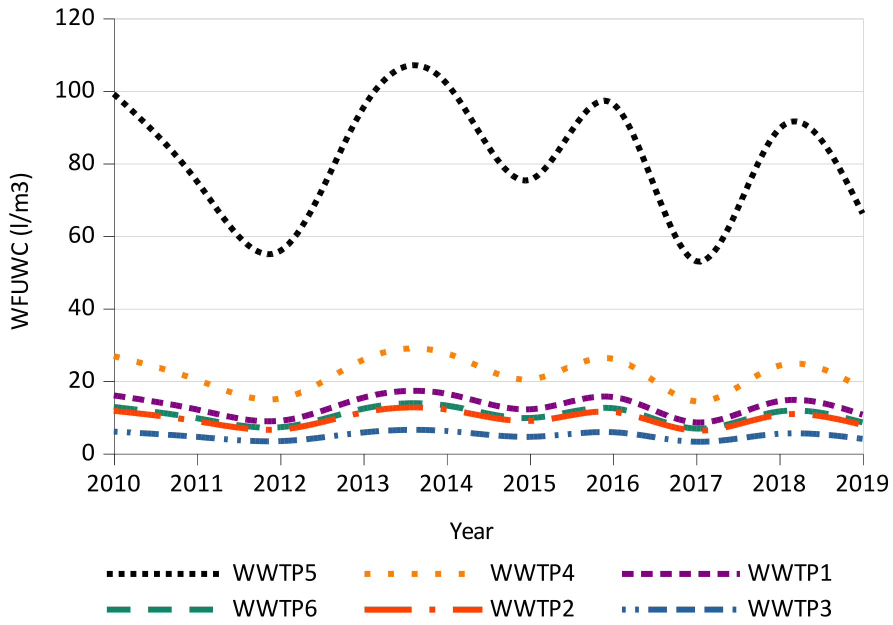

Fig. 4.3 WFUWC values for the WWTPs used as an example, calculated with the electricity generation mix in Spain.

The results of the WFUWC index calculated for the RO plant are shown in Table 4.6. In this case, the WFUWC index refers only to the impact on water resources of the gate-to-gate electricity consumption by the osmosis plant; it does not account for the reject water generated by the process nor the energy used to pump it.

Table 4.6 WFUWC for the RO plant.

WFWUC (l/m³)									
2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
81.55	61.72	45.91	78.44	83.95	62.06	79.36	43.60	73.67	54.45

Figure 4.4 depicts the WFUWC index calculated for the RO plant (Table 4.6) together with its trend line.

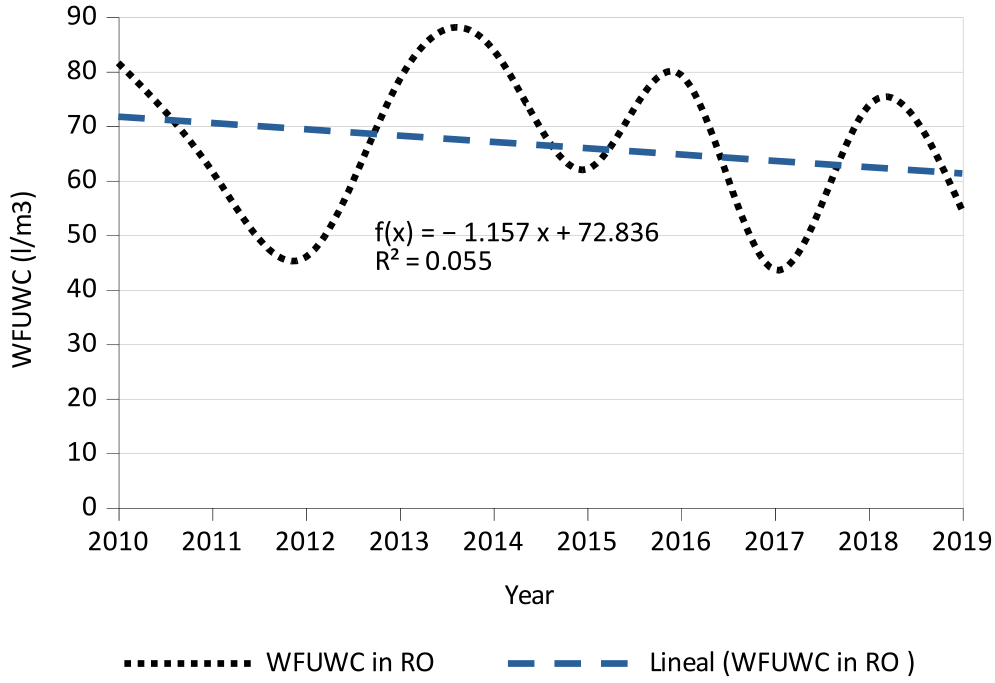

Fig. 4.4 WFUWC values for the RO plant example calculated with the electricity generation mix in Spain.

4.5.4 Calculation of the WFUWC index with the electricity generation mix in other countries

A facility or process can also be evaluated by applying different electricity generation mixes in the calculation of the WFUWC index.

By way of example, we take the electricity generation mix in Austria, Denmark and the European Union average (EU27) (European Commission, Directorate-General for Energy 2020). First, we calculate the WF for each one (Eq. 4). Table 4.7 shows the WF of the electricity generation mix in each country and the EU27.

We then apply these results (Table 4.7) to the facilities analysed earlier. Figure 4.5 shows the results of the WFUWC index. This calculation does not account for countries' electricity imports and exports.

Table 4.7 WF of the electricity generation mixes in Austria, Denmark and the EU27.

	WF (l/kWh)									
Country	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Austria	157.92	156.29	178.03	182.04	186.22	169.71	171.63	160.91	164.09	159.93
Denmark	1.97	1.77	1.62	1.72	1.5	1.22	1.42	1.12	1.16	0.79
EU27	33.6	29.03	31.34	34.57	35.8	32.52	33	28.64	32.67	31.03

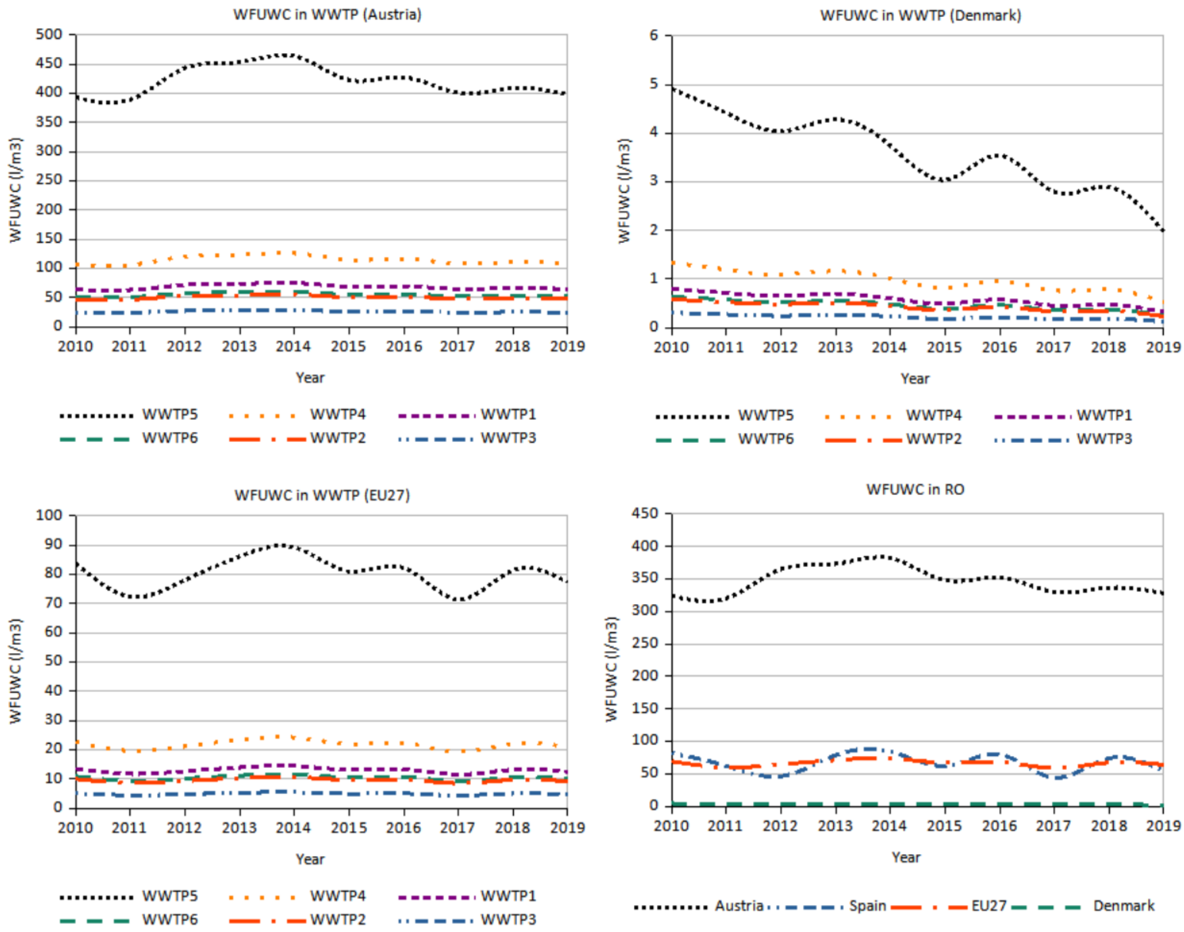

Fig. 4.5 WFUWC values for the WWTPs and the RO plant calculated using the electricity generation mix in different countries and the EU27.

4.6 Discussion

The results show the WFUWC index expressed in l/m^3 for the six WWTPs and the RO plant for the period 2010-2019, applying the electricity generation mix in Spain (Tables 4.5 and 4.6). Figures 4.3 and 4.4 graphically depict the values of the WFUWC index for each of the analysed plants.

WWTP 5 shows a much higher impact than the rest. Its WFUWC index is 102.01 l/m^3 (Table 4.5), which indicates that 102.01 litres of water are needed solely for the electrical energy consumed in the treatment of 1000 litres of water. That is, the index does not take into account the WF caused by the effluent. WWTP 5 is classified as poor on the scale proposed in Table 4.2.

The RO plant also registers notably high WFUWC index values. In 2014, the maximum index value is 83.95 l/m^3 (Table 4.6). Based on its WFUWC, it is classified as fair (Table 4.2).

Among other reasons, the differences in the WFUWC index can be explained by the fact that electricity consumption may vary depending on the location of the facilities—due to differences in altitude and in the local climate—and also depending on operational efficiency and the size, state and age of the facilities (Morera et al. 2016).

The application of different electricity generation mixes (Table 4.7) in the facilities points to several practical uses of the WFUWC index. For example, it enables an evaluation of the use of water over a period of time and in different locations (see Figure 4.5).

The data in Figure 4.5 reach maximum values in 2014 for Austria and the EU27, with values of 463.87 l/m^3 and 89.18 l/m^3, respectively, and in 2010 for Denmark, with 1.97 l/m^3. In the case of the RO plant, Austria registers a value of 381.75 l/m^3 in 2014.

The large difference between countries is explained by the relevance of the electricity generation mix in the WFUWC index. As can be seen, Austria registers very high values in 2014, with a WFUWC index of 463.87 l/m^3 for WWTP5 and of 381.75 l/m^3 for the RO plant; these values are classified as very bad according to the proposed categories in Table 4.2. These results are close to the threshold at which the process of treating water generates the same impact as the treated water (WFUWC = 1 m^3/m^3). Austria registers such high values because its electricity generation is mainly hydroelectric, which has a large WF. Conversely, in Denmark—with maximum WFUWC index values of 4.91 l/m^3 for WWTP5 and 1.64 l/m^3 for the RO plant—wind energy predominates, which has a very small WF.

Over the analysed period, we observe positive progress in the dewaterization of the electricity generation mix (see Figures 4.2,4.4 and 4.5), due to the increased production of renewable energy. However, biomass and hydroelectricity, which are considered renewable, significantly increase the WFUWC, as can be seen in Figure 4.2 in the years 1996 and 2014. This fact is evidence of the divergence between dewaterization and decarbonization (Mekonnen et al. 2016; Vanham et al. 2019) in certain electricity generation scenarios (Gagnon and Vate 1997; Räsänen et al. 2018).

Finally, the examples we present here highlight the critical role of the electricity generation mix in the resulting WFUWC index; hence the importance of jointly planning water and energy policies (Gleik 1994; Lee et al. 2017). Given the growing global demand for electricity (IRENA 2019; IEA 2021), the application of the index in the analysis of other regions can help ensure progress in the dewaterization of the UWC, while its use in other fields can give rise to new lines of research.

4.7 Conclusion

This research sheds light on the conflict between the goals of dewaterization and decarbonisation. Although water is the main raw material of the UWC, relatively little academic attention has been paid to the use of water in the operation of the UWC, referred to as the water-energy-water nexus. The main contribution of this research is the proposed WFUWC index for measuring the volume of water used by the energy consumed to produce one cubic metre of water.

The proposed index underscores the idea that it is not only consumers who can make an effort to save water; the operation of the UWC itself plays a fundamental role in the availability of drinking water. Accordingly, the operators of the UWC must also pay attention to the water used via the energy used.

The main conclusions of this research are as follows:

- The WFUWC is an effective tool for assessing the impact of UWC infrastructure on water resources. It is simple to calculate and easily interpretable for analysis and decision-making, which makes it suitable for the purposes of communication and awareness-raising, and helpful for operators, policymakers and other stakeholders.
- It is applicable to varying types of infrastructure and different energy mixes, which makes it useful for the evaluation and planning of future UWC infrastructure. This is particularly important in areas facing water scarcity, a problem expected to become more common around the world as a result of climate change.

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CAPITULO 5. RESULTADOS Y CONCLUSIONES

5.1 Resumen de la investigación

La tesis se centra en el caso del municipio de Torre Cardela y su problemática con el suministro de agua potable que, unido a la problemática de la gestión del ciclo del agua en un pequeño municipio, presenta un problema por contaminación de nitratos en sus captaciones de agua potable. Esto supuso la declaración de No Aptitud del agua de abastecimiento para consumo humano en 1996. En 2012, se instaló una planta de osmosis inversa para eliminar los nitratos, obteniendo agua apta para consumo sin tratamiento adicional pero, en cambio, la operación de la planta supuso un notable incremento de los costes del servicio. En 2015, se inició el Proyecto LIFE-Ecogranularwater para reducir los costos del tratamiento de agua mediante una planta demostrativa de tecnología granular aeróbica secuencial.

El objetivo de la tesis estaba ligado al análisis de los aspectos socioeconómicos y ambientales de la implantación de la nueva tecnología. Se exploraron dos aspectos principales: desde el lado de la demanda, la disposición de la población a pagar más por el agua para aproximarse al principio de recuperación de costes y la comparación de las plantas de osmosis inversa y tecnología granular aeróbica secuencial en términos de costos e impacto ambiental a través del análisis de sus ciclos de vida. Además, se propuso una metodología innovadora para evaluar el impacto que sobre los recursos hídricos tiene el consumo eléctrico en la producción de agua potable, destacando la interconexión entre agua y energía en CUA.

En definitiva, la investigación aborda un problema real en el suministro de agua potable del municipio de Torre Cardela y propone soluciones prácticas, sostenibles, replicables y transferibles.

5.2 Resultados

En esta sección, se presentan los resultados de tres investigaciones con las que se abordaron diversos factores del proyecto LIFE Ecogranularwater.

En la primera investigación, expuesta en el capítulo 2, los resultados sugieren que, al menos en lo que respecta al agua para usos residenciales, no hay evidencia de que las personas vinculadas a la agricultura formen un grupo de interés que represente un obstáculo para aumentar los precios del agua para usos residenciales en áreas rurales. Además, a pesar de la baja proporción de usuarios dispuestos a pagar más por el agua en áreas rurales, lo cual se debe principalmente a los bajos niveles de ingresos, existe cierto margen para aumentar el precio del agua para usos residenciales. Las evidencias obtenidas implicarían un aumento medio hipotético en las tarifas de agua de entre el 8,7% y el 20,4%, dependiendo del modelo que se elija. A pesar de la baja proporción de usuarios dispuestos a pagar más y contrariamente a lo esperado, la población agrícola muestra una mayor disposición a pagar. Los hallazgos de este estudio sugieren que tales aumentos serían modestos, pero contribuirían a mejorar la recuperación de costos.

El análisis comparativo entre dos tecnologías de tratamiento de agua desarrollado en el capítulo 3 analiza y compara tanto aspectos ambientales como de costes de operación.

En cuanto a la evaluación ambiental, revela diferencias en los impactos asociados a la producción de 1 m^3 de agua potable. Los resultados indican que la osmosis inversa genera impactos ambientales superiores en todas las categorías evaluadas, atribuibles todas ellas al consumo de electricidad durante la fase de tratamiento del agua que es la que concentra el mayor impacto, el 99,99 % en la osmosis inversa y el 98,39% en la planta demostrativa. A modo de ejemplo, en la categoría de impacto de Huella de Carbono, la planta de tecnología granular tiene unas emisiones de 0,4 kg CO_2 eq/m^3 frente a la osmosis inversa en la que esta cantidad se eleva a 1,02 CO_2 eq/m^3, lo que supone un 60,08% más de Huella de Carbono.

Por otro lado, la evaluación económica se realiza en dos etapas. En la primera, se estimaron los costos financieros asociados a la producción de 1 m^3 de agua utilizando ambas tecnologías. La planta demostrativa requiere más recursos humanos, lo que implicaba mayores costos de personal, sin embargo, presentaba un consumo energético muy reducido y menores costos en el uso de reactivos, sin costos asociados al uso de membranas. En conjunto, los costos financieros de producir 1 m^3 de agua fueron de un 30% más bajos que los costos de producción con osmosis inversa. En una segunda etapa, se incorporaron los costos ambientales y de recursos. Mientras que la tecnología aeróbica granular secuencial no presentaba impacto ambiental por el agua rechazada, ya que no es representativa debido a su despreciable índice de rechazo, la osmosis inversa mostraba costos ambientales y de oportunidad sustanciales debido al alto porcentaje de agua rechazada. Siguiendo las directrices de la Directiva Marco del Agua, el costo de producir 1 m^3 de agua se estimó en un 43% menos con la tecnología granular aeróbica secuencial en comparación con la osmosis inversa.

Para finalizar, los resultados obtenidos en el capítulo 4 muestran la funcionalidad del índice desarrollado. Como queda patente en el análisis de ciclo de vida de la osmosis inversa expuesto en el capítulo 5, la energía es el principal insumo generador de impactos ambientales. En las aplicaciones prácticas del índice WFUWC, llevadas a cabo en la publicación relacionada con el capítulo 4, para la planta de ósmosis de Torre Cardela arroja una cifra máxima de 83,95 l/m^3 para el año 2014. Teniendo en cuenta el consumo eléctrico del análisis del ciclo de vida para ambas plantas, la nueva tecnología tendría un índice 81,81 % menor. Si se traslada al mix eléctrico de Austria se obtendría para ese mismo año 381,75 l/m^3, poniendo de relieve la importancia que el mix de generación eléctrica de cada país tiene sobre los recursos hídricos y como el consumo de energía por el ciclo urbano del agua aumenta la presión sobre este recurso.

5.3 Conclusiones generales

Las principales conclusiones obtenidas en esta tesis han sido las siguientes:

No se aprecia que exista algún grupo de interés que pudiese estar influyendo en los precios de la tarifa del agua. Desde el punto de vista de la demanda, en aras de la sostenibilidad económica del CUA y del principio de recuperación de costes, existe un ligero margen para la subida del precio del servicio. De este análisis se evidencia que, debido a la limitada explotación de las economías de escala y al coste asociado a la tecnología seleccionada para la potabilización del agua, todo ello conlleva unos gastos que resultan difíciles de afrontar por parte del municipio, sin comprometer el acceso a un servicio de agua asequible en este contexto socioeconómico.

La planta demostrativa para la eliminación de nitratos basada en una tecnología granular aeróbica secuencial, desarrollada en el proyecto LIFE 16 ENV/ES/196 ECOGRANULARWATER, muestra ventajas comparativas en varios aspectos que la hacen más adecuada para la potabilización en pequeños municipios. En general, los costes asociados son menores permitiendo adaptarse a las limitaciones económicas de estas localidades y ambientalmente, al eliminar el rechazo, contribuye a la eliminación definitiva del nitrato no agravando el problema de la contaminación con el vertido. Este es un aspecto a considerar ya que la osmosis solo elimina nitrato en el circuito del CUA. El desempeño ambiental de la nueva planta demostrativa podría mejorar aún más, en futuros desarrollos, con la selección de materiales con menor impacto. En resumen, se trata de una evolución en cuanto a la ecoeficiencia de la potabilización de aguas contaminadas por nitrato en pequeñas poblaciones.

La metodología desarrollada se muestra como una herramienta efectiva para el análisis del impacto que sobre los recursos hídricos tiene el consumo de energía eléctrica. En ocasiones, implantaciones de tecnologías intensivas en cuanto a energía para resolver problemas de escasez de agua, incrementa más la presión sobre este recurso ya que el CUA y el consumo eléctrico entran en competencia directa por el agua. El índice WFUWC (Water Footprint of the Urban Water Cycle) permite a operadores, gestores políticos y grupos de interés realizar una evaluación y planificación más consecuente para la implantación de nuevas infraestructuras en el CUA. Adicionalmente, permite avanzar en la reducción de la dependencia del agua que a través de la energía eléctrica tienen el CUA y tomar conciencia de la repercusión del impacto que el mix de generación eléctrica tiene para los recursos hídricos.

5.4 Implicaciones del trabajo de investigación para el ámbito académico

Además de las conclusiones derivadas de la investigación llevada a cabo en esta tesis, que principalmente tienen aplicaciones prácticas en la gestión del CUA, se realiza un aporte académico significativo en varios aspectos. En primer lugar, en el capítulo 2 se amplía el ámbito de análisis y evaluación de potenciales soluciones para la recuperación de costes en el CUA desde una perspectiva centrada en la demanda. Por otro lado, en el capítulo 3, la aplicación del Análisis de Ciclo de Vida para la evaluación ambiental de las dos tecnologías implementadas en Torre Cardela, refuerza la utilidad y eficacia de esta metodología como herramienta para realizar comparaciones integrales y precisas. Finalmente, en el capítulo 4, mediante el desarrollo de una nueva metodología para evaluar el impacto del consumo de energía en el CUA sobre los recursos hídricos, se introduce un nuevo enfoque para el análisis del impacto que en los recursos hídricos tiene el consumo de energía eléctrica.

5.5 Contribuciones teóricas y empíricas

La investigación desarrollada y expuesta en esta tesis proporciona una serie de contribuciones significativas a la literatura científica en áreas que incluyen la valoración contingente, análisis del ciclo de vida, análisis de coste-efectividad, huella hídrica y el nexo agua-energía. A lo largo de los capítulos 2, 3 y 4, se lleva a cabo una investigación aplicada a un problema real en el municipio de Torre Cardela, con el objetivo de evaluar y proponer soluciones desde una perspectiva científica dentro de los marcos teóricos mencionados.

Se han conseguido contribuciones teóricas y empíricas concretas:

1. Se ha proporcionado evidencia respecto a la factibilidad de que, grupos de influencia vinculados a la agricultura, el principal sector económico en los municipios rurales, puedan influir en la modificación de los precios de la tarifa del agua. Además, dentro del marco de esta investigación, se ha presentado evidencia sobre la disposición al pago que existe en este tipo de poblaciones.

2. Se incorpora de manera concreta el enfoque teórico del análisis integrado ambiental y económico. Este enfoque holístico se materializa mediante la aplicación conjunta del análisis del ciclo de vida, permitiendo una comprensión exhaustiva de las externalidades asociadas a las tecnologías utilizadas. La realización simultánea de ambos análisis es fundamental para entender los impactos concretos de la implantación o evaluación de una tecnología, destacándose el artículo publicado como un ejemplo ilustrativo de esta práctica.

3. Mediante el desarrollo de la novedosa metodología para calcular el índice WFUWC (Water Footprint of the Urban Water Cycle) y su aplicación práctica, se logran varias contribuciones destacadas. Se contribuye metodológicamente a la evaluación cuantitativa del impacto del consumo de energía eléctrica por parte del CUA en los recursos hídricos, proporcionando herramientas para el análisis del nexo agua-energía. En consonancia con lo anterior, se propone el concepto "agua-energía-agua" para resaltar la competencia generada por los recursos hídricos en la producción de agua cuando se utiliza energía eléctrica. Se formula además el novedoso concepto de "Dewaterization", relacionado con la necesidad de reducir el uso de agua asociado al consumo de electricidad en las instalaciones del CUA, con el fin de aliviar la presión sobre los recursos hídricos, que constituyen la principal materia prima del propio CUA.

5.6 Implicaciones para la práctica empresarial

El marco general de la tesis se ha desarrollado dentro del proyecto LIFE ECOGRANULARWATER (LIFE16 ENV/ES/196) con un cometido principal que era aportar una solución efectiva a la potabilización de agua de un pequeño municipio y una variable eminentemente empresarial. Uno de los objetivos más importantes a lograr era la replicabilidad y transferibilidad del proyecto. Por tanto, el análisis se ha abordado desde el inicio con el objetivo de buscar soluciones adaptadas al contexto de los pequeños municipios rurales y con una lógica de desarrollo empresarial.

Relacionada con la investigación llevada a cabo en el capítulo 2 se aporta un análisis desde el punto de vista de la demanda para que lo gestores puedan acercarse al objetivo de la recuperación de costes.

En el estudio de la planta demostrativa de tecnología granular aeróbica secuencial son varias las aportaciones a la práctica empresarial. Por un lado, es la primera vez que se evalúa esta tecnología y se compara con una tecnología de osmosis inversa que es la más común. Se pone sobre relieve el principal factor de impacto económico y ambiental para que sea considerado por los gestores a nivel de explotación y, por último, se indican los insumos y materiales con más impacto ambiental en su ciclo de vida para tender hacia nuevos desarrollos de la planta demostrativa desde una perspectiva de ecodiseño.

Con la implementación de la propuesta metodológica del capítulo 4, los gestores y responsables políticos de la gestión del agua, tienen una herramienta para evaluar el impacto de las infraestructuras actuales, planificar infraestructuras futuras, e incidir en las fuentes de energía eléctrica, todo ello con el objetivo de “dewaterize” el CUA y minimizar su competencia por los recursos hídricos.

5.7 Limitaciones y futuras líneas de investigación

La principal limitación del estudio, es que se trata de un caso relacionado con una planta de nuevo desarrollo, y, por tanto, se reduce a un municipio y a que la comparativa entre plantas se hace con una planta demostrativa. Modelos más desarrollados de ésta podrían aportar mayores ventajas.

Las futuras líneas de investigación que se abren están relacionadas con el análisis de los grupos de poder y causas objetivas de establecimiento de las tarifas de agua en pequeños municipios con gestión propia del CUA para avanzar hacia la recuperación de costes. Así mismo, profundizar con nuevos desarrollos de la planta demostrativa desde perspectivas de ecodiseño al mismo tiempo que de rentabilidad económica. Por último, la aplicación de la nueva metodología desarrollada en sectores que utilizan el agua como materia prima, como podría ser la agricultura, y, el cálculo del indicador en grandes gestores de aguas para obtener ratios comparativos, resultan campos de interés.

5.8 Consideraciones finales

La tesis recoge un novedoso análisis y conceptos derivados de la evaluación económica y ambiental realizada en el marco del proyecto ECOGRANULARWATER (LIFE16 ENV/ES/196). Se trata de una investigación aplicada cuyo principal objetivo era aportar evidencia científica para obtener una visión real de la implantación de una nueva tecnología de potabilización de agua. El principal objetivo fue ofrecer una solución a la problemática de pequeños municipios, y además de esta meta se han conseguido otras que aportan conocimiento y campo de trabajo al sector del CUA actual y futuras investigaciones académicas.